UNIVERSITY OF CALIFORNIA

Santa Barbara

Fine-Scale Analyses for Improving Conservation and Sustainability Efforts in Agricultural Landscapes of Neotropical Savannas

A dissertation submitted in partial satisfaction of the

requirements for the degree Doctor of Philosophy

in Geography

by

Fernanda de Figueiredo Ribeiro

Committee in charge:

Professor Dar Roberts, Co-chair

Professor Frank Davis, Co-chair

Professor Kelly Caylor

Laura Hess, Ph.D.

September 2020

The dissertation of Fernanda de Figueiredo Ribeiro is approved.

Laura Hess

Kelly Caylor

Frank Davis, Committee Co-chair

Dar Roberts, Committee Co-chair

August 2020

Fine-Scale Analyses for Improving Conservation and Sustainability Efforts in

Agricultural Landscapes of Neotropical Savannas

by

Fernanda de Figueiredo Ribeiro

Acknowledgements

The work I developed in this dissertation was only possible thanks to substantial support and encouragement from several people who I am deeply grateful, including professors, researchers, staff members, colleagues, friends, and family.

First, I would like to thank my advisors, Dar Roberts and Frank Davis, for your substantial support and advice throughout these past six years of my graduate studies -- I have learned an immense amount from both of you! Dar, thanks for the opportunity to join the Viper lab, for changing my perspective on remote sensing and for knowing that I should follow the path to become a remote sensing scientist, even when I did not know it myself. Frank, thanks for so many insightful conversations, for teaching me all of landscape ecology and conservation planning, and for the opportunity to work together at the La Kretz Research Center. I am also extremely grateful to my other committee members, Kelly Caylor, who provided lots of great discussion, invaluable feedback, encouragement, and support – also, thanks for always pushing me to see the big picture! And Laura Hess, who taught me everything I know of object-based remote sensing and also served as a mentor for me; I have no words to describe how grateful I am. Laura, thanks for so many eCognition lessons, encouragement, support, and for this beautiful collaboration and friendship we developed.

Thanks to the Coordenação de Aperfeiçoamento de Pessoal de Nível Superior (CAPES) Foundation and the Science without Borders program, Brazilian Ministry of Education, for providing the financial support that allowed me to develop most of my doctorate research; and to the Department of Geography at UCSB for providing me several Teaching Assistantships throughout these years. I am also very grateful for the opportunity I had to collaborate with research fellows at the GIS Laboratory from the Smithsonian Conservation Biology Institute; a special thanks to Peter Leimgruber for welcoming me at the SCBI, Ramiro Crego for substantial collaboration on the landscape connectivity models, and Lacey Hughey for introducing me to the group. I also would like to thank Radhika Dave for the consultancy opportunity at IUCN and for trusting me with the challenging task of coordinating efforts for the Bonn Challenge Barometer 2020 update in Brazil. I would like to send a big thank you to my research collaborators: Fernanda Brum, Nadinni Sousa, and Carlos Klink, who all provided immensurable support, feedback, and insights regarding conservation planning in the Cerrado. I also would like to thank Bruno Walter, Braulio Ferreira Dias, Ricardo Machado, Aline Leão, David Oren, Neiva Guedes, Miguel Marini, Tulio Dornas, Mauro Lambert Ribeiro, and Osmar Abilio, who all provided vital feedback and help on my research.

Thanks to all Viper lab fellows who were part of my everyday life in these past six years, in particular Susan Meerdink, Erin Wetherley, and Seth Peterson, who I am extremely grateful for so many great discussions, coffee breaks, and substantial encouragement; and David Miller, Chris Kibler, Rachel Green, and Michael Allen, for always providing support when I needed. Many thanks to my Santa Barbara friends who were all an essential part of this process: Blake, Sara, Sari, Brandi, Tammy, Lacey, Max, Delphine, Thomas, Alicia, Chema, Valentina, Javier, and Francine; and especially my Brazilian friends Natasha, Camilla, Henrique, Osvaldo, Alex, Luana, Yennie, Pedro, Thati, Minhoca, Walter, Waltinho, Mariana, Marcia, Rafael, Erica, and Tati Kuplich -- being an international student involves additional challenges and having this amazing community throughout these years was crucial! Another big thanks to my friends Janet Nackoney, Miriam Carvalho, Michael Goulding, Bruce Forsberg, John Melack, Sally MacIntyre for your great support and encouragement throughout these years!

Finally, and mostly importantly, I am eternally grateful to my best friend, lab-mate, life partner, and love of my life, Gabriel Daldegan, for boarding on this journey together. Thank you so much for taking great care of me when I most needed, for always supporting me and cheering me up throughout these years; I would never have completed this dissertation without you! I am also deeply thankful to my parents and my brother, Mauro, Andrea, and Roberto, for their endless encouragement, love, and inspiration, which were vital to accomplish this research. A big special thanks to my research heroes, Dad and Gabriel, for always providing immensurable support and feedback on my dissertation.

Curriculum Vitae

Fernanda de Figueiredo Ribeiro

Education

2014 – 2020	Ph.D. in Geography, University of California, Santa Barbara, CA, USA
2009 – 2013	BSc. in Geography, University of Brasília, Brasília, DF, Brazil

Professional and Research Experience

2020	Visiting Research Scientist, Smithsonian Conservation Biology Institute, Front Royal, VA
2020	Geospatial Data Scientist Consultant, International Union for Conservation of Nature and Natural Resources, Washington, DC
2015 – 2019	Teaching Assistant, Department of Geography, University of California, Santa Barbara, CA
2018	Graduate Research Fellow, La Kretz Research Center at the UC Sedgwick Reserve, Santa Barbara, CA
2013, 2014	GIS Consultant, The Nature Conservancy Brazil
2013	Research Assistant, Laboratory of Planning for Biodiversity Conservation – Labio, Department of Zoology, University of Brasília, Brazil
2009 – 2013	Undergraduate Research Assistant, Laboratory of Spatial Information Systems – LSIE, Department of Geography, University of Brasília, Brazil
2010 – 2012	Undergraduate Teaching Assistant, Department of Geography, University of Brasília, Brazil
2012	GIS Intern, World Wildlife Fund – WWF Brazil
2011	GIS Intern, Brazilian National Water Agency
2010	Undergraduate Research Assistant, Pantanal Research Center, Brazil

Publications

Ribeiro, F.F., Roberts, D.A., Hess, L.L., Davis, F.W., Caylor, K.K. and Daldegan, G.A., 2020. Geographic Object-Based Image Analysis Framework for Mapping Vegetation Physiognomic Types at Fine Scales in Neotropical Savannas. Remote Sensing, 12(11), p.1721.

Daldegan, G.A., Roberts, D.A., **Ribeiro, F.F.**, 2019. Spectral mixture analysis in Google Earth Engine to model and delineate fire scars over a large extent and a long time-series in a rainforest-savanna transition zone. Remote Sens. Environ. 232, 111340.

Daldegan, G.A., De Carvalho, O.A., Guimarães, R.F., Gomes, R.A.T., **Ribeiro, F.F**. and McManus, C., 2014. Spatial patterns of fire recurrence using remote sensing and GIS in the Brazilian savanna: Serra do Tombador Nature Reserve, Brazil. Remote Sensing, 6(10), pp.9873-9894.

Salmona, Y.B., **Ribeiro, F.F.** and Matricardi, E.A.T., 2014. Parques “no papel” conservam? O caso do parque dos Pireneus em Goiás. Boletim Goiano de Geografia, 34(2), pp.295-310.

Fellowships, Grants and Awards

2019 Student Presentation Award, Society for Conservation GIS Conference

2015 – 2019 Jack and Laura Dangermond Student Travel Award, Department of Geography, UC Santa Barbara

2014 – 2018 Graduate Student Fellowship through the Science without Borders program, Brazilian Coordination for the Improvement of Higher Education Personnel (CAPES)

2016 Student Travel and Research Summer Awards, Earth Research Institute

2013 Research Fellow, Brazilian National Council of Technological and Scientific Development (CNPq)

2012 Fieldwork Grant from the O Boticatio Group Foundation for Nature Protection to conduct my undergraduate research at the *Serra do Tombador* Private Reserve of Natural Heritage, GO, Brazil

2011 Undergraduate Research Fellowship, Inter-American Institute for Cooperation on Agriculture and the University of Brasília, Brazil

2010 Undergraduate Research Fellowship, Ministry of Agrarian Development and the University of Brasília, Brazil

2010 Undergraduate Research Fellowship, Brazilian National Council of Technological and Scientific Development (CNPq)

Selected Conference Presentations

2019 **Ribeiro, F.F.,** Davis, F., Roberts, D.A., Caylor, K.K., Hess, L. L., Daldegan, G.A., Brum, F. T. *Is the Forest Code legislation protecting Neotropical savanna habitats in agricultural landscapes? A high-resolution Gap analysis of the Brazilian Cerrado*. American Geophysical Union Fall Meeting. San Francisco, CA. (Poster)

2019 **Ribeiro, F.F.,** Davis, F., Roberts, D.A., Daldegan, G.A. *Setting conservation priorities at fine spatial scales in a complex Neotropical savanna landscape*.
Society for Conservation GIS – SCGIS 21st Annual Conference. Monterey, CA. (Presentation)

2018 **Ribeiro, F.F.**, Hess, L., Roberts, D.A., Davis, F., Daldegan, G.A. *Challenges and contributions of using high spatial resolution data for biodiversity conservation planning in tropical savannas*. American Geophysical Union Fall Meeting. Washington, D.C. (Poster)

2017 **Ribeiro, F.F.**, Davis, F., Roberts, D.A., Caylor, K., Nackoney, J., Daldegan, G.A. *Vegetation mapping and habitat suitability analysis of Anodorhynchus hyacinthinus in the Brazilian savanna*. Ecological Society of America Annual Meeting. Portland, OR. (Poster)

2017 **Ribeiro, F.F.**, Davis, F., Roberts, D.A., Caylor, K., Nackoney, J., Daldegan, G.A. High spatial resolution mapping of land cover types in Cerrado heterogeneous landscapes. Society for Conservation GIS – SCGIS 20th Annual Conference. Monterey, CA. (Presentation)

Abstract

Fine-Scale Analyses for Improving Conservation and Sustainability Efforts in Agricultural Landscapes of Neotropical Savannas

by

Fernanda de Figueiredo Ribeiro

Regional maps featuring the fine-scale heterogeneity of neotropical savannas are necessary for delineating species habitats and for supporting conservation and ecological analyses. The Brazilian neotropical savanna is the most floristically diverse savanna in the world and is amongst the top 36 global priorities for conservation. In this dissertation, I used a suite of fine-scale geospatial analyses and remote sensing imagery in support of improving biodiversity conservation efforts in Cerrado private lands. In Chapter 2, I developed a systematic framework using Geographic Object-Based Image Analysis (GEOBIA) that incorporated spectral and spatial features in a novel environmental spatial ruleset developed to map a wide range of Cerrado vegetation structural types at 5-m resolution. This framework mapped 13 land cover categories effectively, of which 11 were physiognomic types. Map accuracy was 87.6%. The results show that high spatial resolution imagery is appropriate for discriminating Cerrado land cover classes and the GEOBIA framework is essential for refining land cover categories to ecological classes (physiognomic types). To the best of our knowledge, this is the first map to feature a wide range of detailed physiognomic types with high map accuracy at high spatial resolution. In Chapter 3, I

developed a fine-scale spatially explicit gap analysis to estimate possible protection status of physiognomic types and important habitats for endangered and endemic avifauna from areas under land use regulation (i.e. Brazil's Forest Code). I also assessed the potential compliance status of rural properties with land use regulation by land property size. Moreover, I proposed a quantitative approach to support current policy implementation and improve their guidelines. The results indicate that instruments of policy implementation such as Legal Reserves and Areas of Permanent Preservation are essential for ensuring protection of a wide diversity of physiognomic types and essential habitats for endemic and endangered species. I demonstrated that allocation for mandatory set asides can be optimized to maximize biodiversity by considering the representativeness of unprotected physiognomic types. Moreover, the results suggest that land property size might be a reliable indicator to target illegal land clearing within areas under current policy. Thus, efforts to enforce and monitor policy compliance can be improved by targeting land property size. In Chapter 4, I investigated alternative conservation priority-setting schemes to allocate privately protected areas in a Cerrado commodity-driven agricultural landscape, aiming to improve habitat protection and increase landscape connectivity between protected areas. The results suggest that unprotected vegetation in Cerrado private lands is critical to maintaining regional structural connectivity between large protected areas such as national parks and ecological stations. I found that additional conservation set-asides are important for complementing habitat representation and increasing habitat protection and landscape connectivity beyond efforts implemented by current policies. Thus, conservation in private lands represents an opportunity to reconcile conservation and agricultural production.

Table of Contents

Chapter 1. Introduction

1.1 Motivation

Human activities have already compromised millions of species and ecosystems, leading to global declines in biodiversity (Butchart et al., 2010; Cardinale et al., 2012). To feed the increasing human population and meet global demands for food, fiber, and energy consumption, crop and livestock production has continued to expand (Kremen and Merenlender, 2018; Tilman et al., 2009). Land conversion and degradation, however, imposes one of the main threats to terrestrial biodiversity through habitat loss and fragmentation (Joppa et al., 2016). For instance, land conversion alone is responsible for ~80% of all threatened terrestrial birds and mammals, mostly driven by agricultural expansion in the tropics (Tilman et al., 2017). To mitigate such threats, protected areas are created to promote the long-term protection of natural landscapes for species persistence and provision of ecosystem services (Margules and Pressey, 2000).

Conservation planning to allocate new protected areas generally favors large, well-connected habitat patches to avoid fragmentation effects on species and ecosystems (e.g., Kennedy et al., 2016a). Although essential, large blocks of natural vegetation are increasingly scarce in most tropical regions due to agricultural expansion (Hansen and DeFries, 2007). Despite this, agricultural landscapes in the tropics still retain high biodiversity levels and provide essential ecosystem services at the landscape level (Galetti et al., 2009; Magioli et al., 2016). The remaining habitats in tropical agricultural landscapes act as wildlife refuges for several endemic and endangered species (Chiarello, 2000; Magioli et al., 2016) and as stepping-stones connecting large protected areas (Banks-Leite et al., 2014; Rocha et al., 2014). Thus, agricultural landscapes are essential to tropical conservation strategies, and a complementary component to large protected areas (Klink, 2019; Magioli et al., 2016; Melo et al., 2013).

With much of the world's biodiversity located in private lands, these areas should be effectively managed to provide essential ecosystem services and important resources for species, and to maintain connectivity between protected areas (Kamal and Grodzinska-Jurczak, 2014). Therefore, conservation strategies in private lands represent a global opportunity to complement current protected area networks, and mitigate climate change and habitat loss (Bingham et al., 2017; Borrini-Feyerabend et al., 2013; Cortés Capano et al., 2019; Kamal et al., 2015; Knight, 1999; Stolton et al., 2014). Planning for conservation in private agricultural landscapes is particularly challenging due to private property rights, high costs of land acquisition, and lack of financial incentives to landowners (Naidoo et al., 2006; Schuster et al., 2018). A strong articulation between new policies supporting protection and monitoring of natural ecosystems, private governance, and engagement of stakeholders in the agricultural sector is necessary for improving conservation efforts in private agricultural landscapes in the tropics (Assunção and Chiavari, 2015; Klink, 2019; Strassburg et al., 2017).

While most spatial planning approaches to allocate protected areas use coarse to medium spatial resolution datasets to identify broad-scale conservation priorities, these datasets do not feature small and heterogeneous patches remaining in human-dominated landscapes (Paese et al., 2010). If not explicitly accounted for in the spatial planning process, small habitat patches within agricultural landscapes may be converted to other land uses without regulatory restrictions (Wintle et al., 2019). To help overcome this challenge, the growing availability of fine-scale Earth observation data offers a clear opportunity to improve conservation priority-setting in tropical heterogeneous and fragmented regions (Nagendra et al., 2013), such as the Brazilian Cerrado (Ribeiro et al., 2020). This dissertation takes a step towards filling this gap by using remote sensing and a suite of fine-scale geospatial analyses to improve biodiversity conservation in private agricultural landscapes of the Cerrado.

1.2 Background

The Cerrado is a neotropical savanna composed of a high diversity of physiognomic types embedded in a vegetation mosaic of forest, savannas, and grasslands occupying the central plateau of Brazil (Oliveira-Filho and Ratter, 2002; Ribeiro and Walter, 2008). It has a tropical climate characterized by a wet season from October to April, and dry season from May to September, when rainfall can be close to zero (Grimm, 2011). Plant distribution across this biome is mostly determined by topography, soil texture, depth, and nutrient content fire regime, and water availability (Cole, 1986; Oliveira-Filho and Ratter, 2002). Spatial variation in these environmental conditions results in high beta diversity across the biome as well as highly variable physiognomic types over relatively small distances (Bridgewater et al., 2004; Ribeiro and Walter, 2008).

Brazil is a megadiverse country and a global leader in commodity exports, where fast land conversion imperils high levels of biodiversity found within its tropical forests, savannas and grasslands (Matricardi et al., 2019; Strassburg et al., 2017). Most of Brazil's commodity-driven agriculture is concentrated in the Cerrado, the world's richest savanna in terms of endemicity and diversity of vascular plants, harboring 1.5% of the global flora as endemics (Klink and Machado, 2005; Mendonça et al., 2008; Ratter et al., 1997). The Cerrado has more than 12,000 plant species (~44% endemics: Mendonça et al., 2008; Ratter et al., 1997), 840 birds (Jose\' Maria Cardoso Da Silva, 1997), 227 mammals (Carmignotto et al., 2012), 267 squamates (Nogueira et al., 2011), and 209 frogs (Valdujo et al., 2012). However, its biodiversity remains poorly understood. This biome also provides essential ecosystem services for agriculture and food security, including water quality supply, carbon storage, and livestock forage (Overbeck et al., 2015).

Although its contribution to biodiversity and ecosystem services is evident, the Cerrado has already lost ~50% of its native vegetation cover and no more than 20% remains undisturbed (Strassburg et al. 2017). Although only ~8% of its native vegetation is under current protection, protected areas are key to reducing land conversion in the Cerrado (Brum et al., 2019; Carranza et al., 2014; Françoso et al., 2015a). Since much of the remaining natural vegetation is within private lands, the Cerrado offers new opportunities to improve conservation efforts within its private agricultural areas (Klink, 2019; Strassburg et al., 2017). The implementation of conservation strategies in such settings can be either mandatory, through the Forest Code legislation, or voluntary, through the implementation of private natural reserves (e.g. Private Reserves of Natural Heritage, or RPPNs – *Reserva Particular do Patrimônio Natural*). The Forest Code is a land-use regulation policy that requires all Cerrado landowners to set aside 20% of their lands into Legal Reserves. In addition, it also restricts land use within Areas of Permanent Preservation, which are designated to conserve water resources and prevent soil erosion in environmentally important areas, such as swamps, riparian areas, hilltops, and escarpments (Soares-Filho et al., 2014). RPPNs are privately protected areas of sustainable use defined by Brazil's

National Protected Areas System (*Sistema Nacional de Unidades de Conservação da Natureza* – SNUC: MMA, 2003) and are voluntarily established to protect biodiversity, allowing for a few limited activities such as scientific research, education, recreation, and tourism (Rambaldi et al., 2005).

Most of the remaining Cerrado vegetation is on lands suitable for commodity crops, especially soybeans (88.4%) and sugarcane (68.7%), which are projected to increase production to meet future global demands (Strassburg et al., 2014). Brazil is the world's top producer and exporter of soybeans, which come mostly from the Cerrado's new agricultural frontier – the Matopiba region (https://trase.earth/ last accessed in July 16, 2020), occupying ~73Mha over the states of **Ma**ranhão, **To**cantins, **Pi**aui, and **Ba**hia. The western portion of Bahia State is one of the most modified regions in Matopiba, where over 1Mha were cleared between 2002 and 2010 (Salmona et al., 2016). Particularly, the municipalities of Barreiras, Luís Eduardo Magalhães, São Desidério, and Riachão das Neves are amongst the most productive areas in the Cerrado for soy crops, with most production going to China, Brazil's internal market, and Europe (https://trase.earth, last accessed July 16, 2020). Despite its agribusiness focus, this region supports many species and ecosystems that are endangered due to habitat loss and climate change, such as the Hyacinth Macaw (*Anodorhynchus hyacinthinus*), the Brazilian Merganser (*Mergus octosetaceus*) and the Wagler's Woodcreeper (*Lepidocolaptes wagleri*), with the latter also being an endemic species of restricted distribution (IUCN, 2020). The remaining natural vegetation within Cerrado agricultural landscapes, in particular, has high biodiversity levels and vital ecological functions, and therefore, should be incorporated in future conservation strategies (Magioli et al., 2016).

Maps featuring the fine-scale heterogeneity of Cerrado physiognomic types can be used as biodiversity surrogates in Cerrado conservation planning approaches, especially where *in-situ* observations are unavailable (Monteiro et al., 2020). However, land-cover mapping of tropical savannas still faces major uncertainties related to their definition and classification (Bond and Parr, 2010; Eiten, 1972a; Huntley and Walker, 1982; Parr et al., 2014). These uncertainties relate to growth patterns (associated with seasonality of contrasting dry and wet seasons) and to admixtures of life forms and land cover categories at operational sensor scales (Cord et al., 2010; Herold et al., 2008). Most current imagery has a moderate to coarse spatial resolution (≥ 30 m) failing to resolve the fine-scale heterogeneity of savannas (Ferreira et al., 2003, 2007). By contrast, imagery at fine (< 10m) spatial resolution can better capture the diversity in vegetation structure of heterogeneous and complex savanna landscapes (Arroyo et al., 2010a; Gibbes et al., 2010; Girolamo-Neto et al., 2018; Nagendra et al., 2013; Nagendra and Rocchini, 2008).

1.3 Research Objectives and Overarching Questions

This research aims to improve our understanding of conservation efforts in private agricultural landscapes in the Cerrado through fine-scale geospatial analysis.

The overarching questions that motivated this study are:

Question 1 – Can the wide range of Cerrado heterogeneous physiognomic types be accurately mapped at fine spatial scales using Geographic Object-Based Image Analysis?

Question 2 – To what extent is the biodiversity of a Cerrado agricultural landscape protected by Brazil's Forest Code? How much biodiversity is left unprotected by current policy? How much biodiversity can be incorporated if areas under the Forest Code policy are allocated according to representation of natural habitats? Can policy compliance be monitored by targeting land property size?

Question 3 – How do different biodiversity target levels, along with connectivity, affect land requirements for efficient conservation in Cerrado private lands? How much of unprotected land is required to adequately represent habitats in Cerrado privately protected areas? How does a greater emphasis on connectivity affect conservation land requirements?

To thoroughly answer these questions, I generated a map of Cerrado physiognomic types. Given the challenges of mapping Cerrado physiognomic types with traditional pixel-based methods at medium to coarse spatial resolution, in Chapter 2, I developed a systematic GEOBIA framework using single-date high spatial resolution imagery and a novel environmental spatial ruleset developed to identify a wide range of Cerrado vegetation structural types. This framework proved to be a robust method to differentiate a larger number of physiognomic types at a higher accuracy than previously reported in several studies regarding Cerrado land cover mapping. In Chapter 3, I developed a fine-scale spatially explicit gap analysis to estimate possible protection status of physiognomic types and important habitats for endangered and endemic avifauna from areas under the Forest Code policy. I also assessed the potential compliance status of rural properties with the Forest Code by land property size. Moreover, I proposed a quantitative approach to support the implementation of Forest Code and improve policy guidelines. In Chapter 4, I investigated alternative conservation priority-setting schemes to allocate privately protected areas in a Cerrado commodity-driven agricultural landscape, aiming to improve habitat protection and increase landscape connectivity between protected areas.

Chapter 2. Geographic Object-Based Image Analysis framework for mapping vegetation physiognomic types at fine scales in Neotropical savannas

Fernanda F. Ribeiro[1], Dar A. Roberts[1], Laura L. Hess[2], Frank W. Davis[3], Kelly K. Caylor[1,3], Gabriel Antunes Daldegan[1]

[1] Department of Geography, University of California, Santa Barbara, CA 93106, USA
[2] Earth Research Institute, University of California, Santa Barbara, CA 93106, USA
[3] Bren School of Environmental Science and Management, University of California, Santa Barbara, CA 93106, USA

This chapter is based on a manuscript published in the journal *Remote Sensing*.

Ribeiro, F.F., Roberts, D.A., Hess, L.L., Davis, F.W., Caylor, K.K. and Daldegan, G.A., 2020. Geographic Object-Based Image Analysis Framework for Mapping Vegetation Physiognomic Types at Fine Scales in Neotropical Savannas. Remote Sensing, 12(11), p.1721.

2.0 Abstract

Regional maps of vegetation structure are necessary for delineating species habitats and for supporting conservation and ecological analyses. A systematic approach that can discriminate a wide range of meaningful and detailed vegetation classes is still lacking for neotropical savannas. Detailed vegetation mapping of savannas is challenged by seasonal vegetation dynamics and substantial heterogeneity in vegetation structure and composition, but fine spatial resolution imagery (<10 m) can improve map accuracy in these heterogeneous landscapes. Traditional pixel-based classification methods have proven problematic for fine spatial resolution data due to increased within-class spectral variability. Geographic Object-Based Image Analysis (GEOBIA) is a robust alternative method to overcome these issues. We developed a systematic GEOBIA framework accounting for both spectral and spatial features to map Cerrado structural types at 5-m resolution. This two-step framework begins with image segmentation and a Random Forest land cover classification based on spectral information, followed by spatial contextual and topological rules developed in a systematic manner in a GEOBIA knowledge-based approach. Spatial rules were defined *a priori* based on descriptions of environmental characteristics of 11 different physiognomic types and their relationships to edaphic conditions represented by stream networks (hydrography), topography, and substrate. The Random Forest land cover classification resulted in 10 land cover classes with 84.4% overall map accuracy and was able to map 7 of the 11 vegetation classes. The second step resulted in mapping 13 classes with 87.6% overall accuracy, of which all 11 vegetation classes were identified. Our results demonstrate that 5-meter spatial resolution imagery is adequate for mapping land cover types of savanna structural elements. The GEOBIA framework, however, is essential for refining land cover categories to ecological classes (physiognomic types), leading to a higher number of vegetation classes while improving overall accuracy.

Keywords: GEOBIA; land cover mapping; high spatial resolution imagery; savanna; Cerrado biome; vegetation types

2.1 Introduction

Monitoring patterns and trends in tropical savannas still faces major uncertainties related to their definition and classification (Bond and Parr, 2010; Eiten, 1972b; Huntley and Walker, 1982; Parr et al., 2014). This uncertainty is reflected both in general land cover classification and maps featuring vegetation physiognomic types (e.g., life form, vegetation cover). For example, savannas are poorly defined in global land cover products, and variation in their physiognomic types is not well classified at local scales (Bastin et al., 2017; Eiten, 1972b; Symeonakis et al., 2018). Most current technology featuring moderate to coarse spatial resolution (> 10 m) fails to resolve the fine-scale heterogeneity of savannas. Major issues in their discrimination relate to growth patterns (associated with seasonality of contrasting dry and wet seasons) and to admixtures of life forms and land cover categories at operational sensor scales (Cord et al., 2010; Herold et al., 2008).

Savannas occupy a significant area of the tropics, covering approximately 20% of the world's land surface (Huntley and Walker, 1982; Scholes and Archer, 1997). Tropical savannas, for example the Argentinian Chaco, the African Miombo, and the Brazilian Cerrado, are often intermixed with riparian forests, swamps, and marshes (Scholes and Archer, 1997). They are composed of a herbaceous stratum in a discontinuous tree and shrub cover of varying height and density (Eiten, 1982, 1972b; Huntley and Walker, 1982). The Cerrado, a neotropical savanna in Brazil, is the most floristically diverse savanna in the world, with more than 12,000 plant species (Mendonça et al., 2008), including numerous endemics (Felfili and Felfili, 2001; Silva et al., 2006). Moreover, the Cerrado provides critical ecosystem services such as carbon storage (Grace et al., 2006) and plays a major role in provision of water resources by hosting the headwaters of the three largest watersheds in South America.

Land cover mapping of savannas has been conducted mostly at regional scales, using optical sensors available at moderate (10–500 m) to coarse (>500 m) spatial resolution, such as the Landsat series and Moderate Resolution Imaging Spectrometer (MODIS). Most studies focus on multi-temporal analysis for change detection (Coulter et al., 2016; Mayes et al., 2015; Müller et al., 2015; Oliveira et al., 2014), deforestation monitoring (Beuchle et al., 2015; de Oliveira et al., 2017a; Johansen et al., 2015; Trancoso et al., 2014), and land surface phenology (Ferreira et al., 2003; Jin et al., 2013; Schwieder et al., 2016). Specific challenges to savanna land cover classification are related to: (a) high sensitivity to sensor resolution due to discontinuous tree canopy cover (Whiteside et al., 2011); (b) high seasonal variation in ecosystem properties, cloud cover, and data availability (Sano et al., 2007); and (c) smoke and haze due to frequent fires in the dry season (Cochrane, 2003).

As for other savannas, discriminating spectrally similar shrubs from trees with moderate-to-coarse resolution imagery has proven challenging for the Brazilian Cerrado (Ferreira et al., 2003, 2007). Sano et al. (2010) used image segmentation and visual interpretation of Landsat to produce a map of natural and converted areas for the entire Cerrado region. Other Landsat-based studies have focused on local sites to investigate methods for mapping fractional woody cover, such as spectral unmixing (Ferreira et al., 2007), and Support Vector Machine classification of multi-year phenologic profiles based on the Tasseled Cap Transform (Schwieder et al., 2016). Several studies took advantage of multi-temporal rather than single-date imagery to overcome spectral similarities in woody cover using characteristic phenological patterns (Ferreira et al., 2003; Hill et al., 2017;

Müller et al., 2015; Ratana et al., 2005; Schwieder et al., 2016). Although these approaches are useful for broad-scale analyses, they depict coarse structural vegetation classes (Ferreira and Huete, 2004; Franklin, 1991; Franklin et al., 1991; Schwieder et al., 2016) and cannot resolve the structural heterogeneity essential for regional biodiversity and ecosystem assessments (Nagendra and Rocchini, 2008).

A critical problem in mapping Cerrado physiognomic types concerns the definition of classes. Most previous remote sensing studies considered a widely adopted vegetation nomenclature for the Cerrado physiognomies based on structural attributes and floristic composition (see Ribeiro and Walter, 2008). Vegetation maps exhibiting the diverse structural variation in vegetation types are critical for representing fine-scale savanna habitat patterns. Spectrally based remote sensing analyses based on floristic classification systems (such as Ribeiro and Walter , 2008) may not succeed in identifying structural differences in vegetation and may require extensive field work for species identification. Thus, they may not be suitable for regional scale mapping using multispectral imagery classification alone. Geographical characteristics related to edaphic conditions (e.g., topography, soils), however, can potentially help identify some physiognomic types not strictly based on species composition.

Remote sensing imagery at fine (< 10m) spatial resolution can better capture the diversity in vegetation structure of heterogeneous and complex savanna landscapes (Arroyo et al., 2010b; Gibbes et al., 2010; Girolamo-Neto et al., 2018; Nagendra et al., 2013; Nagendra and Rocchini, 2008). However, traditional pixel-based classification methods have proven problematic for fine spatial resolution data due to increased within-class spectral variability, potentially leading to inconsistent results (Blaschke et al., 2014; Hay et al., 1996). Geographic Object-Based Image Analysis (GEOBIA) bridges remote sensing and Geographic Information Science by defining image objects as entities and focusing on the conceptual modeling of defined land cover classes at multi-scales. Thus, GEOBIA is a robust alternative approach to address within-class spectral variability issues in land cover classification of high spatial resolution imagery and heterogeneous landscapes (Blaschke et al. 2014).

GEOBIA is based on extracting information from Earth Observations using spectral, spatial, structural, and hierarchical properties of an image (Lang, 2008). A fundamental step is to delineate objects of interest, which are strongly associated with image segmentation approaches that cluster relatively homogenous pixels into image objects. One of the significant advantages of the GEOBIA approach is that image objects provide not only diverse spectral information (e.g., mean values per band, standard deviation, mean ratios) but also additional spatial information, such as distance, neighborhood, and topological metrics (Blaschke, 2010; Blaschke et al., 2014). The combination of spectral and spatial properties allows incorporation of contextual information of a given object using ontologies/semantics to create hierarchical conditional rules tailored to classify meaningful object definitions in a knowledge-based classification (Blaschke et al., 2014; Hay and Castilla, 2008).

Efforts to map fine-scale structural variation in savanna ecosystems using GEOBIA have obtained encouraging results compared to moderate spatial resolution data and pixel-based methods (Boggs, 2010; Gibbes et al., 2010; Girolamo-Neto et al., 2018; Kaszta et al., 2016; Whiteside et al., 2011). GEOBIA is also increasingly used for Cerrado studies due to the recent availability of fine (5 m) spatial resolution imagery from the RapidEye sensor at no cost for Brazilian researchers. Initial efforts to evaluate the utility of high spatial resolution imagery for discriminating and mapping Cerrado physiognomic types have demonstrated improved discrimination of structural classes and higher map accuracy compared to coarser resolution imagery (Girolamo-Neto et al., 2017; Orozco Filho, 2017; Teixeira et al., 2015). Such efforts include using supervised object-based classification with several input object features in a GEOBIA context, such as in Girolamo-Neto (2017) and Girolamo-Neto et al. (2018); or strict knowledge-based classification by defining conditional rules based on shape and brightness parameters, such as in Teixeira et al. (2015). However, these studies were tested at sites with limited extent (< 50,000 ha) such as Brasilia National Park, which does not include some major vegetation structural types known for causing misclassification errors (i.e., semi-deciduous versus deciduous forest: Teixeira et al., 2015), or featured coarse vegetation classes as opposed to detailed physiognomic types. A systematic approach that can discriminate a wide range of meaningful and detailed vegetation classes is still lacking for the Cerrado biome.

The primary goal of this study is to develop a systematic framework to discriminate detailed Cerrado physiognomic types in a semi-automatic manner using single-date high spatial resolution imagery. The rationale for mapping detailed physiognomic types at fine scales stems from the potential of such maps to (1) improve our understanding of species habitat requirements and conditions, as well as our ability to assess ecosystem services and biodiversity (Nagendra and Rocchini, 2008), and (2) provide improved inputs for fire modeling, carbon accounting (Gomes et al., 2020), landscape restoration (Dave et al., 2019), and land-use management (Brannstrom et al., 2008). Our approach takes advantage of GEOBIA and semantics to combine land cover classes and edaphic conditional drivers in the definition of hierarchical contextual rules used to classify a wide range of Cerrado physiognomic types. The specific research questions we aim to answer with this framework are: What accuracies are achievable using spectral information alone? What accuracies are achievable adding spatial context information? How can the widely adopted Cerrado physiognomic types nomenclature be used in a remote sensing analysis? We address these questions using RapidEye imagery (5 m) in a two-step GEOBIA framework that begins with a supervised object-based land cover classification based on spectral information alone, followed by assignment of spectral land cover classes to more detailed physiognomic types using a novel hierarchical spatial and topological ruleset defined by semantics (i.e., descriptive assessment and knowledge). This approach takes advantage of ancillary information on hydrography, topography, and substrate as environmental conditional drivers in the semantic definition of hierarchical contextual rules. This GEOBIA framework was tested for two large study sites covering most major Cerrado physiognomic types. Its main advantages relate to its reproducibility across different areas of this heterogeneous biome, its capacity to discriminate a wide variety of physiognomic types that could not be distinguished in previous studies, and its adaptability to other physiognomic types and to other types of optical imagery.

2.2 Materials and Methods

2.2.1 Study Sites

The Cerrado has a tropical climate characterized by an October–April wet season and May–September dry season, when rainfall can be close to zero (Grimm, 2011). Plant distributions across the Cerrado are mostly determined by topography, soil texture, nutrient content and depth, fire regime, and water availability (Cole, 1986; Oliveira-Filho and Ratter, 2002). Spatial variation in these environmental conditions results in high beta diversity across the biome as well as large variability of physiognomic types over relatively small distances (Bridgewater et al., 2004; Ribeiro and Walter, 2008).

We chose two study sites (**Figure 2.1**) to test our classification framework and compare its accuracy in discriminating savanna vegetation with differing landscape composition and surface heterogeneity. The sites were chosen based on their ecological importance for conservation, their differences in composition and beta diversity, and a combination of imagery and ancillary data availability.

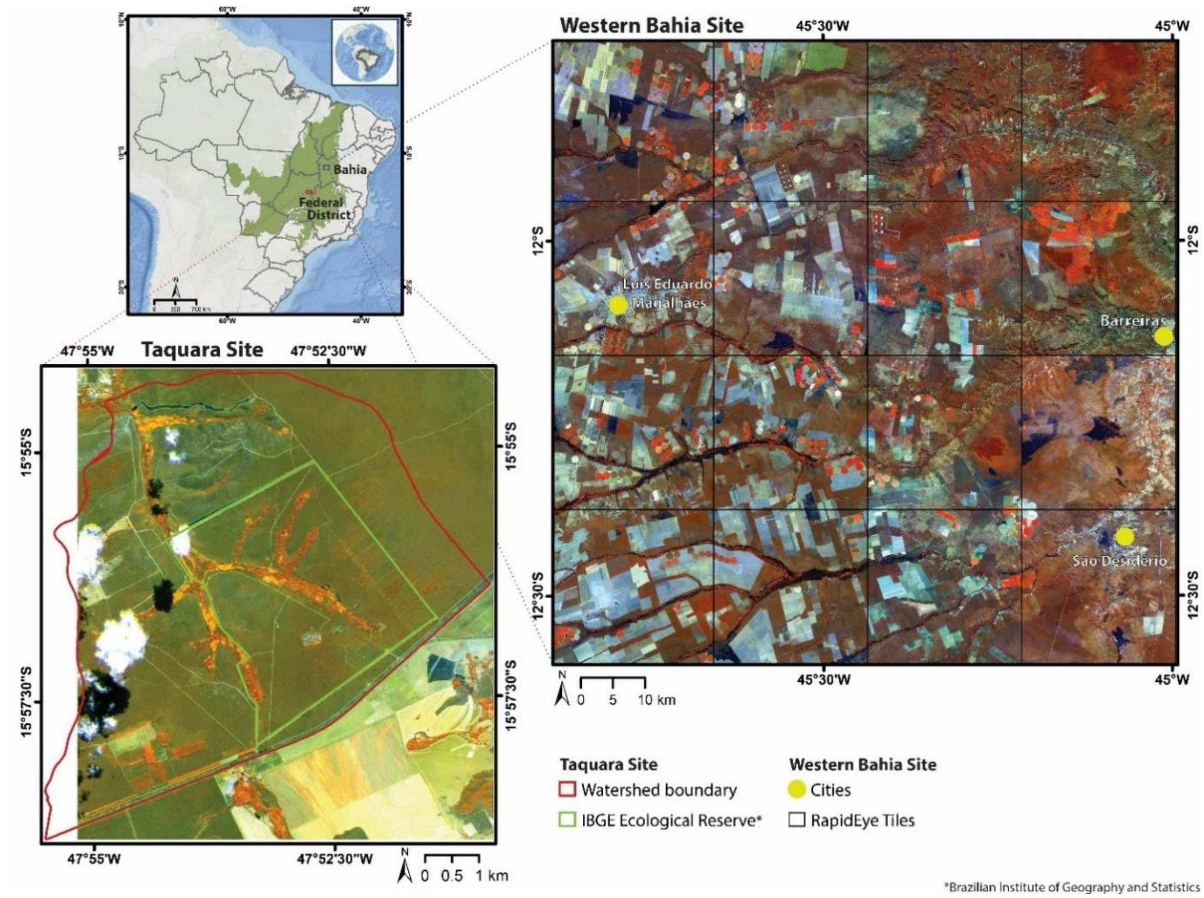

Figure 2.1. Map (upper left corner) showing the Cerrado biome and the location of both study sites and their respective states, in bold. The other two maps show the RapidEye imagery used in the Taquara (encompassing the Brazilian Institute of Geography and Statistics – IBGE Ecological Reserve and the Taquara watershed) and the Western Bahia sites, with false-color composition as R: NIR (band 5), G: Red-edge (band 4), B: Blue (band 1). The Western Bahia site map shows a grid representing the acquired 16 imagery tiles.

2.2.1.1. Study Site 1: Taquara watershed

We initially tested our method at the Taquara site (**Figure 2.1**), a study site for which we had high-quality orthophotos (24 cm resolution) and a greater availability of ground reference and ancillary data that were important for testing our ability to visually identify physiognomic types using air photos when collecting training data. The Taquara site (15°54'S, 47°55'W to 15°58', 47°50'W) comprises an area of 67.3 km^2 located approximately 26 km from downtown Brasília, covering most of the Taquara watershed and its surroundings. This site also contains the Brazilian Institute of Geography and Statistics (IBGE) Ecological Reserve, a protected area created to act as a biodiversity control site for comparison to other Cerrado areas altered by human occupation. The IBGE Ecological Reserve served as one of the Cerrado sites included in the Large Scale Biosphere-Atmosphere Experiment in Amazonia (LBA) (Roberts et al., 2003), and was the first International Long Term Ecological Research (ILTER) site in the Cerrado biome .

The watershed is located in the Cenozoic bed from the Paranoa Group and terrain is relatively flat, with elevation in the region varying between 1040 and 1196 meters. Mean annual precipitation is 1426 mm and mean annual temperature is 23ºC (Silva and Bergamini, In prep.). The site has considerable diversity of plants and soil types, representing most of the typical physiognomic types found across the Cerrado. Soils are mostly acidic, low fertility Oxisols (Latosols) supporting savanna ecosystems. Organic, nutrient-rich hydromorphic soils occur locally in the area and often support forest ecosystems. Most of the site is covered by savanna ecosystems, but grasslands located on small hills, and gallery forests with surrounding wetlands following small streams are also present. Common tree species include *Pterodon pubescens*, *Bowdichia virgilioides*, *Vochysia thyrsoidea*, and *Dalbergia miscolobium*, while grasses are dominated by perennial species such as *Echinolaena inflexa*, *Schizachyrium tenerum*, *Trachypogon spicatus*, and *Axonopus chrysoblepharis* (Pereira and Furtado, 2011).

2.2.1.2. Study Site 2: Western Bahia

We also tested whether our method could be applied to a larger and even more heterogeneous site with minimal ground reference and ancillary data. This study site comprises an area of 9409 km^2 and is located on the western side (11°43'S, 45°52'W to 12°36'S, 44°59'W) of the São Francisco River watershed, the largest river basin entirely located in Brazilian territory (**Figure 2.1**). The Western Bahia site is not only naturally heterogeneous but also has considerable complexity due to historical land use conversion of natural savanna to pasture and row crop agriculture (Oliveira et al., 2014). Elevation ranges from a maximum of 808 m across karstic mesas/plateaus (known as *Chapadões do São Francisco*) to 433 m in the lowest point in the São Francisco Depression, with annual precipitation ranging from 800 mm at lower elevations to 1600 mm at highest elevations (Brannstrom et al., 2008; Nou and Costa, 1994). The plateaus are composed of Proterozoic rocks from the Bambui Group and Cretaceous beds from the Urucuia Group. Diverse soils include deep well-drained Oxisols (Latosols) of medium texture in the highest parts of the plateau and sandy texture (sandy quartz) on irregular terrain, rocky soils (Lithosols) of sandy to medium texture on steep slopes and escarpments, and hydromorphic/organic soils across floodplains (Nou and Costa, 1994). This variety of edaphic conditions supports diverse vegetation types mostly consisting of savanna ecosystems across the plateaus, wetlands and riparian vegetation along floodplains, and semi-deciduous forest restricted to cliffs and to the eastern part of the plateau, which is possibly due to local concentrations of calcium carbonate in the soil and higher moisture conditions (Nou and Costa, 1994). In general, lower elevation sites have greater physiognomic diversity compared to the plateaus (Silva and Bates, 2002).

Common tree species within the savanna ecosystem include *Anacardium occidentale* and *Miconia ferruginea*, and the grass layer is dominated by annual species such as *Ichnantus hoffmannseggii* (Santana et al., 2010). Seasonally dry tropical forests are mostly found along escarpment slopes, and are composed of deciduous or semideciduous tree species such as *Astronium urundeuva*, *Piptadenia macrocarpa*, *Chorisia speciosa.*, *Tabebuia spp.*, *Cavanillesia arborea*, and *Cedrella fissilis* (Furley and Ratter, 1988).

2.2.2 Methods Overview

The GEOBIA framework used in this study is divided into two major steps (Levels 1 and 2), in addition to pre and post-processing stages: (a) pre-processing; (b) land cover classification (Level 1 processing); (c) physiognomic types classification (Level 2 processing); (d) area estimates; and accuracy assessment (statistics). These procedures are shown and described in detail in **Figure 2.2**.

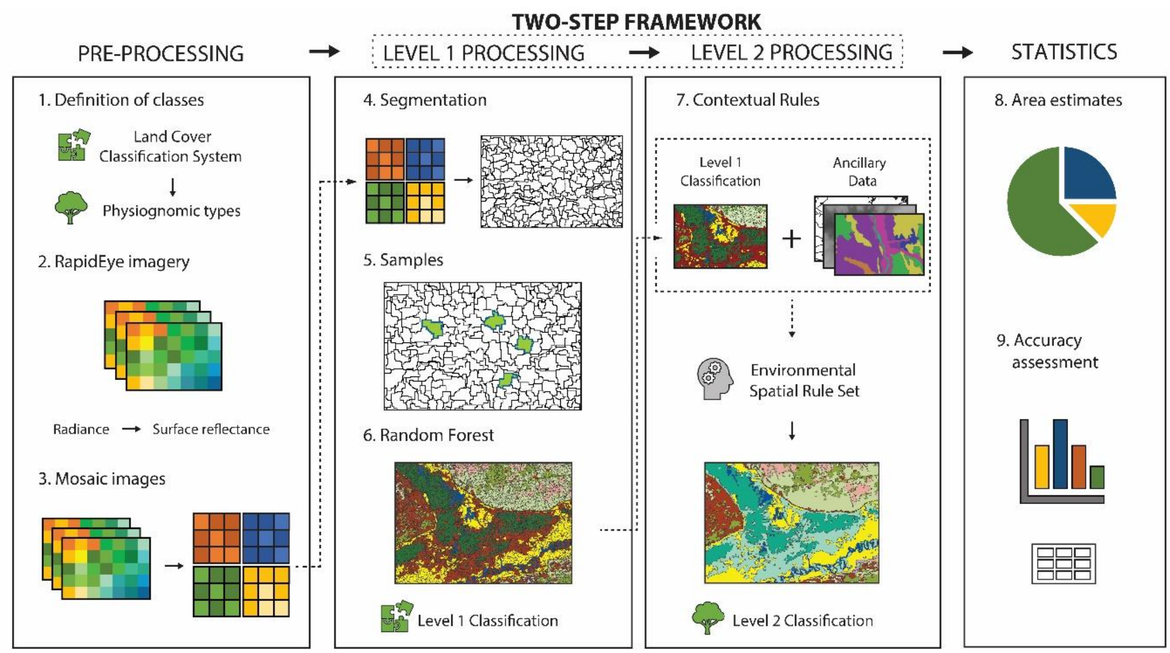

Figure 2.2. Workflow of the Geographic Object-Based Image Analysis (GEOBIA) classification framework to map Cerrado land cover and physiognomic types.

2.2.2.1. Definition of Classes

The Cerrado physiognomic types have been defined by many authors such as Coutinho (1978), Eiten (1978), and Oliveira-Filho and Ratter (2002). The most recent vegetation terminology proposed by Ribeiro and Walter (2008) has been widely adopted by the scientific community in Brazil. This scheme, however, is based on criteria such as environmental edaphic conditions and species composition that are not reliably detected by multispectral sensors. Thus, translating these on-the-ground Cerrado classification schemes to a land cover classification derived from remote sensing is a challenging task.

The United Nations Food and Agriculture Organization's (FAO) Land Cover Classification System (LCCS) is a flexible and systematic framework designed for land cover classification terminology at any given scale and for any data source (Di Gregorio and Jansen, 2000; Nou and Costa, 1994). The LCCS framework defines classes at different levels, starting with broad distinctions (e.g., Primarily Vegetated Areas, Primarily Non-Vegetated Area) within a dichotomous key and then adds specific attributes through a hierarchical framework (e.g., life form, cover, height). Classes are then defined as a function of the intended level of detail (scale) for the land cover classification based on a combination of the spatial and spectral resolution of the imagery, which makes the LCCS appropriate for object-based classification (Radoux et al., 2017).

To allow for standardization among Cerrado classes, we used the LCCS as a reference in the first classification level. The classes were defined *a priori*, based on a literature review of major physiognomic types found across the Cerrado. In accordance with the RapidEye spatial and spectral characteristics, the quality of the images (e.g., off-nadir viewing angles), and the recommended scale for mapping Cerrado physiognomic types (Ribeiro and Walter, 2008), the map scale was defined as 1:25,000. We followed the LCCS criteria based on dominant life form, vegetation cover, and structure, as well as water seasonality (**Figure 2.3**). In the second classification level (**Figure 2.3**), we followed the nomenclature described by Ribeiro and Walter (2008) for our map legend of physiognomic types (**Table 2.1**).

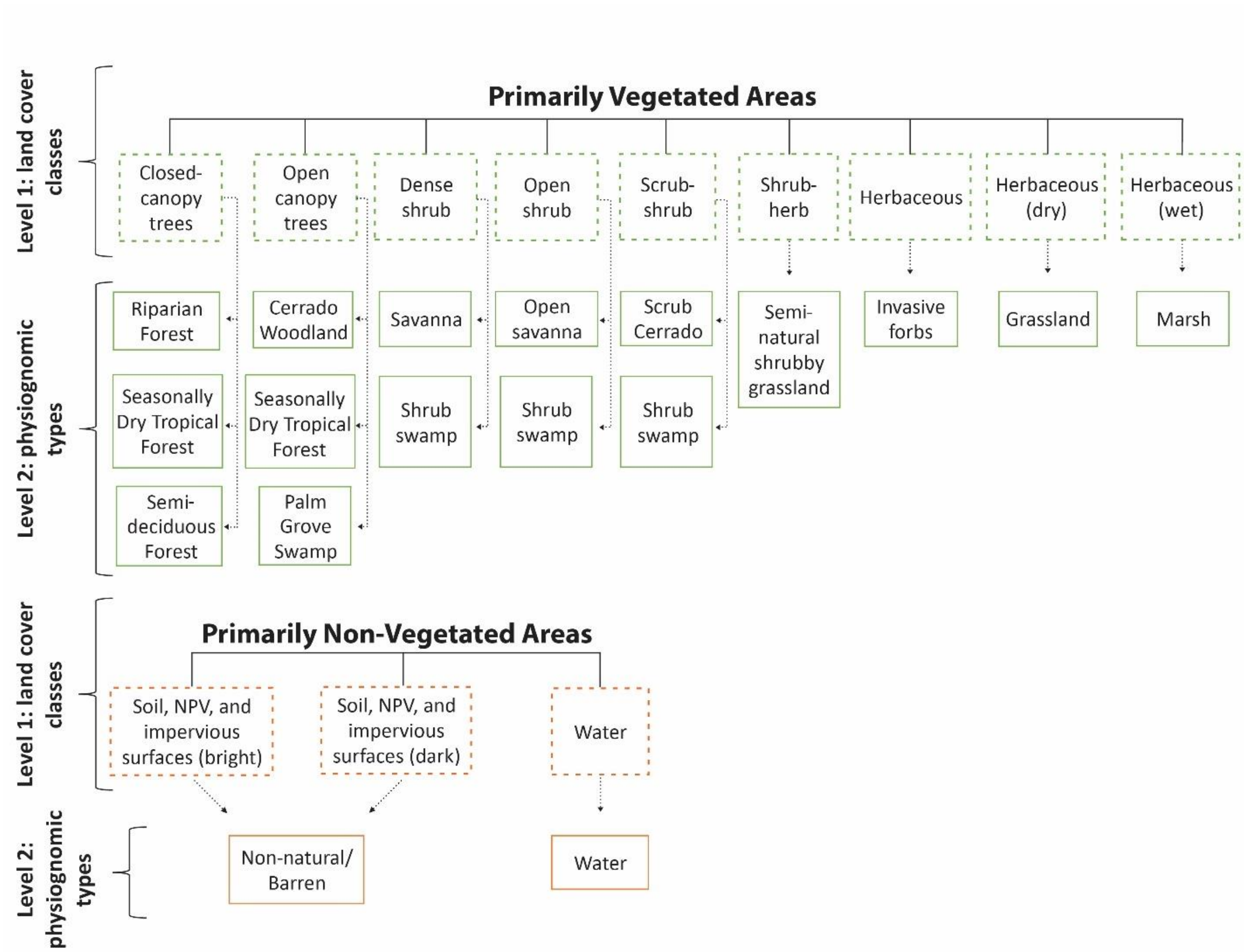

Figure 2.3. Land cover classes used on each classification level. Level 1 represents the Land Cover Classification System (LCCS) classes defined for the Cerrado biome, which is appropriate for mapping with multispectral imagery at fine spatial scales (< 10m). Level 2 represents the corresponding physiognomic types for each LCCS class. The arrows represent the corresponding physiognomic type category (level 2) derived from the LCCS classification (level 1).

Table 2.1. Description of the physiognomic types used in the classification.

Ecosystems	Physiognomic types (English; *Portuguese*)	Description
Forest	Riparian Forest; *mata riparia, mata de galeria, mata ciliar*	Closed-canopy semi-deciduous and evergreen trees following rivers and streams. This class includes gallery forests with a variety of soil moisture regimes

Ecosystems	Physiognomic types (English; *Portuguese*)	Description
	Seasonally Dry Tropical Forest; *mata seca (semi-decidual, decidual, sempre-verde), floresta estacional*	Closed-canopy semi-deciduous, deciduous, and/or evergreen trees across nutrient-rich environments on interfluves. This class is associated with mountainous terrain, such as cliffs
	Semi-deciduous Forest; *mata semi-decidual; cerradão; mata seca*	Closed-canopy semi-deciduous trees with dense layer of xeromorphic shrubs located across flat interfluvial terrain. This class contains tropical dry forest and/or sclerophyll forest, which can occur in different successional stages due to recent deforestation or fire activity
Savanna	Cerrado Woodland; *cerrado denso*	Open canopy semi-deciduous trees over an open herbaceous layer and dense layer of xeromorphic shrubs
	Savanna; *cerrado tipico*	High density of xeromorphic shrubs over an herbaceous layer with scattered to medium density of trees; may contain elements of transition to caatinga vegetation
	Open Savanna; *cerrado ralo; campo sujo*	Low density of xeromorphic shrubs and sub-shrubs over a closed herbaceous layer, which may contain scattered trees throughout the landscape
Grassland	Grassland; *campo limpo, campo limpo com murundus*	Treeless herbaceous layer
	Non-natural Shrubby Grassland; *campo sujo, campo sujo degradado, capoeira*	Sparse xeromorphic shrubs over an open herbaceous layer with strong presence of exposed soil. This class may contain degraded areas, such as abandoned pastures and agricultural areas
	Scrub Cerrado; *campo sujo denso, campo cerrado, scrub*	High density of xeromorphic shrubs and sub-shrubs, with occasional scattered deciduous trees and no presence of herbaceous layer or soil. It may contain elements of transition to caatinga vegetation
Wetlands	Shrub Swamp; *vereda, scrub de vereda*	High to low density of shrubs and sub-shrubs, usually clustered, over a seasonally flooded herbaceous layer
	Palm Swamp; *vereda*	High to low density of palm trees (most commonly *Mauritia flexuosa*), either clustered throughout a seasonally flooded herbaceous layer, or aligned along a water course
	Marsh; *brejo, campo limpo umido*	Seasonally flooded herbaceous layer composed mainly of grass species. This class usually surrounds riparian forests and contains palm and shrub swamps

2.2.2.2. Imagery Acquisition and Pre-Processing

Most RapidEye imagery used in this study was acquired in the 2011 dry season, representing the imagery with the best quality available. The commercial RapidEye sensor, launched in 2009, operates in a constellation of five satellites in the same orbit, providing multispectral images with a spatial resolution of 6.5 meters resampled to a 5-meter grid (at the Level 3A), and a tile size of 25 km by 25 km. The sensor has a swath width of 77 km, daily off-nadir coverage, and radiometric resolution of 12 bits, scaled up to a 16-bit dynamic range. RapidEye's spectral resolution covers the visible and near-infrared bands ranging from 440 to 850 nm, including a red-edge band (690 to 730 nm). The imagery is available through the Ministry of Environment (MMA) Geocatalog and is accessible to Brazilian researchers at no cost. The collection covers the entire country and is composed of varying off-nadir angles and temporal coverage, which is limited to inconsistent dates mostly available for the years 2011 through 2015, depending on the area of interest.

Although the orthorectified Level 3A RapidEye product is provided with radiometric, geometric, and terrain corrections, additional corrections were made for improving consistency in the product. Atmospheric corrections and reflectance retrieval were performed using ACORN 4.0 software for all individual imagery tiles. Calibration files corresponding to image spectral response, gain, and offset, as well as acquisition parameters, were created from the metadata provided. Water vapor and atmospheric visibility parameters were determined by a trial-and-error analysis of a dark object reflectance (e.g., pure water pixels) and following the ACORN user guide suggestions for areas of dry conditions.

In total, we used 17 RapidEye imagery tiles. One tile corresponds to most of the Taquara watershed, covering the IBGE Ecological Reserve and its surroundings (**Figure**

2.1). After applying the atmospheric correction and reflectance retrieval, the image was subset to the bounding box extent of the watershed to facilitate data processing. The other 16 images correspond to the Western Bahia study site (**Figure 2.1**). The pre-processing steps were applied to individual imagery tiles, and tiles with the same acquisition date were mosaicked. The individual mosaics were processed separately, at both classification levels, and merged together to derive statistics for the complete study site (i.e., accuracy assessment and landscape composition). The Taquara tile was acquired on August 11th 2013, and the Western Bahia tiles were acquired from June to October 2011. Details of imagery tiles, dates, and sensor angle-viewing characteristics are summarized in **Table A.1**.

2.2.2.3. Level 1 Classification: Major Land Cover Types

Segmentation

Our GEOBIA approach starts with segmenting the images into homogeneous image objects to ensure neighboring pixel similarity at an adequate scale. The segmentation was performed through the multi-resolution segmentation (MRS) algorithm, proposed by Baatz and Schäpe (2000) and implemented in eCognition Developer 8.0 software. For all images, the segmentation used all five spectral bands of the RapidEye image, with higher weights for near-infrared (NIR) (760–850 nm), red-edge (690–730 nm), and red (630–685 nm) bands due to their importance in discriminating vegetation types (Ferreira et al., 2003; Schuster et al., 2012). The multi-resolution algorithm implemented in eCognition uses a bottom–up merging approach as an optimization procedure to identify similar, homogenous, neighboring pixels and cluster them into a single object. eCognition accounts for a scale parameter as a level of aggregation of image objects and uses a stop criterion in the optimization algorithm. Thus, the scale is a crucial part of GEOBIA, as it defines the size of image objects as well as their level of heterogeneity. The multi-resolution algorithm also accounts for shape and compactness parameters. We visually inspected multiple combinations of scale, shape, and compactness parameters and selected a combination of scale = 10, shape = 0.3, and compactness = 0.7. These parameters are in accordance with other GEOBIA studies that use high spatial resolution imagery and have small image object scale (Myint et al., 2011).

Collection of training data and Random Forest model

The training data were collected through a process of visual interpretation assisted by orthophotos and Google Earth images covering both study sites. We defined standard parameters for visual interpretation of the classes, which were assisted by ancillary data (such as other vegetation maps available for the sites) and one field excursion conducted in the dry season of 2018 to each site for confirmation of class categories in areas that were still unclear after examining the available resources.

Training samples were collected in proportion to class abundance, with abundant classes having a higher number of training data compared to classes that were rare across the landscape (**Table A.2**). All training was done at the object scale, in which each sample corresponds to an image object generated in the segmentation process. Spectral variables (e.g., statistics and indices), also known as object features in GEOBIA, were attributed to each training polygon (**Table 2.2**), including three indices: Normalized Difference Vegetation Index – NDVI (Rouse et al., 1973), Normalized Difference Vegetation Index with red-edge band – NDVI-RE (Schuster et al., 2012), and Normalized Difference Water Index – NDWI (McFeeters, 1996).

All training samples and their respective statistical attributes were used as input in the '*random forest*' package in RStudio developed by Liaw and Wiener (2002) based on Breiman (2001). All parameters were set to default on the random forest classification algorithm, which resulted in a model based on 500 decision trees used to classify all image objects derived from the multi-resolution segmentation. The result of this process is the Level 1 land cover classification based on the LCCS land cover classes.

Table 2.2. Summary of selected features used in the random forest model. NDVI: Normalized Difference Vegetation Index; NDVI-RE: Normalized Difference Vegetation Index with red-edge band; NDWI: Normalized Difference Water Index.

Selected features/ Statistics	Description
Brightness	Sum of mean values of all layers (spectral bands) divided by the total number of spectral bands
Mean Value	Mean (reflectance) value of each spectral band within an image object
Standard Deviation Value	Standard deviation (reflectance) value of each spectral band within an image object
Customized attributes	
NDVI	(NIR–Red)/(NIR+Red)
NDVI-RE	(RedEdge–Red)/(RedEdge+Red)
NDWI	(Green–NIR)/(Green+NIR)

2.2.2.4. Level 2 classification: physiognomic types

In accordance with our goal of classifying physiognomic types in a semi-automatic manner, we developed a series of spatial contextual rules for each study site in order to refine the Level 1 land cover map (**Table 2.3**). We combined the Level 1 LCCS classification with hydrographic data (stream networks and hydromorphic soils) developed by Ribeiro (2011) for the Taquara site (scale 1:10,000); and by the Laboratory of Spatial Information Systems – LSIE at the University of Brasília (Brazil), in partnership with the

Inter-American Institute of Commerce and Agriculture and the Brazilian Ministry of National Integration, for the Western Bahia site (scale 1:2,000). Slope and elevation were derived from the NASA Digital Elevation Model – NASADEM (Crippen et al., 2016) available at a resolution of 1 arc-sec (approximately 30 m).

The spatial rules were developed based on environmental characteristics of the vegetation physiognomic types described in Pereira and Furtado (2011), Nou and Costa (1994), and Ribeiro and Walter (2008), in addition to personal and expert knowledge of the study sites. Specific elevation and slope thresholds were based on recommendations from the Brazilian Agricultural Research Corporation – Embrapa (1979). The same contextual and topological rules were applied for both study sites, except that thresholds used for elevation and slope were adapted to each site's characteristics.

Table 2.3. Spatial contextual rules used to characterize LCCS land cover classes into physiognomic types; rules and classes with "*" were only applied for the Taquara watershed site, and rules with "**" were only applied for the Western Bahia site due to the absence of these classes in the other study site.

Level 1 classes	Spatial rules	Level 2 classes
Closed Canopy	1. Within hydromorphic soils	Riparian Forest
	2. Within steep slopes (>20%) and not adjacent to perennial streams and water (relative border to 'streams' = 0)**	Seasonally Dry Tropical Forest**
	3. Adjacent to streams and water (relative border to 'streams' > 0)	Riparian Vegetation
	4. Within high elevation (>670m) and flat terrain (slope < 8%)**	Semi-Deciduous Forest**
Open Canopy	1. Within hydromorphic soils	Palm Swamp
	2. Within steep slopes (>20%)**	Seasonally Dry Tropical Forest**
	3. All other conditions	Cerrado Woodland

Level 1 classes	Spatial rules	Level 2 classes
Dense Shrub	1. Within hydromorphic soils	Shrub Swamp
	2. All other conditions	Savanna
Open Shrub	1. Within hydromorphic soils	Shrub Swamp
	2. All other conditions	Open Savanna
Scrub–Shrub	1. Within hydromorphic soils**	Shrub Swamp**
	2. All other conditions **	Scrub Cerrado**
Herbaceous (wet)	1. Within hydromorphic soils	Marsh
	2. Within steep slopes (>20%)**	Shade**
	3. All other conditions	Non-Natural/Barren
Herbaceous (dry)		Grassland*
Herbaceous		Invasive Forbs and Shrubs*
Water	1. Isolated small objects (size < 60 pixels) not within hydromorphic soils **	Shade**
	2. Within steep slopes (>20%)	Shade**
	2. All other conditions **	Water**
Shrub–Herbaceous		Semi-Natural Shrubby Grassland**
Soil, NPV***, impervious surfaces (bright)		Non-Natural/Barren
Soil, NPV***, impervious surfaces (dark)		

***Non-photosynthetic vegetation

The combination of mountainous terrain (e.g., escarpments/cliffs) and fine spatial resolution resulted in a high presence of shadows in the Western Bahia site imagery.

However, RapidEye's spectral resolution does not allow us to distinguish shadows from water, a well-known source of confusion in remote sensing and multispectral high resolution images (Franklin et al., 1991). We therefore developed a "shade" mask using a topological rule in eCognition. We also used visual interpretation to create a "cloud" and "shade from cloud" mask, and a "water body" mask was created for the Taquara site, given that this class was rare and small enough (<0.01%) to not be included in the model. Because we were exclusively interested in natural areas, a land use mask from our database (data from Oliveira et al., 2014; Ribeiro, 2011) was used in each study site to exclude paved roads, agricultural, and urban areas from validation.

2.2.5. Accuracy Assessment Procedures

Measuring thematic map accuracy is a crucial step to determine error sources and calculate producer and user map accuracies. Moreover, it is a way to analyze potential weakness and strengths of classification methods. However, it is not a straightforward task and can include many uncertainties (Congalton, 1991; Olofsson et al., 2014; Powell et al., 2004; Richards, 1996; Stehman, 1997). In traditional pixel-based classification, thematic accuracy is assessed by estimating the proportion of correctly classified pixels for each class. This approach assumes that pixels have the same size, and thus, one can estimate the proportion of area correctly classified (Richards, 1996). However, it is recommended that image objects are used as sampling units in object-based accuracy assessments, instead of the traditional point-sampling from pixel-based classification (MacLean and Congalton, 2012). Image objects have variable areas across the landscape in GEOBIA-derived thematic maps, leading to a greater impact on error estimates from large misclassified objects than small polygons. Thus, area count should be accounted for in accuracy assessments of GEOBIA classification (Radoux and Bogaert, 2014).

The Random Forest algorithm generates an out-of-bag (OOB) error estimate using subsampling and bootstrapping, accounting for samples not used as training in the model (Breiman, 2001). To minimize inflated accuracies from the OOB error due to spatial autocorrelation, we performed additional independent validation for both study sites. This independent validation was done by comparing randomly selected polygons from our classified images (excluding training samples) to a series of ancillary data, including orthophotos, Google Earth imagery, and digital photographs (taken on the ground), when available, following recommendations from Richards (1996). Given the lack of fine-scale time series imagery available for the entire landscape, experts with local knowledge of the sites were also consulted to validate the classes that are influenced by seasonality. For instance, the natural seasonality of seasonally dry tropical forest and semi-deciduous forest required local knowledge when the time series of Google Earth imagery was not available. We performed the sampling selection using the original segments/objects derived from eCognition, which contain information at both map levels. The number of samples for each category was determined based on the final map (physiognomic types), but accuracy estimates were performed for both levels using the same polygon. We used an equal proportion of randomly selected polygons for categories that were sparsely represented across the landscape (≤ 10%), which resulted in a total of 50 polygons per class. For classes with high landscape abundance (i.e., non-natural/barren, open savanna, savanna for both sites), we performed a stratified random selection based on a total number of samples of 225 for each site. The classes “semi-deciduous forest” and “shrub swamp” were not accounted for in the validation process for the Taquara site because they each represent less than 0.5% of the landscape. In total, we selected 725 polygons for the Western Bahia site and 525 polygons for the Taquara study site.

A common issue in estimating accuracy from thematic maps is the potential error that can be included in the reference data (Olofsson et al., 2014; Powell et al., 2004; Stehman et al., 2009). To minimize error and bias from the interpreter in the accuracy assessment, our validation procedures were performed by two authors trained in photointerpretation of the regions and with previous experience working in the Cerrado. Error matrices were generated for each map level using an area-weighted approach based on independent sampled image objects (Radoux et al., 2017; Radoux and Bogaert, 2014). The traditional count-based accuracy assessment was also performed for comparison (**Tables A.5–A.8**). They were used to derive traditional statistical accuracy measures for both map levels, such as overall agreement, user's accuracy, and producer's accuracy.

2.3 Results

2.3.1 Segmentation Results

The MRS algorithm generated a different number of image objects (**Figure 2.4**) for each mosaic or image tile processed, which is expected since they have different extents. The imagery tiles acquired in September 16th and September 13th resulted in 384,011 and 389,664 objects, respectively. The August and October mosaics have similar extent (4 image tiles) and resulted in 1,157,419 and 979,083 objects, respectively. The June mosaic contains 6 image tiles and thus resulted in a much larger number of objects, a total of 2,804,338.

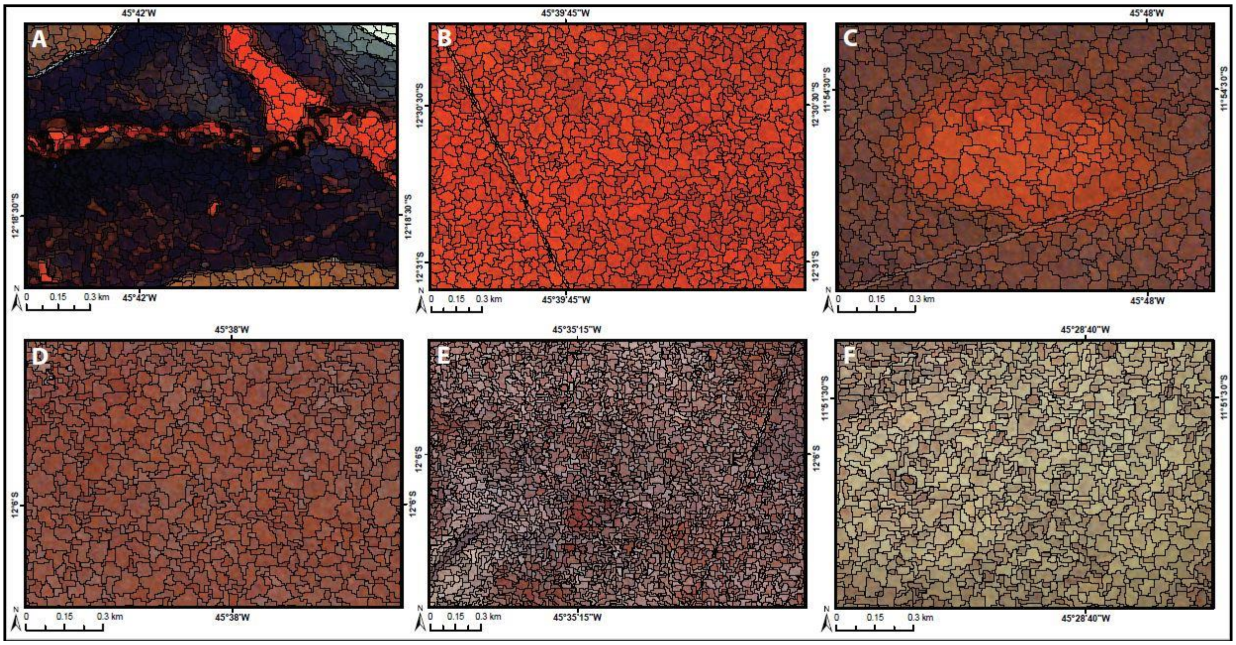

Figure 2.4. RapidEye imagery with color composite as R: NIR (band 5), G: Red-edge (band 4), B: Blue (band 1); and segmentation result (image objects in black) derived from the multi-resolution algorithm, featuring the major land cover types: A) wetland: marsh, riparian forest, palm grove swamp, and shrub swamp; B) open canopy trees (cerrado woodland); C) scrub–shrub (scrub cerrado); D) dense shrub (savanna); E) open shrub (open savanna); and F) shrub–herb (semi-natural shrubby grassland).

2.3.2 Accuracy Assessment

The OOB error for the Taquara site was 7.8%. Given that the Random Forest classification was performed by mosaic for the Western Bahia site, the OOB error estimates were then generated for each mosaic. The September 13th image had the lowest OOB error (3.5%). The October mosaic had the second lowest OOB error (4.4%), followed by the September 16th imagery (6.3%), the August mosaic (7.0%), and the June mosaic (7.3%).

These relatively small differences in the OOB error estimates could be due to a combination of reasons such as atmospheric conditions on a particular day (e.g., active fire was present in the June mosaic, and haze was present in the September 16th imagery), possible rain close to the imagery date, differences in sensor angle-viewing, and particular characteristics of the classified image (e.g., one specific image can have more disturbed areas and hold higher heterogeneity compared to the other).

The mean decrease in accuracy is a percent estimate of variable importance in the random forest model (**Figure 2.5**). The NDVI was the most important variable in all models, except in the June mosaic, in which NDWI had the highest importance. This is likely due to water content available in the soil during early dry season (i.e., June), whereas later in the dry season, some physiognomies (i.e., grass and shrublands) are more impacted by water limitation. The other indices, NDVI (red-edge) and NDWI, also contributed significantly (>12%) in all models. Considering only the RapidEye spectral bands, the near-infrared (band 5) had a contribution above 13% in all models, whereas the red and red-edge bands had high importance (>15%) in the October and June mosaic models, respectively.

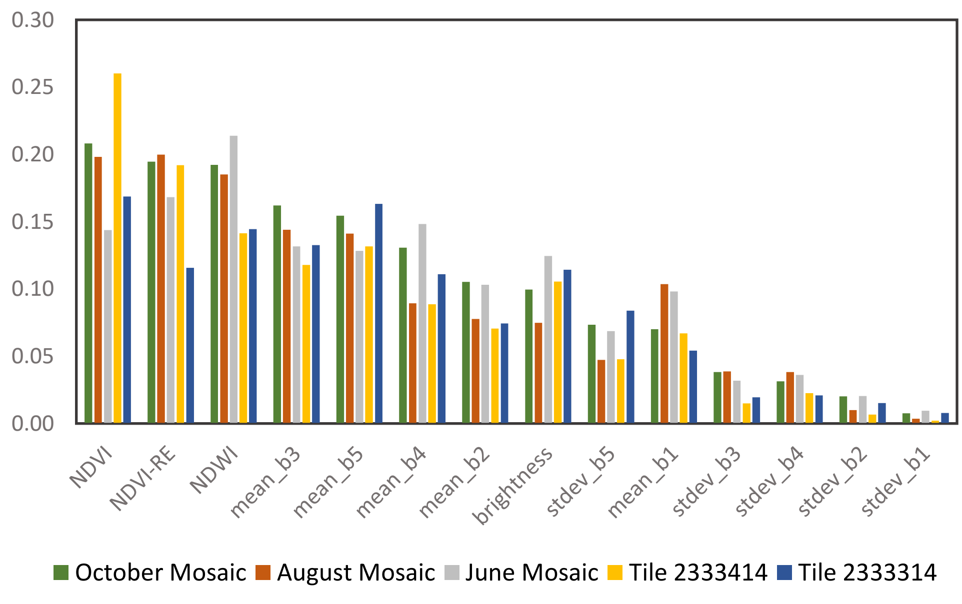

Figure 2.5. Mean decrease in accuracy showing the importance of each variable used in the random forest model, for each processed image.

Additional accuracy estimates based on the independent randomly selected image objects were performed for the entire map and not for individual mosaics. Error matrices were developed for each map level (**Tables 2.4, 2.5**) and were reported as area (in hectares) count of each image object, following best practices for object-based accuracy assessment proposed by Radoux et al. (2017). Statistical measures of overall agreement, as well as user's and producer's estimates were derived from the error matrices (**Tables 2.4, 2.5**). For comparison, we also generated error matrices and statistical measures based on the regular polygon count approach (**Tables A.5–A.8**). Accuracy assessments for the Taquara watershed are found in the supplementary material (**Tables A.3–A.4**).

Table 2.4. Error matrix (reported in ha), overall accuracy, producer's accuracy, and user's accuracy for the Western Bahia Level 1 LCCS classification. The number in parentheses corresponds to the number of independent test samples used for validation.

	Closed canopy (125)	***Dense shrub (90)***	***Herbaceous (wet) (50)***	***Open canopy (125)***	***Open shrub (100)***	***Scrub–shrub (59)***	***Shrub–herb (50)***	***Soil, NPV, impervious (bright) (30)***	***Soil, NPV, impervious (dark) (46)***	***Water (50)***
Closed canopy	**17.8**	0.1	0	2.0	0	0.5	0	0	0	0
Dense shrub	0	**15.8**	0.1	0.5	2.3	0.2	0.4	0	0	0
Herbaceous (wet)	0	0	**12.6**	0.2	0.5	0	0	0	0	0
Open canopy	0.7	1.7	0.5	**13.9**	0	0.3	0	0	0	0
Open shrub	0	1.5	1.8	0	**13.9**	0.3	0.6	0	0	0
Scrub–shrub	0.7	0	0	0	0.4	**7.6**	0.2	0	0	0
Shrub–herb	0	0	0	0	0.7	0	**6.8**	0.1	0	0
Soil, NPV, impervious (bright)	0	0	0	0	0	0	0.3	**2.8**	1.9	0
Soil, NPV, impervious (dark)	0	0	0	0	0	0	0.9	0.1	**9.6**	0
Water	0	0	1	0	0	0	0	0	0	**7.7**
Overall accuracy (%)	84.4									
Producer's accuracy (%)	92.5%	82.4%	81.3%	83.6%	77.7%	86.1%	74.4%	95.4%	83.6%	100%
User's accuracy (%)	87.3%	81.7%	94.5%	80.8%	77.1%	85.2%	89.7%	56.6%	91.0%	93.3%

Table 2.5. Error matrix (reported in ha), overall accuracy, producer's accuracy, and user's accuracy for the Western Bahia Level 2 classification. The number in parentheses corresponds to the number of independent test samples used for validation.

	Cerrado woodland (50)	***Marsh (50)***	***Non - natural/ barren (76)***	***Open savanna (74)***	***Palm swamp (50)***	***Riparian forest (50)***	***Savanna (75)***	***Scrub cerrado (50)***	***Semi-Deciduous forest (50)***	***Shrubby grassland (50)***	***Shrub swamp (50)***	***Tropical dry forest (50)***	***Water (50)***
Cerrado woodland	**6.2**	0	0	0	0	0.2	0.7	0.3	0.1	0	0	0	0
Marsh	0	**12.6**	0	0	0.2	0	0	0	0	0	0.5	0	0
Non-natural/ Barren	0	0	**14.3**	0	0	0	0	0	0	1.2	0	0	0
Open savanna	0	0	0	**11.7**	0	0	1.3	0.3	0	0.6	0	0	0
Palm swamp	0.1	0.5	0	0	**5.0**	0.4	0	0	0	0	0.7	0	0
Riparian forest	0.2	0	0	0	0	**5.9**	0	0	0	0	0.3	0	0
Savanna	0	0	0	1.5	0	0	**15.2**	0	0.3	0.4	0	0	0
Scrub cerrado	0	0	0	0.4	0	0	0.0	**6.7**	0.6	0.2	0	0	0
Semi-Deciduous forest	0.4	0	0	0	0	0	0.1	0.2	**8.5**	0	0	0	0
Shrubby grassland	0	0	0.1	0.7	0	0	0	0	0	**6.8**	0	0	0
Shrub swamp	0	1.9	0	0	0.2	0.1	0.1	0	0	0	**5.0**	0	0
Tropical dry forest	0.3	0	0	0	0	0	0.4	0	0	0	0	**7.1**	0
Water	0	0.6	0	0	0	0	0	0	0	0	0	0	**7.7**
Overall accuracy (%)	87.6												
Producer's accuracy (%)	85.8%	81.3%	99.5%	81.8%	92.4%	89.4%	85.5%	89.5%	89.4%	74.4%	77.3%	100%	100%
User's accuracy (%)	81.7%	94.5%	92.5%	84.4%	75%	92.3%	87.5%	84.2%	92.9%	89.7%	69.2%	91.4%	93.3%

2.3.3 Landscape Composition: Area Assessments

We estimated landscape composition (**Table 2.6**) for the total area of the Western Bahia site, as well as for each mosaic, based on the final (Level 2) physiognomic type map (**Figure 2.6**). The most abundant classes in the landscape are non-natural/barren areas, open savanna, and savanna. The rarest physiognomic types found are seasonally dry tropical forest, shrub swamp, and palm swamp. Considering the individual mosaics, the June mosaic and the September 13th image have the highest amount of natural vegetation, whereas the October mosaic has the highest concentration of non-natural/barren areas. The Level 2 classification map and landscape composition of the Taquara watershed is found in the supplementary material (**Figure A.1**, **Table A.9**).

Table 2.4. Estimate of landscape composition for the Western Bahia site considering the proportion of the mapped area of each physiognomic type, reported in percentage, with respect to the total mapped area (i.e., entire study site extent) and for each individual image/mosaic.

	Total (%)	**September 13th image**	**September 16th image**	**June mosaic**	**August mosaic**	**October mosaic**
Cerrado Woodland	4.9	4.2	3.2	7.0	1.7	5.9
Marsh	1.2	0.2	0.0	0.3	1.4	2.9
Non-Natural/Barren	26.6	14.5	31.6	11.5	33.2	43.9
Open Savanna	26.2	35.9	32.1	22.9	37.2	15.5
Palm Swamp	0.8	0.1	0.1	0.2	0.7	2.0
Riparian Forest	1.1	0.4	0.4	1.3	0.8	1.4
Savanna	26.2	39.4	14.6	40.0	11.1	20.9
Scrub Cerrado	1.9	0.0	0.0	4.4	1.2	0.0
Semi-Deciduous Forest	1.5	0.5	0.1	3.4	0.4	0.2
Shade	0.1	0.0	0.0	0.3	0.1	0.0
Semi-Natural Shrubby Grassland	8.0	4.2	17.1	6.3	10.4	6.6
Shrub Swamp	0.6	0.1	0.1	0.5	1.3	0.2
Seasonally Dry Tropical Forest	0.6	0.2	0.4	1.5	0.1	0.0
Water	0.3	0.5	0.3	0.2	0.3	0.3

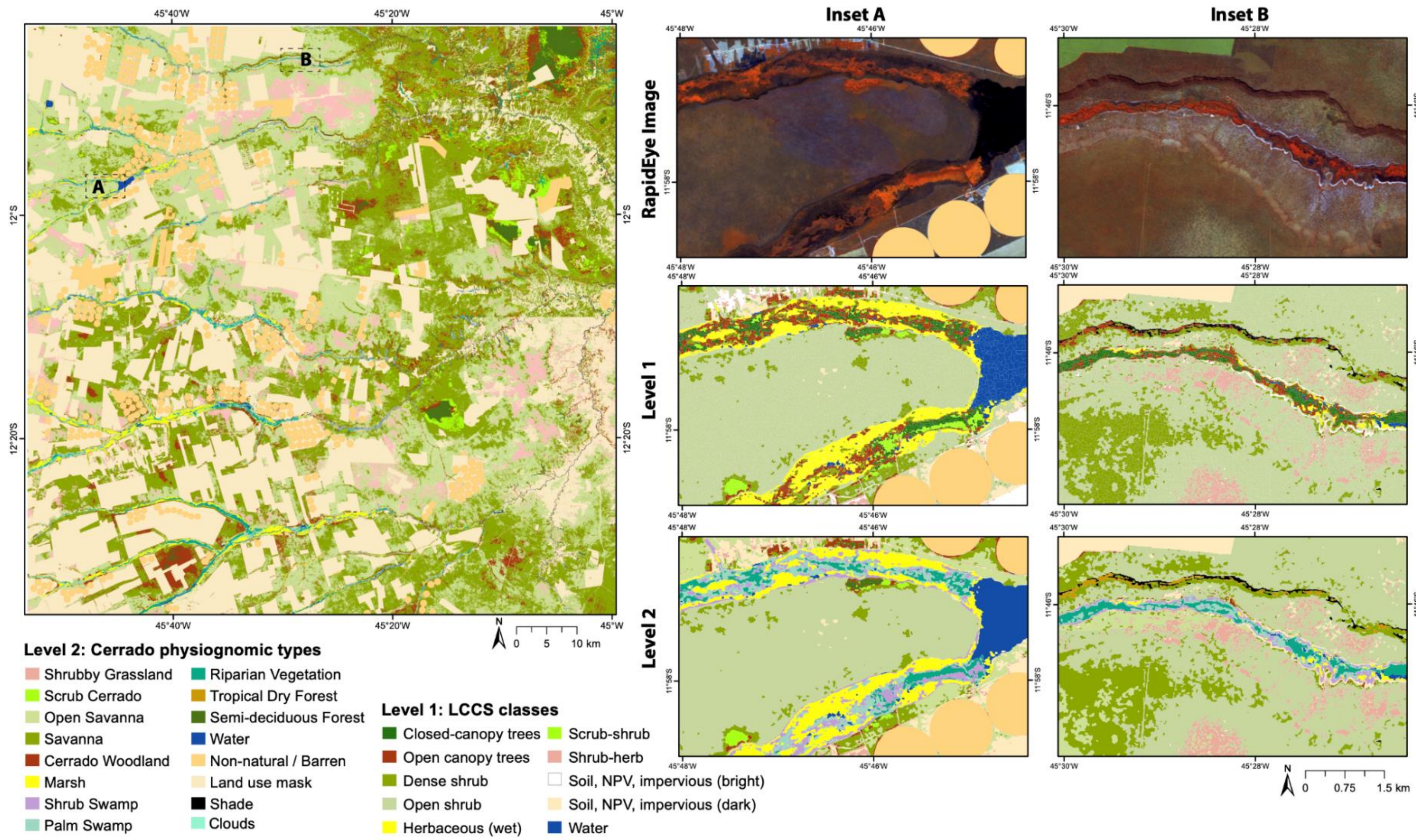

Figure 2.6. Final map of physiognomic types (Level 2) for the Western Bahia study site, with insets A and B showing zoomed in examples of classification Levels 1 and 2.

2.4 Discussion

Land cover mapping in the Cerrado has generally used multispectral imagery with medium to coarse spatial resolution and pixel-based approaches, such as in Muller et al. (2015), Schwieder et al., (2016), Ferreira et al. (2007), and Reynolds et al. (2016). As demonstrated by Sano et al. (2010) and Schwieder et al. (2016), these types of imagery do not capture the fine-scale heterogeneity present within the savanna ecosystem gradient and thus are not appropriate to discriminate differences in vegetation structure, often leading to low accuracy results (such as 71% and 63%, respectively). Distinguishing the fine-scale heterogeneity of Cerrado physiognomic types is crucial for identifying species habitats and estimating plant diversity. For instance, the Hyacinth Macaw (*Anodorhynchus hyacinthinus*) is an endangered species of small population inhabiting the Cerrado that is heavily dependent on palm trees present in wetlands for breeding and foraging; discriminating the different structural types present in seasonal wetlands (i.e., palm swamp, shrub swamp, and marsh) can improve estimates related to their occurrence and habitat quality and availability. Given the challenges of mapping Cerrado physiognomic types with traditional pixel-based methods at medium to coarse spatial resolution, we developed a systematic GEOBIA framework using single-date high spatial resolution imagery accounting for a novel environmental spatial ruleset developed to identify a wide range of Cerrado vegetation structural types. This framework was shown to be a robust method to differentiate a larger number of physiognomic types at a higher accuracy than previously reported in several studies regarding Cerrado land cover mapping.

Our results show an improvement in classification accuracy compared to studies using similar image characteristics and object-based methods to map Cerrado physiognomic types, such as in Girolamo-Neto et al. (2018, 2017) and Orozco-Filho (2017). In the Level 1 LCCS classification, we mapped a total of 10 land cover classes (of which 7 correspond to vegetation types) and reached 82% overall accuracy, while others have discriminated 8 classes and reached an overall accuracy of 67.7% (Girolamo-Neto et al., 2018), or 81% accuracy while considering 7 classes (Orozco Filho, 2017). Girolamo-Neto (2017) used the RapidEye imagery in a method similar to ours (i.e., segmentation + RF classifier) in the Level 1 LCCS classification, but with a different class legend, and reached an overall accuracy of 74.3% to classify 5 land cover classes. Previous studies aiming to classify Cerrado physiognomic types used vegetation taxonomy based on structural parameters and species composition defined either by Coutinho (1978), IBGE (2012), Ribeiro and Walter (2008), or a combination of them. The disparity in map accuracy and number of discriminated classes between our LCCS results and previous studies suggest that nomenclatures considering floristic composition (used in most previous Cerrado remote sensing studies) might not be appropriate for multispectral imagery alone as we initially suspected. It is important to note that standardization of land cover classes—that are comparable across scales and appropriate to the imagery characteristics (e.g., spectral and spatial resolutions)—is essential to produce accurate and meaningful results. As in other studies aiming to standardize land cover classes for remote sensing applications (Jansen and Gregorio, 2002; Kosmidou et al., 2014), we used the LCCS to define appropriate Cerrado land cover classes for the RapidEye imagery and tested if the defined classes could be spectrally discriminated and mapped at high accuracy.

Given that there is no other study using LCCS classes for the Cerrado, we cannot compare accuracy assessments for specific classes from our Level 1 classification with other studies. However, our results demonstrate that accounting for RapidEye's spectral information alone accurately discriminates our defined LCCS land cover classes, distinguishing some structural variation within savanna (i.e., open shrubland; dense shrubland; open canopy) and grassland (i.e., herbaceous; shrub–herbaceous) ecosystems. Despite encouraging results, single-date RapidEye spectral properties alone were not able to discriminate variations within forest structural elements (i.e., riparian forest versus semi-deciduous forest) given that most closed-canopy classes (e.g., seasonally dry tropical forest and sclerophyll forest) are composed of broad-leaf semi-deciduous (or deciduous, for a subtype of seasonally dry tropical forest) trees. It could also not differentiate some variations between terrestrial ecosystems and seasonal wetlands. For instance, shrublands (i.e., dense and open shrub) could not be distinguished from wetland shrubs (i.e., shrub swamp). Despite that, grasslands (i.e., herbaceous) were distinguished from marsh (i.e., herbaceous–wet) accounting only for its spectral properties. This result is consistent with previous studies that demonstrated an improvement in classification accuracy of terrestrial and wetland ecosystems when using multispectral fine spatial resolution imagery (McCarthy et al., 2018).

Defining environmental contextual rules in addition to using spectral properties proved an effective strategy in discriminating within-class variations across ecosystems as confirmed by high accuracy results for those classes. Other works also accounted for such classes but frequently merged similar physiognomic types into one class for higher accuracy estimates. For instance, our classes "marsh", "palm swamp", and "shrub swamp", if merged, would be equivalent to the class “floodplains with palm trees” in Girolamo-Neto (2018), and “veredas” in Orozco-Filho (2017). The same is true for our classes “riparian forest”, “seasonally dry tropical forest”, and “semi-deciduous forest”, which are equivalent, if merged, to the class “forest” in Orozco-Filho (2017). It is known that map accuracy tends to decrease as a function of the number of classes (Smith et al., 2002); however, our GEOBIA approach showed an inverse pattern, which is a major contribution of this study. Applying our environmental spatial ruleset to the Level 1 LCCS map resulted in a thematic map (Level 2 classification) with a larger number of classes and a higher overall agreement accuracy. Despite the fact that two classes of the Level 1 LCCS map were merged in the physiognomic types map (both “soil, NPV, impervious” classes became “non-natural/barren”), four new classes were added to the Level 2 map and the overall accuracy improved by around 3%. Considering both user’s and producer’s estimates for the physiognomic types classification (**Table 2.5**), the highest accuracies (>80%) are among the classes non-natural/barren, marsh, seasonally dry tropical forest, riparian forest, savanna, and open savanna. In general, all classes representing savanna and forest ecosystems resulted in a high (>80%) producer’s accuracy.

Most studies aiming to test methodological approaches to map Cerrado physiognomic types were developed for one study area, usually of small extent (<50,000 ha) and not covering some major physiognomic types, such as in Ferreira et al. (2007), Teixeira et al. (2015), and Girolamo-Neto (2018, 2017), which can be problematic for making portability assumptions to other Cerrado areas. Exceptions include studies from Schwieder et al. (2016), who tested methods for three study sites of similar vegetation composition, and Silva and Sano (2016) that considered four small test sites of different composition to map three major vegetation classes (i.e., savanna, forest, grasslands). To bring a higher level of confidence in testing the portability of our GEOBIA framework to other regions within the core area of the Cerrado, our method was tested for two study sites: a control site with larger availability of datasets (i.e., the Taquara site), and another covering a larger extent (>900,000 ha) and supporting different composition and heterogeneity levels, covering a total of 11 major physiognomic types that are present across the Cerrado. The high accuracy results for both study sites indicate that this framework should be portable to other areas in the core Cerrado region. However, further analysis is necessary to adjust it for areas of transition to other biomes where unique local flora composes additional physiognomic types (e.g., *carrasco*, *capão*). In addition, we suggest future studies to explore adapting this framework to similar ecosystems in other continents, such as the African and Australian savannas.

Despite its robust ability to classify a wide range of physiognomic types, our method was not able to differentiate classes of similar structure for which edaphic conditional drivers were not available in our dataset. This is the case for classes that would be separable from each other with detailed information about soil types and/or species composition. For instance, seasonally dry tropical forest located in areas of flat terrain (plateaus/mesas) could not be differentiated from sclerophyll forest, which co-occurs in the same terrain type, so they were combined into a single semi-deciduous forest class. We could only identify seasonally dry tropical forests within steep slopes, which could be discriminated using a fine-scale Digital Elevation Model. Additionally, transitional enclaves of denser caatinga vegetation (a deciduous xerophyte type) were also not possible to differentiate from savanna. Similarly, a rocky savanna type (cerrado rupestre), which is present in the Western Bahia region and structurally similar to open savanna, could not be discriminated.

Recent advances in remote sensing, such as imaging spectroscopy and Light Detection and Ranging (LiDAR), are improving vegetation studies in savannas (Baldeck et al., 2014; Baldeck and Asner, 2013; Vaughn et al., 2015). They have great potential to overcome gaps and uncertainties related to savanna patterns and processes, such as species discrimination (Cho et al., 2012; Colgan et al., 2012; Naidoo et al., 2012) and plant community composition (Baldeck et al., 2014), as well as major drivers and impacts on woody structure (Asner et al., 2009; Vaughn et al., 2015). A potential solution for overcoming remaining issues related to Cerrado structure and floristic composition would be the availability of a detailed (< 1:10,000) soil types map or a combination of LiDAR and hyperspectral imagery. Moreover, publicly available multispectral imagery, such as the Sentinel-2 MSI sensor, are also promising to improve discrimination of Cerrado physiognomic types due to their combination of fine spatial and spectral properties, free availability, and larger areal coverage allowing for regional scale analysis.

2.5 Conclusions

The semi-automatic method proposed here combines image spectral properties (mean reflectance, standard deviation) with standard spectral indices (e.g., NDVI, NDWI) in a Random Forest land cover classification, and uses a novel spatial contextual ruleset to classify land cover categories into physiognomic types in a systematic manner. Our study demonstrates that high spatial resolution imagery is appropriate for discriminating Cerrado land cover classes. The Random Forest algorithm was effective in mapping structural differences within savanna ecosystems, in addition to distinguishing wetlands from terrestrial ecosystems. A combination of ancillary data and spatial rules, however, allowed characterizing physiognomic types while increasing the number of classes and improving map accuracy. Despite the demonstrated success of our method, caveats include high computational costs for processing a large volume of data, lack of automated methods to determine MRS initial parameters (scale, shape, and compactness), and low temporal availability for RapidEye data available at no cost for monitoring purposes and for improving discrimination of classes.

Detailed maps differentiating physiognomic types are essential for conservation strategies, and a consistent classification method is currently lacking in the Cerrado. To the best of our knowledge, our study is the first to propose a systematic method to map Cerrado physiognomic types resulting in a high accuracy assessment and a large number of classes for areas of different heterogeneity. Thus, we conclude that the proposed framework is effective to accurately map physiognomic types across the Cerrado biome at fine spatial scales. Given the availability of RapidEye data for the entire Brazilian Cerrado, application of our framework could improve region-wide mapping in support of conservation and ecological analysis.

Acknowledgments

This research was funded by the Coordenação de Aperfeiçoamento de Pessoal de Nível Superior (CAPES) Foundation, Ministry of Education of Brazil, through the Science Without Borders program, grant numbers 13684/13-2 (F.F.R.) and 13711/13-0 (G.A.D.). We would like to thank Bruno Machado Teles Walter for providing support in defining the LCCS land cover classes and their associations to Cerrado physiognomic types; Otacilio Santana for providing assistance with identifying the physiognomic types present in the study sites and for providing field work data for the Western Bahia site; the IBGE Ecological Reserve and Mauro Lambert Ribeiro for permission to conduct research, support with field work, and granting access to their geospatial database; Aline Leão and The Nature Conservancy Brazil for providing field work data and granting access to their geospatial database for the Western Bahia site; Sandro de Oliveira, Osmar Abílio de Carvalho Junior, and the Laboratory of Spatial Information Systems (LSIE) at the University of Brasilia for kindly sharing their geospatial database for the entire Western Bahia region; Tatiana Kuplich from the Brazilian National Institute for Space Research (INPE) for providing the RapidEye imagery used in this study; and Natasha Picciani for assisting with high quality figures.

Chapter 3. Is the Brazilian Forest Code protecting Neotropical savanna habitats? A fine-scale gap analysis of a Brazilian Cerrado agricultural landscape

Fernanda F. Ribeiro[1], Frank W. Davis[2], Gabriel A. Daldegan[1], Fernanda T. Brum[3], Nadinni O. M. Sousa[4,5], Laura L. Hess[6], Kelly K. Caylor[1,2], Dar A. Roberts[1]

[1]Department of Geography, University of California, Santa Barbara, CA 93106, USA
[2]Bren School of Environmental Science and Management, University of California, Santa Barbara, CA 93106, USA
[3]Programa de Pós Graduação em Ecologia e Conservação, Universidade Federal do Paraná, Curitiba, PR 19031, Brazil
[4]Programa de Pós-graduação em Biologia Animal, Universidade de Brasília, Brasília, DF 70910, Brazil
[5]Secretaria de Biodiversidade, Ministério do Meio Ambiente, Brasília, DF 700068, Brazil
[6]Earth Research Institute, University of California, Santa Barbara, CA 93106, USA

3.0 Abstract

Conservation in commodity-driven agricultural landscapes is particularly challenging due to conflicting interests with economic growth and high costs of land acquisition. Therefore, privately managed lands play an essential role in ensuring species habitat protection, especially in regions with high deforestation rates. The Brazilian Forest Code policy regulates land use and management practices in private lands throughout the country, thus offering great opportunity to protect biodiversity in regions under commodity-driven agricultural expansion, such as the Cerrado biome. In this study, we quantified the benefits of the Forest Code to biodiversity conservation in Cerrado agricultural areas. We developed a fine-scale spatially explicit gap analysis approach to estimate the potential biodiversity conservation gaps from the Forest Code policy in a Cerrado landscape under intensive farming activities for commodity production. We then investigated how private properties can help address conservation gaps by analyzing their compliance status with the Forest Code. Finally, we developed a simulated spatial planning analysis to understand how set-asides allocation in private lands can be improved to maximize biodiversity conservation. Our results demonstrate that the Forest Code is essential for guaranteeing protection of Cerrado's biodiversity in agricultural landscapes. For our study area, this policy protected > 18% of Cerrado physiognomic types, provided > 30% of habitat protection for all bird species in our list, and protected 100% of primary habitats for 2 endangered and 1 endemic birds. Our study indicates that large and medium farms hold the largest proportion of unprotected vegetation as well as the largest non-compliant areas with the Forest Code. Thus, public policies can target farm category to improve law enforcement efforts in Cerrado commodity-driven agricultural landscapes. In addition, the simulated spatial

planning analysis demonstrates that improvements can be made in set-asides allocation to maximize biodiversity protection without compromising agricultural activities.

3.1 Introduction

When effectively managed for protection of natural resources and ecosystem services, private lands can provide essential habitat for species and can improve connectivity between protected areas (Kamal and Grodzinska-Jurczak, 2014). As such, privately managed lands have an essential role in climate change mitigation and biodiversity conservation strategies, while also providing economic growth (Assunção and Chiavari, 2015; Klink, 2019; Soares-Filho et al., 2014). Planning for conservation in rural agricultural landscapes is, however, particularly challenging due to private property rights, high costs of land acquisition, and lack of financial incentives to landowners (Naidoo et al., 2006; Schuster et al., 2018). Strategies such as land-use regulation, public policies, law enforcement, and international market demands for voluntary zero-deforestation commitments (i.e. the Amazon Soy Moratorium: Gibbs et al., 2015) provide incentives for land use and conservation planning within agricultural landscapes (Assunção and Chiavari, 2015; Klink, 2019).

Brazil's Forest Code, created in 1965 and last updated in 2012 (Federal law 12.727/2012), is part of the Federal environmental legislation that regulates land use and management practices on private lands (Soares-Filho et al., 2014). This land-use policy requires landowners to set aside part of their rural property for protection (80% in the Amazon, 35% for non-forested areas within the rainforest-savanna transition zone, and 20% in all other biomes); these legally protected fractions of the property are called Legal Reserves. In addition, this legislation guarantees the protection of sensitive areas, such as water resources, hilltops, and plateau escarpments through Areas of Permanent Preservation (Soares-Filho et al., 2014). Thus, the Forest Code offers critical opportunities to realize the potential benefits of conservation efforts on private lands in Brazil. However, those benefits depend on whether Forest Code protection is effective in both representing and contributing to the persistence of biodiversity across the suite of environments and habitats in areas dominated by commodity production.

The Forest Code policy is essential for protecting ecosystems, natural resources, and ecosystem services, and for improving connectivity between protected areas (Soares-Filho et al., 2014). Legal Reserves alone protect ~167 Mha of natural vegetation across Brazil, covering a larger area than the total protected areas network of the country (~151 Mha: Metzger et al., 2019). Aside from areas under Forest Code protection, more than 100 Mha of native vegetation in Brazil are within private lands and can be legally converted to anthropogenic activities (Metzger et al., 2019; Soares-Filho et al., 2014; Strassburg et al., 2017). Around 50% of these areas are within the Cerrado biome (Metzger et al., 2019), a Neotropical savanna biodiversity hotspot that originally covered more than 2 million km^2 in the central Plateau of Brazil. It is estimated that if all unprotected land in the Cerrado were completely cleared, about 385 million tons of carbon stock would be added to the atmosphere budget (Vieira et al., 2018). This estimate represents more than double the carbon stocks preserved by the current Cerrado protected areas network (Medeiros and Young, 2011).

Despite restricting land-clearing on private lands, the Forest Code alone does not guarantee full compliance by landowners or even deforestation/land-clearing reduction. High deforestation/land-clearing rates are common practices in Brazil, mainly due to financial agricultural incentives and lack of policy enforcement (Azevedo et al., 2017, 2015). As a consequence, the Forest Code policy traditionally shows low compliance rates, especially by small landholders (Michalski et al., 2010; Zimbres et al., 2018). Previous studies have quantified illegal deforestation in areas protected by this policy and evaluated its compliance status at the rural property level. This is well documented for southern and eastern Amazon states, such as Mato Grosso (Michalski et al., 2010; Richards and VanWey, 2016; Stickler et al., 2013; Zimbres et al., 2018) and Pará (Azevedo et al., 2017), where monitoring programs such as PRODES (the official deforestation monitoring program from the Brazilian National Institute for Space Research – INPE) are well established. In terms of farm compliance status, it is estimated that about 26% of all Cerrado areas under Forest Code protection are non-compliant with the legislation, and full compliance would require restoration of nearly 5Mha of cleared land (Vieira et al., 2018). Deforestation estimates in areas under Forest Code protection are less common for other biomes, such as the Atlantic Forest (but see Rezende et al., 2018) and the Cerrado (but see de Oliveira et al., 2017a; Stefanes et al., 2018; Vieira et al., 2018). Moreover, the conservation status of biodiversity (i.e. physiognomic types, species-specific suitable habitats) within areas protected by the Forest Code has not been quantified.

Gap analysis is an efficient method for identifying conservation gaps in biodiversity management areas. Conservation gap analysis can be useful for establishing new reserves or improving land management practices by estimating the distribution and conservation status of several components of biodiversity (Scott et al., 1993). This analysis has been used to estimate gaps in the protected areas networks – both public and private – of many regions (Davis and Stoms, 1996; Fearnside and Ferraz, 1995; Goettsch et al., 2019; Scott et al., 2001) including in the Cerrado protected areas system (Carvalho et al., 2017; de Oliveira et al., 2017a; Marini et al., 2009).

This study quantitatively analyzes the effectiveness of Brazil's Forest Code for achieving a representative, well-connected network of protected sites in a commodity-driven agricultural subset of the Cerrado biome. First, we conduct a fine scale spatially explicit gap analysis to estimate possible protection status of physiognomic types and important habitats for endangered and endemic avifauna. Using those data, we also: 1) estimate the potential land conversion within Forest Code areas that might be set for restoration, 2) assess the potential compliance status of rural properties with the Forest Code by farm size categories, and 3) develop a simulated spatial planning analysis of set aside allocation for improving policy guidelines.

3.2 Materials and Methods

3.2.1 Study Site

The Cerrado is a biodiversity-rich biome composed of a high diversity of physiognomic types embedded in a vegetation mosaic of forest, savanna, and grassland ecosystems (Oliveira-Filho and Ratter, 2002; Ribeiro and Walter, 2008). Supporting the highest plant diversity amongst all savannas, the Cerrado has more than 12,000 plant species of which 44% are endemic (Mendonça et al., 2008; Ratter et al., 1997). It is estimated that ~50% of its original area has been converted to other human activities (http://terrabrasilis.dpi.inpe.br/en/publications/ last accessed in 09/15/2019) and just ~20% remains undisturbed (Strassburg et al., 2017).

Currently, the most active agricultural frontier in the Cerrado consists of 337 municipalities in the States of Maranhão, Tocantins, Piaui, and Bahia, a region known as "Matopiba" (Brazilian Decree n. 8447). Agricultural expansion targeting commodity crops such as soybeans, cotton, and coffee, and extensive beef production, plays a central role in land use change across the biome and especially in the Matopiba (Klink and Machado, 2005; Spera et al., 2016). This region is responsible for around 10% of the national crop production (IBGE, 2015; Miranda et al., 2014) and about 45% of the total forest carbon emissions in the Cerrado between 2010 and 2013 (Noojipady et al., 2017).

Our study area is part of the Matopiba agricultural frontier, spanning an area of 9,409 km^2 on the western side of Bahia State (11°43'S, 45°52'W to 12°36'S, 44°59'W, **Figure 3.1**). The area under analysis was delineated based on the availability of high-resolution imagery to produce a fine-scale land cover classification distinguishing Cerrado physiognomic types. The study region features two Key Biodiversity Areas (KBAs: Sawyer et al., 2017) that overlap two other national conservation priority areas identified by federal institutions and the major stakeholders involved (WWF-Brazil, 2015). In addition, with the highest commodity grain productivity and Gross Domestic Product (GDP) within Matopiba (IBGE, 2015), the study region is economically and politically significant, as well as ecologically relevant.

The western portion of the study area is composed mainly of plateaus at a maximum elevation of 808 m, whereas the eastern portion contains lowlands (known as *Depressão do Rio São Francisco*) with a minimum elevation of 433 m (Nou and Costa, 1994). Annual precipitation patterns in western Bahia have a positive correlation with elevation, ranging from ~800 mm in lowlands to ~1,600 mm across the plateaus (Brannstrom et al., 2008). Thus, rainfall declines from west to east, a gradient that is a major determinant in land price and crop production. This region is one of the most productive areas of the Cerrado, with soybean (71%) being the dominant crop type. It covers part of the municipalities playing a major role in Brazil’s agricultural production: Barreiras, Luis Eduardo Magalhães, São Desidério, and Riachão das Neves, which together represented 3% of Brazil’s soybean production in 2018 (https://sidra.ibge.gov.br/pesquisa/pam/tabelas last accessed in 08/21/2020).

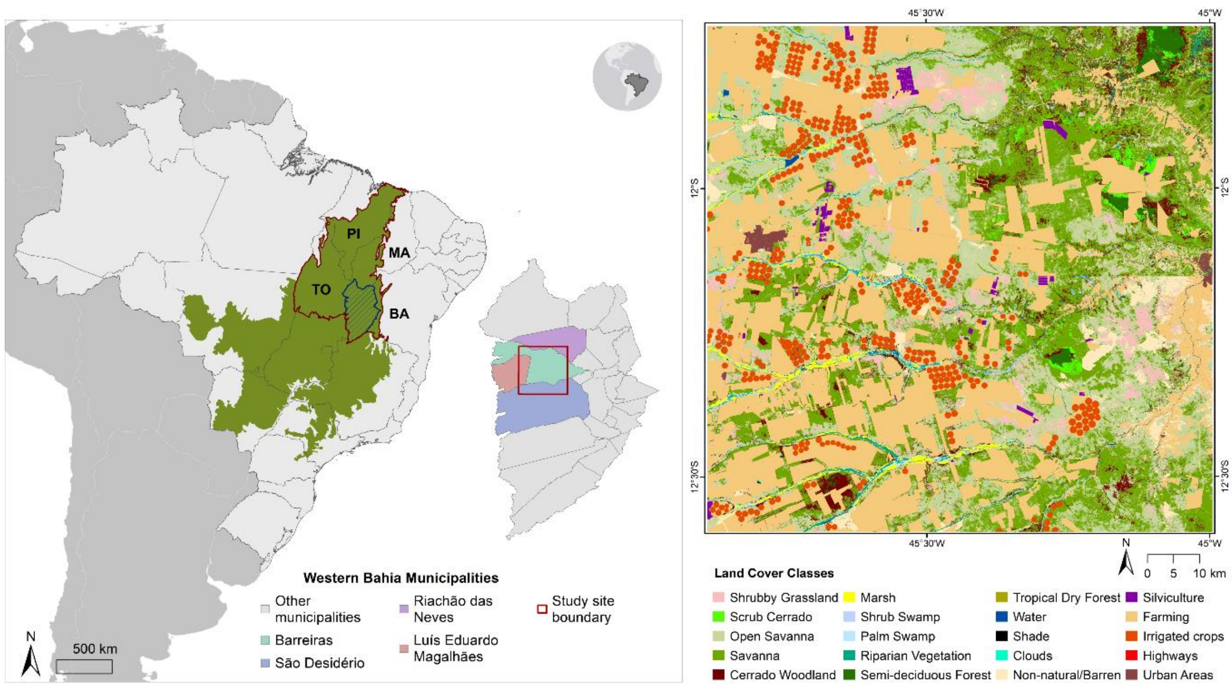

Figure 3.1. Location map of the study area in the western part of Bahia State, part of the Matopiba region (region defined by the acronym of the states of **Ma**ranhão, **Pi**auí, **To**cantins, and **Ba**hia). The inset land cover map features vegetation physiognomic types mapped at 5-meter resolution, developed by Ribeiro et al. (2020).

3.2.2 Datasets

We used a land cover product from Ribeiro et al. (2020) based on 2011 RapidEye imagery. This product contains 11 classes of Cerrado physiognomic types, in addition to the following classes: water, shade (mostly representing topographical and tree shadows), clouds, non-natural/barren (representing barren agricultural areas, unpaved roads, exposed soil, and potentially degraded areas). In order to account for updated land use information, we used two other products to complement our land cover product: a) a land use map from the Brazilian Foundation for Sustainable Development (FBDS) based on 2013 RapidEye imagery, and b) a land use map based on visual interpretation of PRISM/ALOS imagery for 2009 (updated by de Oliveira et al., 2017a for 2011). This map was developed at a scale of 1:2,000 by the Laboratory of Spatial Information Systems (LSIE) at the University of Brasilia (UnB) in partnership with the Inter-American Institute of Commerce and Agriculture (IICA), and the Ministry of National Integration. We masked out active crops and farming activities from our land cover map, in addition to urban areas and major highways. From the FBDS LUC product, we used the classes “farming” (converted areas), “silviculture”, and “urban areas” as a mask in our land cover map. For the LSIE LUC product, we only considered the class “irrigated cropland” in order to distinguish planted irrigated crops from other farming activities and “major highways”. In addition, we used a separate geospatial dataset derived from the same LSIE project providing information on stream networks, hydromorphic soils, springs, and the plateau border.

We calculated the Areas of Permanent Preservation buffer zones following the definition in the 2012 legislation. The Legal Reserve boundary datasets were downloaded from the Environmental Registry System (SICAR: www.car.gov.br last accessed on 06/08/2019), an online federal platform providing georeferenced data for all rural properties in Brazil. SICAR has improved transparency for monitoring compliance status with the Forest Code, providing a new pathway to understand the implications of land use regulation on retaining biodiversity and providing solutions to increase sustainability in agricultural landscapes (Soares-Filho et al 2014). In order to evaluate policy compliance status of rural properties, we consolidated a geospatial dataset containing land parcel information. This dataset contains georeferenced data provided by The Nature Conservancy (TNC) Brazil for two municipalities (Luís Eduardo Magalhães and São Desidério). In addition, we complemented the information provided by TNC with georeferenced data from SICAR and from the Brazilian National Institute for Colonization and Agrarian Reform (INCRA). However, not all rural properties were registered in these three datasets, resulting in a total of 89% of rural properties coverage in the study site.

We categorized land parcels into four classes: micro, small, medium, and large properties. These size categories follow the official guidelines determined by INCRA, classified by a unit measure called fiscal module. The fiscal module for all four municipalities present in our study site is 65 hectares, and the categories are established as follows: a) micro: less than 65 hectares, b) small: between 65 and up to 260 hectares, c) medium: between 260 and 975 hectares, and d) large: larger than 975 hectares (**Figure B.1**).

3.2.3 Gap Analysis

3.2.3.1 Habitat Suitability Index Models

Wildlife habitat suitability models are crucial in conservation planning to evaluate biodiversity loss, habitat degradation, and conservation status at a scale compatible with management actions (Scott et al., 1993). In terms of conservation assessment, birds are important environmental indicators and are the best-known group of organisms (García-Moreno et al., 2007). The Cerrado contains about 48% of Brazil’s avifauna, representing around 840 species with approximately 40 endemic to this biome (Lopes, 2009; Jose´ Maria Cardoso Da Silva, 1997). To assess the protection status of essential avian habitats, we developed habitat suitability models for 9 bird species that were either endemic or endangered inhabitants of the study site (**Table 3.1**). The birds were selected based on a combination of factors: a) the endangered species list used for selecting Priority Areas for Cerrado Conservation (WWF-Brazil, 2015); b) the endemic species lists developed by Silva (1997), Lopes (2009), and Marini et al., (2009) for Neotropical savannas; c) species occurrence in the study site, confirmed by registry at the Wikiaves website (www.wikiaves.com.br, last accessed on 7/2019); and d) habitat information availability. In addition, we included *Lepidocolaptes wagleri*, a species recently described by Silva and Straube (1996) and documented as occurring in our study site. The species is considered endangered and endemic, with its distribution restricted to the left bank of the São Francisco River. Moreover, three of these species are classified as vulnerable to climate and land-use changes (Borges et al., 2019).

Species-specific habitat suitability maps were derived in ArcGIS 10.5 (ESRI, 2017) from land cover and topographic data, resulting in 4 suitability levels: 1 – primary habitat for both breeding and foraging; 0.5 – suitable habitat for either breeding or foraging; 0.2 marginal habitat required for dispersal; 0 – unsuitable (**Table 3.1**, for more details see **Table B.1**). Species habitat requirements were determined by an extensive literature review and opinions assessed through personal communications with a group of ornithologists and local birdwatchers. Home range information was derived from the literature or through information provided by Kennedy et al., (2016b) from The Nature Conservancy's Dow Project (**Table B.1**).

Table 3.1. List of species used for developing habitat suitability models and their major habitat requirements found in the literature and through personal communications with ornithologists.

Species	Conservation Status (IUCN) / Endemism	Primary habitat	Suitable habitat	Marginal habitat
Anodorhynchus hyacinthinus	Vulnerable / No	Palm swamp	Escarpments, riparian forest, open savanna, shrubby grassland	Marsh, shrub swamp
Lepidocolaptes Wagleri	Endangered / Yes	Tropical dry forest, semi-deciduous forest	Upland riparian forest	Woodland
Euscarthmus rufomarginatus	Near Threatened / Yes	Shrubby grassland, open savanna	Savanna	
Cypsnagra hirundinacea	Least Concern / Yes	Shrubby grassland, open savanna	Savanna	
Melanopareia torquata	Least Concern / Yes	Shrubby grassland, open savanna	Savanna	Marsh
Neothraupis fasciata	Least Concern / Yes	Savanna, open savanna	Shrubby grassland, cerrado woodland	
Herpsilochmus longirostris	Least Concern / Yes	Riparian forest	Semi-deciduous forests, savanna, woodland	Wetlands
Myiothlypis leucophrys	Least Concern / Yes	Wet riparian forest	Upland riparian forest	Wetlands; Semi-deciduous forests
Syndactyla dimidiata	Least Concern / Yes	Wet riparian forest	Upland riparian forest, semi-deciduous forests, palm and shrub swamps	Savanna, woodland, marsh

3.2.3.2 Forest Code Analysis

We consolidated a Forest Code dataset containing information of Legal Reserves and Areas of Permanent Preservation. The latter aims to conserve water resources and soil stability through buffer zones surrounding these environmentally sensitive areas (Soares-Filho et al 2014). Areas of Permanent Preservation present in our study area include buffer zones along streams and rivers, springs, hydromorphic soils (used as a proxy for wetlands/*veredas*), and plateau borders. We calculated buffer zones in ArcGIS 10.5 using the LSIE product, following Forest Code guidelines (**Table 3.2**).

Table 3.2. Forest Code guidelines on buffer widths regarding each category of Areas of Permanent Preservation

Areas of Permanent Preservation	Buffer width (m)
Rivers and streams	
Up to 10 meters	30
10 ~ 50 meters	50
Hydromorphic soil (*Veredas*)	50
Along plateau borders	100
Dams	15*
Water springs	50

*15 m is the minimum buffer size for dams with an area larger than 20ha and located in rural landscapes, but size varies with their license type and time of implementation.

All overlaps between Areas of Permanent Preservation and Legal Reserves were assigned to Areas of Permanent Preservation. Then, we performed a spatial intersect between our consolidated Forest Code dataset and the LULC map to estimate the following biodiversity gaps: a) area (ha) covered by physiognomic types within Areas of Permanent Preservation and Legal Reserves, b) protection status (i.e. fraction of a given class under Forest Code protection relative to the total area occupied by that class in the landscape), and c) fraction of a given class left unprotected by the Forest Code policy. A second spatial intersect was performed between our Forest Code dataset and the bird habitat suitability maps to estimate the same biodiversity gaps described for physiognomic types.

In order to evaluate policy compliance status by rural property size, we extracted LULC information within Areas of Permanent Preservation and Legal Reserves for each individual land parcel that was completely within the study area boundary, excluding boundary properties. Rural properties were then classified into 4 categories: full compliance, noncompliant, partial compliance, and LR not reported. The first category corresponds to properties in compliance with both Areas of Permanent Preservation and Legal Reserves (or only with Legal Reserves, in case the property does not have Areas of Permanent

Preservation within their land); the second category corresponds to properties noncompliant to both policy instruments (or properties that do not have Areas of Permanent Preservation but hold a fraction of illegal land conversion within their Legal Reserve boundary); the third category corresponds to properties in compliance with either policy instrument while holding a fraction of illegal land conversion in the other category (e.g., properties compliant to Legal Reserve but having some converted land within Areas of Permanent Preservation or vice-versa); and the last category relates to properties that had not reported their Legal Reserve at the outset of our study. This category may include properties that are not in compliance with the regulation but may also include cases where the Legal Reserve is allocated in another property (e.g. biodiversity offsetting), which is permitted by legislation. For the noncompliant and partial compliance categories, we considered a minimum area of 0.1 ha of illegal land conversion to avoid issues regarding misregistration of property boundaries.

3.2.3.3 Spatial planning analysis to allocate Legal Reserves

Specific guidelines on how and where to allocate Legal Reserves are not currently provided by the Forest Code policy. The state of Bahia, however, recommends allocating these conservation set asides near Areas of Permanent Preservation to improve landscape connectivity in these areas. Nevertheless, Legal Reserve allocation can be arbitrary and may not be guided by biodiversity conservation needs. We used the *Zonation* spatial conservation prioritization framework and supporting software (Moilanen et al., 2005) to simulate allocating 20% of each land property as mandatory set asides based on complementarity principles to increase biodiversity representation across the landscape. Our specific objective was to evaluate the effectiveness of current conservation set asides in terms of biodiversity representation. For this, we compared current set aside efforts to our simulated solution and made land use policy recommendations for future guideline implementation.

Zonation uses a reverse stepwise removal heuristic and marginal loss function to produce a spatially explicit hierarchical prioritization of a given landscape. Sites are prioritized based on conservation value (i.e. biological value of each grid cell) and the proportional representation of biodiversity features across the study area. *Zonation* also allows for using administrative units to provide a balanced representation of biodiversity features across the entire landscape while considering local priorities through its "weak local administrative units" mode (Moilanen and Arponen, 2011). Given the conceptual nature of Legal Reserves to be allocated within individual land parcels, we used a rural properties layer as administrative units to ensure prioritization of physiognomic types within each land parcel. This mode accounts for the abundance/rarity of physiognomic types at both landscape and rural property levels, thus avoiding the prioritization of a specific feature that is locally rare but globally abundant. Because our rural properties dataset does not cover the entire study site, we filled the gaps by creating polygons as artificial properties. Areas of Permanent Preservation were masked out of our analysis to avoid selecting sites already protected by this policy category.

The marginal loss was calculated by the *Zonation* Additive Benefit Function (ABF) algorithm, which seeks to maximize the proportional (weighted) representation of biodiversity features (i.e. physiognomic types, in our case) while favoring sites carrying high diversity of physiognomic types (Lehtomäki and Moilanen, 2013; Moilanen, 2007). In this analysis, each land cover category from our map was entered as a separate biodiversity feature, and their weights (priorities) were set (**Table B.2**) as: a) 4x weight for physiognomic types least protected by Areas of Permanent Preservation, b) 2x weight for physiognomic types holding higher number of endemic plants, and c) 1x weight for the remainder of physiognomic types. To avoid selecting sites in areas featuring human activities, land-use classes were given negative weights as recommended by Moilanen and Arponen (2011). In order to facilitate planning on a scale compatible with that used by decision-makers in Brazil, we aggregated the original 5-meter cell of each biodiversity feature to 30 meters by using the sum function, which retains the proportion of each class in a given cell.

3.3 Results

3.3.1 Gap Analysis

3.3.1.1 Forest Code Analysis

Forest Code sites constitute a total area of 225,089 ha representing 24% of the study area, of which 10% are under Areas of Permanent Preservation and 14% under Legal Reserves. These sites under policy regulation comprise vegetation (86% of total Forest Code area), converted areas (12%), water (1%), and “other” (i.e. shade and clouds: 0.3%). 40% of Forest Code sites are allocated as Areas of Permanent Preservation (89,467 ha, of which 84% corresponds to natural vegetation) and 60% as Legal Reserves (135,622 ha, of which 88% corresponds to natural vegetation). The Forest Code policy protects more than 30% of all bird species habitats considered (**Table 3.3**) and of most physiognomic types **(Figure 3.2**) in our study area, excepting semi-deciduous forest and shrubby grassland, of which 18% and 24% are under land use policy regulation.

Our results show that Areas of Permanent Preservation and Legal Reserves differ in terms of their protection contribution and biodiversity representation. Regarding the protection status of physiognomic types (**Figure 3.2**), Areas of Permanent Preservation protect 100% of wetlands (i.e. marsh, palm swamp, shrub swamp), 80% of riparian forests, and 70% of tropical dry forest in our study area. Savannas, grasslands, and semi-deciduous forest are, on the other hand, mostly protected by Legal Reserves and have less than 40% of their area under policy protection. In terms of representation (i.e. fraction occupied by each physiognomic type within Forest Code sites), “savanna” and “open savanna” have the largest area (ha) under policy protection (**Figure 3.2**) and are the most represented physiognomic types within Areas of Permanent Preservation (**Table B.3**).

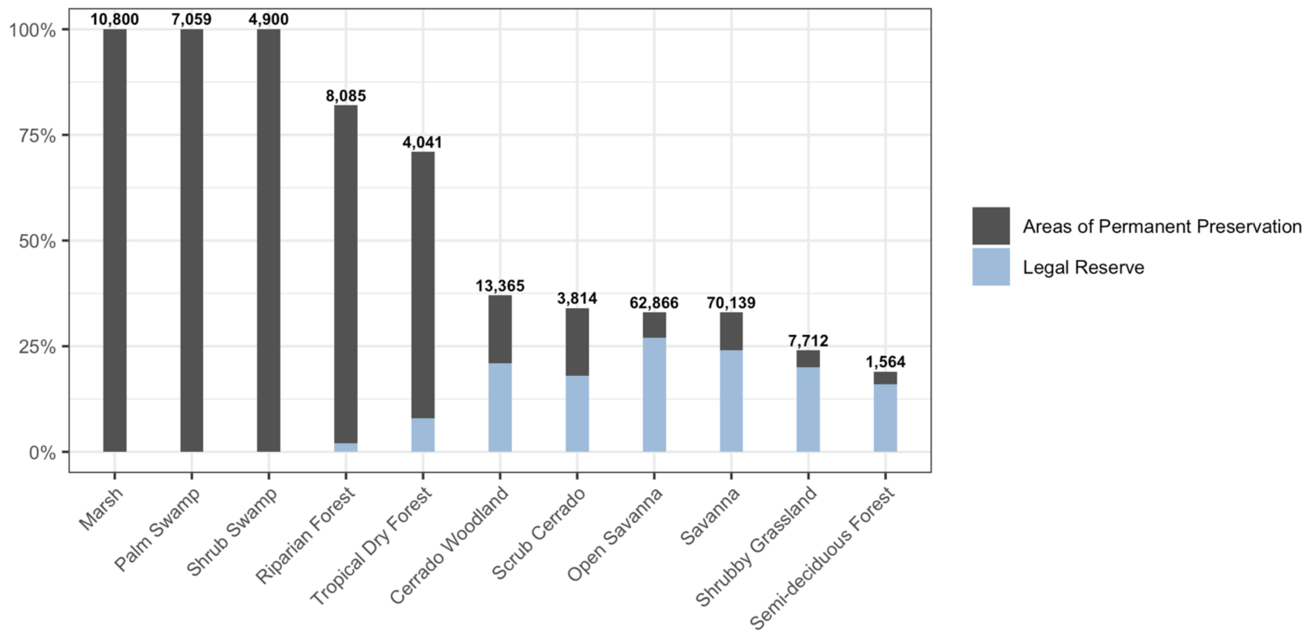

Figure 3.2. Proportion of each physiognomic type protected by the Forest Code. The numbers in bold at the top of each bar represent the total area (in hectares) protected for each class.

We have estimated how physiognomic types and their conservation status are distributed within different categories of Areas of Permanent Preservation (APP) and Legal Reserves (**Figure 3.3**). In terms of conservation status, the hydromorphic soil APP protects all wetland types (i.e. marsh, shrub swamp, palm swamp) and ~50% of riparian forest. The Plateau APP was responsible for ensuring major protection of savannas and grasslands (46%), tropical dry forest (77%), and semi-deciduous forest (48%). The category of APP within 30m streams was also important for increasing protection status of all physiognomic types, especially riparian forest, semi-deciduous forest, savannas, and grasslands. The APPs within dam and perennial spring buffers contributed less than 1% to the total protection status of physiognomic types for our study area (**Figure 3.3, Table B.3**).

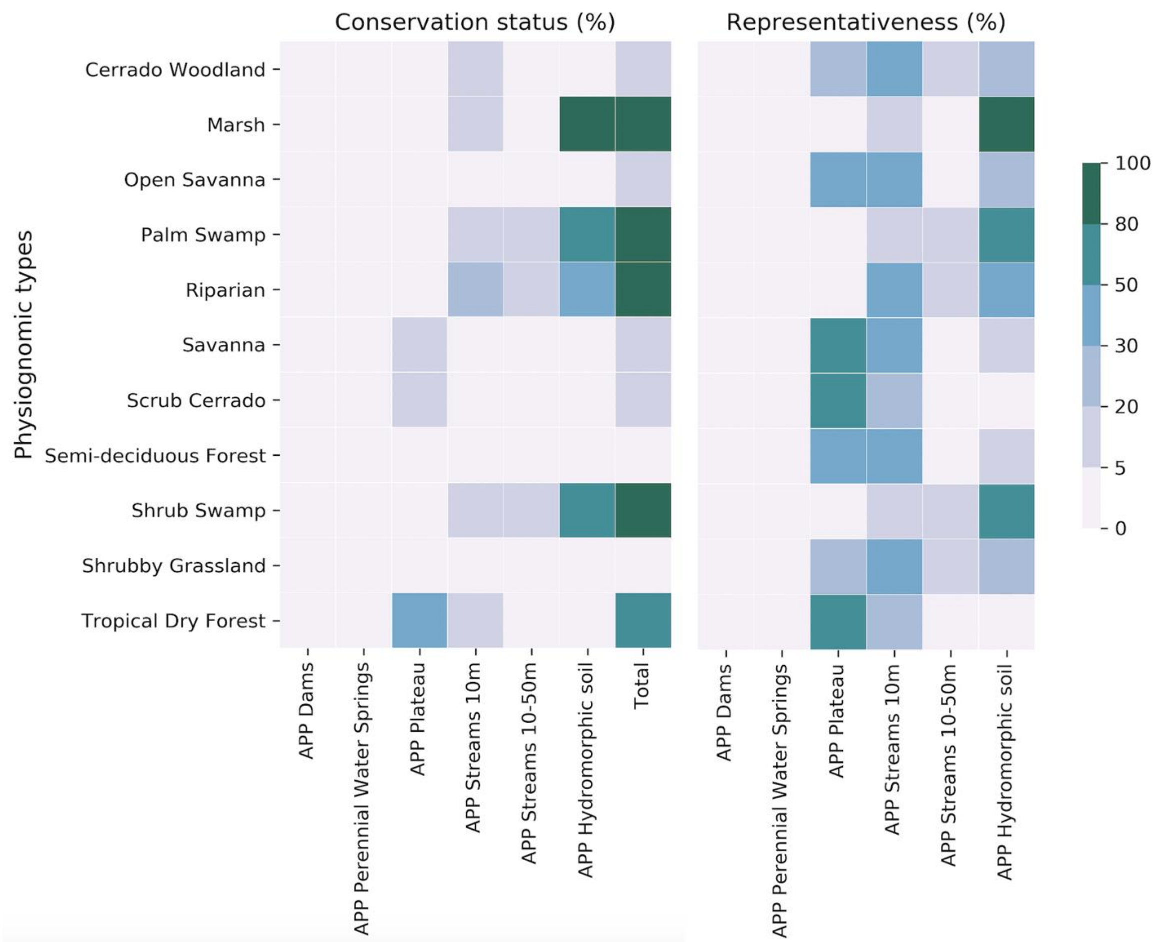

Figure 3.3. Graph showing estimates of the conservation status of physiognomic types and their respective representativeness by categories of Areas of Permanent Preservation (APP).

As part of the gap analysis we estimated the fraction of physiognomic types not protected by current policy efforts (**Figure 3.4**). Semi-deciduous forest has the highest percentage of non-protected area (> 80%), followed by shrubby grassland (> 70%) and all physiognomic types within savanna ecosystems (> 60%). In terms of areal coverage, the savanna and open savanna physiognomic types hold the largest areas (140,040 and 127,989 hectares, respectively) left unprotected by the Forest Code policy.

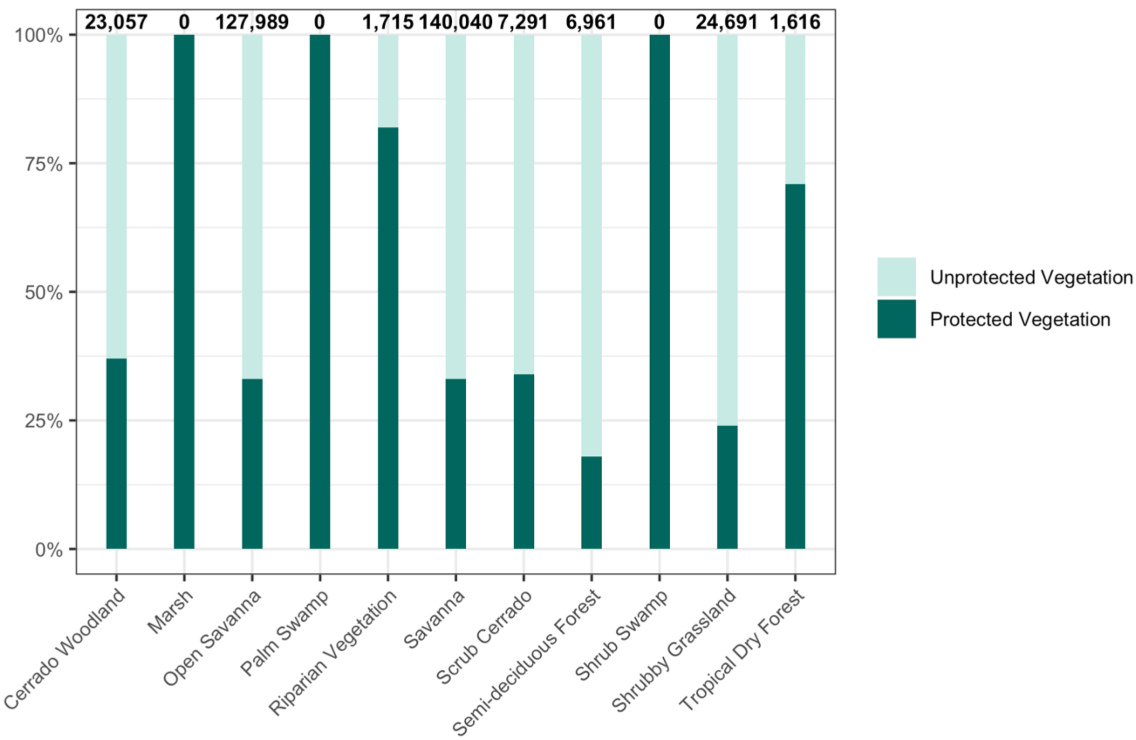

Figure 3.4. Proportion of physiognomic types that are not protected by the Forest Code policy, relative to their distribution across the landscape (reported in percentage). The numbers in bold represent the total area (in hectares) of each physiognomic type not within Forest Code areas.

For bird species habitats (**Table 3.3**), Legal Reserves protect a larger fraction of habitats for *Anodorhynchus hyacinthinus, Cypsnagra hirundinacea, Euscarthmus rufomarginatus, Melanopareia torquata*, and *Neothraupis fasciata*; whereas Areas of Permanent Preservation protect a larger habitat fraction for the remaining species. All species mostly protected by Legal Reserves have more than 60% of their habitats left unprotected by policy regulation. Species mostly protected by Areas of Permanent Preservation have more than 75% of their habitats under protection (< 25 % unprotected), except for *Lepidocolaptes wagleri*, which has ~60% of its habitat unprotected by the Forest Code. Given that savanna ecosystems have lower protection status (**Figure 3.2**), savanna-dependent species (i.e. *Neothraupis fasciata*, *Euscarthmus rufomarginatus*, among others) have lower protection rates compared with forest species. The exception is for *Lepidocolaptes wagleri*, which is an endemic and endangered species of restricted distribution inhabiting tropical dry forests with non-protected status similar to savanna species.

The Forest Code protects a minimum of 30% of all species primary habitats available in the landscape, while 3 species have 100% of their primary habitats under Forest Code protection. Regarding policy instrument categories (i.e. different categories of Areas of Permanent Preservation and Legal Reserves), the Hydromorphic Soil APP category and Legal Reserves protect the largest fraction of 8 species primary habitats, while the Plateau APP category protects the majority of primary habitat for *Lepidocolaptes wagleri* (further details provided in **Table B.4**).

Table 3.3. Protection status (%) of endemic/endangered bird habitats by the Forest Code (FC), the proportion (%) of habitat within Areas of Permanent Preservation (APPs) and Legal Reserves (LRs), the total amount of habitat (in hectares) available in the landscape for each species, the proportion (%) of habitat left unprotected by the Forest Code, and the proportion (%) of total primary habitat protected; category in parenthesis corresponds to the Forest Code category with the largest fraction of primary habitat.

	APPs - % (ha)	**LRs - % (ha)**	**FC Total Protection - % (ha)**	**Total habitat amount (ha)**	**Unprotected habitat - %**	**Total protection status of primary habitats - % (category)**
Anodorhynchus hyacinthinus	16.9 (42,687.7)	22.6 (57,249.5)	39.5 (99,937.2)	252,854.9	60.5	100 (APP – Hydromorphic Soil)
Cypsnagra hirundinacea	7.7 (33,269.9)	24.8 (107,447.1)	32.5 (140,717.0)	433,436.7	67.7	31.6 (Legal Reserve)
Euscarthmus rufomarginatus	7.7 (33,269.9)	24.8 (107,447.1)	32.5 (140,717.0)	433,436.7	67.5	31.7 (Legal Reserve)
Herpsilochmus longirostris	74.9 (19,037.5)	3.7 (933.2)	78.6 (19,970.7)	25,401.0	21.4	85.1 (APP – Hydromorphic Soil)
Lepidocolaptes Wagleri	26.9 (4,767.3)	13.8 (2,441.1)	40.6 (7,208.4)	17,734.3	59.4	36.6 (APP – Plateau)
Melanopareia torquata	9.9 (44,057.1)	24.2 (107,459.5)	34.1 (151,516.6)	444,265.6	65.9	31.7 (Legal Reserve)
Myiothlypis leucophrys	87.8 (15,539.5)	1.5 (256.7)	89.3 (15,796.2)	17,690.5	10.7	100 (APP – Hydromorphic Soil)

	APPs - % (ha)	LRs - % (ha)	FC Total Protection - % (ha)	Total habitat amount (ha)	Unprotected habitat - %	Total protection status of primary habitats - % (category)
Neothraupis fasciata	8.3 (38,932.0)	24.5 (115,149.7)	32.8 (154,081.6)	469,858.0	67.2	33.2 (Legal Reserve)
Syndactyla dimidiata	84.0 (25,380.2)	3.8 (1,145.1)	87.8 (26,525.3)	30,227.3	12.2	100 (APP – Hydromorphic Soil)

We identified some potentially illegal anthropogenic activities and/or degraded land within areas under Forest Code policy. Our results show that 27,361 hectares may be illegally converted within our study area. Separately analyzing both policy instruments in terms of their land cover representation (**Figure 3.5**), Legal Reserves hold a larger proportion (53.2%) of natural vegetation compared to Areas of Permanent Preservation (33.5%), but they also carry a slightly larger area of converted land (7.3%) compared to the latter (~5%).

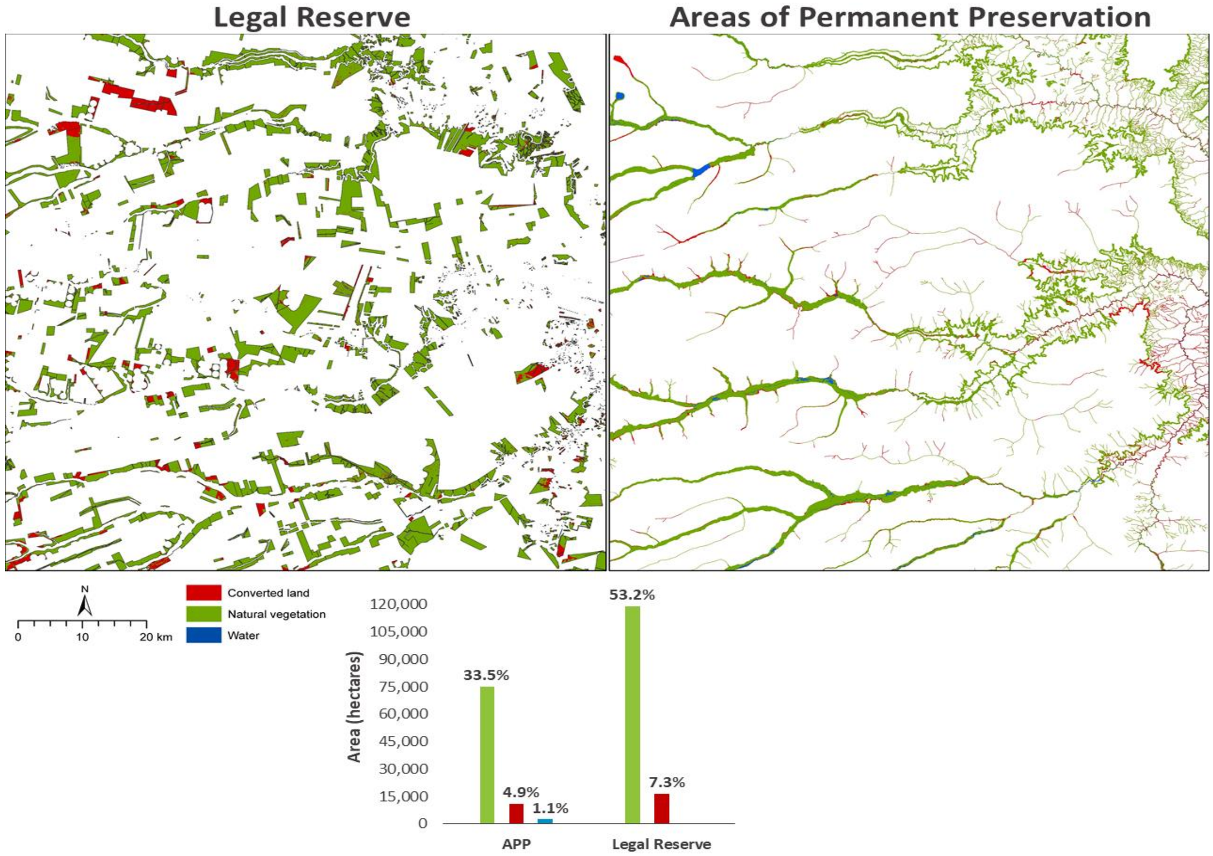

Figure 3.5. Maps showing converted areas, natural vegetation, and water within Legal Reserves and Areas of Permanent Preservation. The graph reports the area (ha) covered by these classes within Areas of Permanent Preservation and Legal Reserves and, in bold, the proportion of each class relative to the total area under Forest Code protection.

3.3.1.2 Compliance status by property size

A total of 4,811 properties fell completely within our study area, spanning an area of 734,025 ha (78% of the total study area). Most land parcels corresponded to micro-properties (78% of the total number of properties), followed by the medium (11%), small (7%), and large (4%) property categories. In terms of areal coverage, large and medium properties occupy ~90% of the area analyzed (further details on property size provided in **Table B.5**). Regarding compliance status, we estimate a total of 3,163 properties (66% of all properties analyzed) that potentially were not in compliance with the Forest Code policy.

These fell into 3 categories: noncompliant (1,816 properties), partially compliant (884 properties), and Legal Reserves not reported (463 properties).

Figure 3.6 shows the spatial distribution of compliance categories of rural properties. We estimate that 38% of all properties are noncompliant to both Areas of Permanent Preservation and Legal Reserves, featuring 14,231 ha of illegal land conversion regarding the Forest Code policy. 18% of all properties (5,578 ha illegally converted) were found to be in partial compliance with the Forest Code. An additional 463 properties (10% of all properties analyzed) did not report their Legal Reserves, and thus, are also not in compliance with current policy. A total of 1,648 properties (34% of all analyzed properties) are in full compliance with the Forest Code policy in our study area, spanning 3,376 ha of protected natural vegetation.

Regarding compliance status by property size, micro-properties (<65 ha) had 1,810 parcels within all three noncompliant categories, representing the category with the largest number of noncompliant properties (**Figure 3.6-A**). In addition to compliance status by number of properties, we also estimated the fraction of compliance status for the categories "noncompliant" and "fully compliant" by property size (**Figure 3.6-B**). Our results indicate that medium and large properties held the largest area of illegal land conversion (12,146 ha or 85% of the total illegal converted areas analyzed). By contrast, micro-properties only held 10% (or 1,294 ha) of the total illegally converted land within areas under Forest Code policy in our study region.

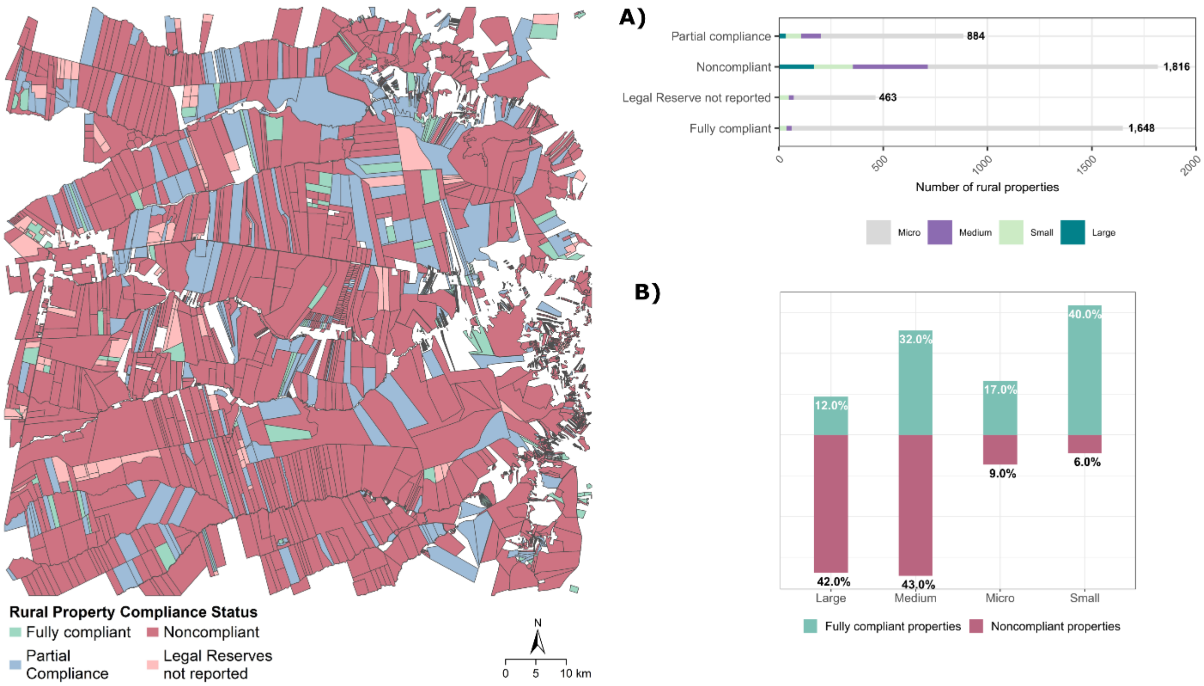

Figure 3.6. Map showing the compliance status of the analyzed rural properties with the Forest Code. A) Bar graph showing the number of properties compliant to the Forest Code by farm size category. Numbers in bold represent the total number of properties within each compliance category. B) Bar graph showing the fraction of compliance status (only for fully compliant and noncompliant categories) by property size.

3.3.1.3 Guideline proposal for allocating Legal Reserves

The *Zonation* analysis (**Figure 3.7**) resulted in 173,939 hectares of simulated Legal Reserves across the landscape, representing the 20% required by the Forest Code to be allocated in each land property (relative to the total area unprotected by Areas of Permanent Preservation).

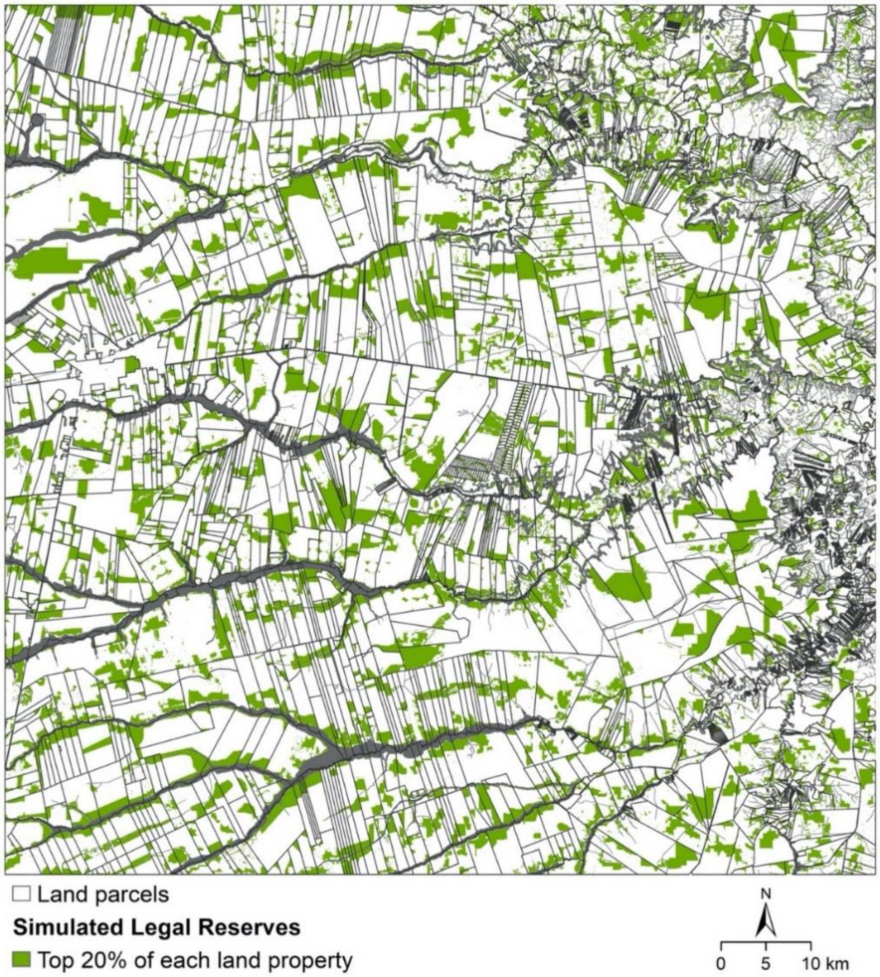

Figure 3.7. Map showing the hierarchical prioritization resulting from analysis in *Zonation*. The top 20% is equivalent to the most valuable areas considering 20% of the landscape, which is the proportion of the area for Legal Reserve allocation.

These simulated set-aside areas have a 40% overlap with the actual Legal Reserves and resulted in a higher protection status and diversity of physiognomic types that are not protected by Areas of Permanent Preservation (**Figure 3.8**). Thus, our *Zonation* solution helped to maximize protection of physiognomic types while accounting for complementarity between policy instruments. For instance, semi-deciduous forest and all physiognomic types within savanna ecosystems, which are underrepresented in Areas of Permanent Preservation were allocated as Legal Reserves in our solution. In addition, results show a substantial increase in protection status of physiognomic types that are underrepresented in actual Legal Reserves, such as Semi-deciduous Forest (52.4%), Shrubby Grassland (50.7%), Scrub Cerrado (76.8%), Cerrado Woodland (65.5%), and Tropical Dry Forest (98.1%). However, our solution also featured areas under anthropogenic land-use, such as farming and urban areas, which had their proportion increased to 8.1% and 9.8%, respectively. Following regulation decree, these areas should be either restored or compensated through biodiversity offsetting strategy (i.e. Environmental Reserve Quotas – CRA in Portuguese acronym).

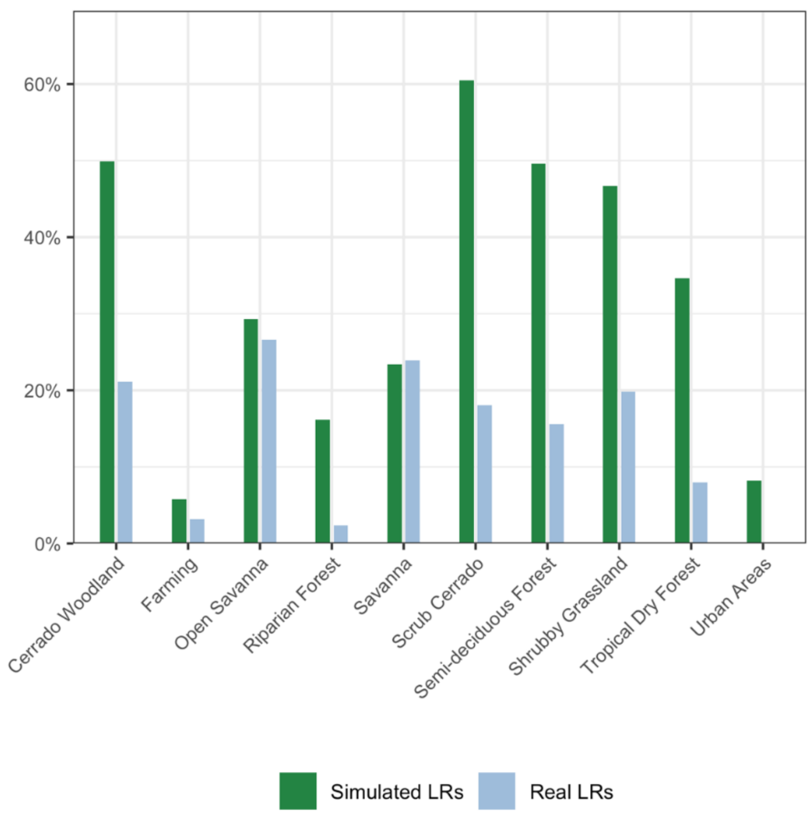

Figure 3.8. Graph featuring the protection status of physiognomic types by simulated Legal Reserves from *Zonation* output, compared to the actual Legal Reserves.

3.4 Discussion

Our approach to estimating biodiversity conservation gaps at fine scale demonstrates that the Forest Code policy has a critical role in ensuring protection of Cerrado biodiversity, being particularly important in productive landscapes that lack public protected areas. Results from our study show that land-use regulation protects a wide diversity of physiognomic types and essential habitats for several endemic and endangered bird species. In our study site, at least 18% of each physiognomic type is protected by land-use regulation, resulting in a total of 194,488 ha protected in one of the most productive regions for commodity crops in the Cerrado. This policy protects more than 30% of primary habitats for all bird species in our list, reaching 100% protection for *Anodohynchus hyacinthinus, Myiothlypis leucophrys,* and *Syndactyla dimidiata*. However, it is important to note that all protection status estimated in our study are based on the current distribution of biodiversity features and not on their original extents.

The gap analysis showed that both policy instruments (i.e. Areas of Permanent Preservation and Legal Reserves) are important and act as complementary efforts to protect Cerrado biodiversity and natural resources. Our results indicate that Areas of Permanent Preservation are critical for protecting wetlands, riparian vegetation, and tropical dry forest (**Figure 3.2**). Our study region features a predominantly parallel (east-west) drainage pattern with narrow strips of riparian vegetation following streams and rivers up to 50m, and surrounding wetlands (e.g., *veredas*). Thus, the buffer zone of 30m and 50m for rivers and hydromorphic soils determined by the Forest Code was enough to ensure protection of 100% of wetlands and 80% of riparian forest, particularly in areas located on the plateau. However, these protection estimates may be much lower in other Cerrado regions having broader riparian zones, especially in transition zones to the Amazon, Pantanal, and Atlantic Forest. Tropical dry forest is effectively protected by Areas of Permanent Preservation (63.5%) in our study area due to its natural occurrence on steep slopes located on interfluves. This estimate might be similar to other tropical dry forest areas of similar topography but may be lower for areas of different topographical characteristics (e.g., flat terrains such as in areas along the São Francisco River in Northern Minas Gerais state).

In contrast, Legal Reserves are essential for protecting physiognomic types of the savanna ecosystem, as well as shrubby grasslands, and semi-deciduous forest (**Figure 3.2**). However, their relative protection status is lower (< 40% protected) compared to wetlands and riparian forests. Possible reasons for this include: a) wide distribution of savannas and open savannas physiognomies in the study site, which consequently results in a lower relative proportion protected; b) several Legal Reserves in our study area are located near Areas of Permanent Preservation due to state recommendations for increasing structural connectivity. Areas surrounding Areas of Permanent Preservation, however, do not feature large concentrations of semi-deciduous forest, cerrado woodland, and scrub cerrado due to soil and geomorphological characteristics of this study site; c) landholders might be avoiding placing Legal Reserves in areas of flat topography and nutrient-rich soils, such as in semi-deciduous forest, which mostly conflicts with areas suitable for commodity-driven agricultural practices. Moreover, the small percentage requirement for set asides allocation in the Cerrado (i.e. 20% compared to its 80% requirement in the Amazon) is another main concern. Our results indicate that, for our study area, the 20% mandatory fraction might be inadequate for guaranteeing protection of physiognomic types mainly dependent on this policy mechanism.

Considering the entire Cerrado biome extent (~203Mha), it is estimated that ~45Mha (~22%) of land corresponds to Legal Reserves, whereas all Cerrado Protected Areas (PA) hold 16Mha (Metzger et al., 2019). The Cerrado PA network is considerably smaller compared to the 116Mha of PAs located in the Amazon, which makes the Forest Code an essential policy for maintenance and protection of Cerrado ecosystems (Metzger et al., 2019). Our results quantitatively corroborate analysis by Metzger et al. (2019) showing that Legal Reserves have unique functions in complementing Areas of Permanent Preservation by securing protection of upland vegetation types in these areas, such as savannas, open savannas, grasslands, woodlands, and semi-deciduous forests. Thus, both policy instruments play important and complementary conservation roles, contributing to conserving distinct vegetation community types, particularly in regions under intense agricultural occupation lacking protected areas.

Results of the *Zonation* analysis suggest that set-asides allocation could be improved if the Forest Code specified allocation guidelines considering the diversity of physiognomic types and their representativeness across the entire landscape, while ensuring complementary protection with other policy efforts. Legal Reserves would benefit from accounting for structural connectivity of patches between land parcels to improve connectivity flows in the landscape. Results from our simulated analysis showed an increase in protection for most underrepresented physiognomic types in Areas of Permanent Preservation compared to the actual Legal Reserves in our study area. Moreover, this analysis is in accordance with conclusions from Kennedy et al. (2016b) that set-asides allocation strategies accounting for representativeness of vegetation types across the whole landscape can be more efficient as opposed to restricting the analysis to individual land parcels. Improving guidelines for Legal Reserve allocation in such manner would require cooperation and coordination across landowners at a landscape-to-regional scale, which could be implemented with the support of agricultural associations and local public institutions.

Despite the demonstrated importance of the Forest Code policy in protecting Cerrado biodiversity, the biome hosts the largest amount of unprotected land (~44Mha across the entire biome) in the country, which can be legally converted to human activities (Metzger et al., 2019). In our study area, this estimate is equal to an area of 333,359 ha (35% of the region) that can be potentially cleared at any time. These unprotected areas support more than 50% of the important habitats for many species, including a vulnerable bird species (*Anodorhynchus hyacinthinus*), an endangered bird species (*Lepidocolaptes Wagleri*), and many endemic plant and animal species. Additional conservation policies and management actions should be considered in these agricultural landscapes. In principle, agricultural production could benefit by ensuring persistence of pollinators and seed dispersal species such as *Cypsnagra hirundinacea*, which is a potential seed disperser (Bagno and Marinho-Filho, 2001) with ~68% of current habitat unprotected.

We identified 27,361 ha of illegal converted land within areas under Forest Code protection, which according to the legislation, must either be prioritized for restoration or compensated through biodiversity offsetting programs (i.e. Environmental Reserve Quota – CRA, acronym in Portuguese). The legislation requires restoration efforts and/or compensation strategies from farms carrying illegal land clearing, but priorities and guidelines vary for areas converted before July 22, 2008. Given that identifying these areas is not the focus of this work, we cannot affirm whether the estimated total area of illegal land conversion in this study region should be restored to natural vegetation. However, joint efforts among monitoring programs such as TerraBrasilis and SICAR are crucial for identifying such areas and setting restoration priorities.

The presence of human activities (i.e. settlements, roads, agriculture) within Forest Code sites implies that some rural properties are not compliant to the legislation. However, full compliance status with current policy often involves high restoration costs, conflicts with agricultural production (although these areas represent less than 1% of total croplands; see Soares-Filho et al., 2014), and compliance enforcement that is often lacking (Azevedo et al., 2017). For our study site, micro properties are the category with the largest number of rural properties potentially non-compliant to both Forest Code policy instruments. However, in terms of areal coverage, large and medium properties are the categories holding the largest portion of illegal land conversion within Forest Code sites. Studies in the state of Mato Grosso (located in the Cerrado transition to the Amazon) show similar patterns of small (< 150 ha) properties having the lowest compliance rates with the Forest Code in terms of number of noncompliant properties (Michalski et al., 2010; Zimbres et al., 2018), whereas large properties in Mato Grosso and Pará States featured the highest deforestation rates compared to the other farm size categories, especially small landholders (L'Roe et al., 2016; Michalski et al., 2010; Richards and VanWey, 2016). Michalski et al. (2010) showed that farms in Mato Grosso State featured higher compliance rates with streams APPs as opposed to other categories of Areas of Permanent Preservation or Legal Reserves, which resulted in higher protection of riparian forest compared to upland forests. In our study site, farms have higher compliance rates with Legal Reserves (1,636 farms) compared to Areas of Permanent Preservation (404 farms). A study in the Atlantic Forest demonstrated that properties non-compliant with Areas of Permanent Preservation are located in areas with lower socioeconomic conditions, indicating a need for improving both environmental and social policies (Rezende et al., 2018). Compliance patterns for the Cerrado within the state of Mato Grosso do Sul were similar to ours and to those reported for the Amazon, with a positive relationship between Legal Reserve non-compliance rates and farm size (Stefanes et al., 2018).

3.5 Conclusions

Fine-scale gap analysis is critical to provide quantitative information regarding Forest Code protection and habitat conservation status. Given the Cerrado's weak protected areas system, the Forest Code policy acts as the main management strategy regarding conservation of native vegetation within agricultural landscapes. Our analysis indicates that Legal Reserves and Areas of Permanent Preservation are both essential for ensuring protection of a wide variety of physiognomic types and important habitats for endemics and endangered species. As verified in our results and implied by its definition, Areas of Permanent Preservation provide greater protection of riparian vegetation, wetlands (e.g., *veredas*), and tropical dry forest; whereas Legal Reserves complement Areas of Permanent Preservation by potentially protecting a larger amount of other physiognomic types, such as savannas, woodlands, grasslands, and semi-deciduous forests. In terms of policy implications, we recommend additional guidelines to improve the provision of Legal Reserves by prioritizing physiognomic types with lower protection status while considering their representativeness across the landscape. Such strategies should include further engagement with stakeholders and can be implemented at the municipality level to be in accordance with Forest Code specifications by land parcel categories. Moreover, policy enforcement efforts can be improved by targeting property size categories that have lower Forest Code compliance status.

Acknowledgments

This research was funded by the Coordenação de Aperfeiçoamento de Pessoal de Nível Superior (CAPES) Foundation, Ministry of Education of Brazil, through the Science Without Borders program. We would like to thank the ornithologists David Oren and Neiva Guedes for sharing notes, calibrating, and validating our habitat suitability maps; Miguel Marini for helping to select the set of bird species used in this study; Tulio Dornas for sharing notes and pictures about species habitats; the birdwatcher Ronaldo Francisco for sharing local pictures, notes, and confirming species occurrence in our study site; Ricardo Machado for valuable suggestions to develop a spatial planning for Legal Reserve allocation. We also thank The Nature Conservancy Brazil and Aline Leão for granting access to their geospatial database for the Western Bahia site; Sandro de Oliveira, Osmar Abílio de Carvalho Junior, and the Laboratory of Spatial Information Systems (LSIE) at the University of Brasilia for kindly sharing their geospatial database for the entire Western Bahia region.

Chapter 4. Private lands as opportunities for improving biodiversity and connectivity conservation across Cerrado protected areas

Fernanda F. Ribeiro[1], Frank W. Davis[2], Gabriel A. Daldegan[1], Fernanda T. Brum[3], Carlos A. Klink[4], Laura L. Hess[5], Kelly K. Caylor[1,2], Dar A. Roberts[1]

[1]Department of Geography, University of California, Santa Barbara, CA 93106, USA
[2]Bren School of Environmental Science and Management, University of California, Santa Barbara, CA 93106, USA
[3]Programa de Pós Graduação em Ecologia e Conservação, Universidade Federal do Paraná, Curitiba, PR 19031, Brazil
[4]Departamento de Ecologia, Universidade de Brasília, Brasília, DF 70910, Brazil
[5]Earth Research Institute, University of California, Santa Barbara, CA 93106, USA

3.0 Abstract

Protected areas are the cornerstone of biodiversity conservation; however, their effectiveness can be compromised when isolated in an inhospitable agricultural matrix. With much of the world's biodiversity located in private lands, privately protected areas are key to improving biodiversity conservation and increasing landscape connectivity between large public protected areas. Conservation strategies in private lands are particularly important in tropical biodiversity hotspots such as the Brazilian Cerrado, where much of the remaining unprotected vegetation is within private ownership. The Brazilian Forest Code is a mandatory land use regulation strategy that protects at least 20% of natural vegetation within Cerrado private lands. Yet, additional conservation efforts are needed to expand protection beyond the Forest Code regulation. Our study examined possible alternatives for allocating important areas for biodiversity and connectivity conservation in Cerrado private agricultural landscapes. This analysis aimed to improve habitat protection and increase landscape connectivity between the surrounding protected areas threatened by isolation due to commodity-driven agricultural expansion. Specifically, we analyzed how the number and location of priority sites for conservation varies depending on biodiversity target levels and attention to habitat connectivity. Our prioritization and connectivity analyses were developed using fine-scale datasets of Cerrado physiognomic types and species-specific habitat suitability models for nine endemic or endangered birds. Our results demonstrate that expanding conservation efforts to include prioritized patches is essential in order to increase habitat representation for endangered and endemic species beyond what is currently protected by current land-use policy. Comparison of biodiversity target levels showed that protecting 10%, 20%, or 30% of our unprotected landscape increases protection by an

average of 10%, 23%, or 37% of all species habitats, respectively. Our results revealed that a 10% protection target combined with areas currently protected by the Forest Code policy would be efficient for achieving high levels of biodiversity protection (i.e. 50% of the total vegetation and 45% of all species habitat could be protected) and that a 30% protection target is necessary for maintaining landscape connectivity while maximizing biodiversity representation. Moreover, our study site demonstrated to be important for maintaining structural connectivity between the surrounding protected areas at the regional scale. The local connectivity analysis showed a 75% overlap with priority sites (using the 30% target) or areas under Forest Code regulation, indicating that current and recommended conservation efforts are important for maintaining structural connectivity across the study site.

4.1 Introduction

Protected areas are the cornerstone of biodiversity conservation, deliberately established and managed to ensure long-term protection and persistence of biodiversity and ecosystem services (Margules and Pressey, 2000). Conservation planning to prioritize new protected areas is increasingly conducted over landscape-to-regional domains and generally favors large, well-connected habitat patches to avoid fragmentation effects on species and ecosystems (e.g., Kennedy et al., 2016a). However, in many regions large and well-connected habitat patches are becoming scarce due to rapid agricultural expansion and increasing global demands for food, fiber, and energy consumption (Kremen and Merenlender, 2018; Tilman et al., 2009). The effectiveness of protected areas can be compromised when they are isolated in an inhospitable agricultural matrix, especially given ongoing climate change impacts (Foley et al., 2005; Haddad et al., 2015; Hansen and DeFries, 2007, Hannah et al. 2008). In such settings, conservation and land management efforts also need to include critical local habitats, movement corridors, and stepping stones in designing conservation strategies (DeFries et al., 2007; Kremen and Merenlender, 2018).

With much of the world's biodiversity located in private lands, improving conservation actions through privately-owned protected areas is a globally important strategy to complement current protected area networks, ensure landscape connectivity, provide essential ecosystem services, mitigate climate change, and meet international agreements such as the Aichi Target 11 and the post-2020 biodiversity framework from the Convention on Biological Diversity – CBD (Bingham et al., 2017; Borrini-Feyerabend et al., 2013; Cortés Capano et al., 2019; Kamal et al., 2015; Knight, 1999; Stolton et al., 2014). Tropical biodiversity hotspots might greatly benefit from conservation strategies on private lands given their high levels of biodiversity and the on-going threat imposed by agricultural expansion. Agricultural landscapes, in particular, are an essential component in tropical conservation strategies and have recently gained attention in mega diverse countries such as Brazil (Klink, 2019; Magioli et al., 2016). Here we examine possible alternative strategies for allocating important areas for biodiversity and connectivity conservation in a biologically rich region undergoing agricultural expansion in the Brazilian Cerrado. This Neotropical savanna is one of the most active agricultural frontiers in the world and plays an important role in maintaining Brazil as a global leader of commodity exports such as soybean, cotton, and beef. At the same time, the Cerrado biome supports exceptionally high plant (> 12,000 species, including 4,800 endemics) and vertebrate diversity (Mendonça et al., 2008; Ratter et al., 1997). At present, less than 20% of its native vegetation is undisturbed and only 8% is under protection (Françoso et al., 2015; Strassburg et al., 2017). Private lands are necessarily the focus of additional conservation efforts, comprising ~44Mha of unprotected native ecosystems (Metzger et al., 2019; Klink, 2019). Cerrado agricultural landscapes hold high biodiversity levels and support vital ecological functions, and therefore, should be incorporated in future conservation strategies (Magioli et al., 2016).

Protected areas under private governance, also known as privately protected areas (PPAs), represent an opportunity to implement actions targeting biodiversity and landscape connectivity conservation outside the public domain (Bingham et al., 2017; Gallo et al., 2009; Hilty and Merenlender, 2003). PPAs can also provide economic returns to landowners through payments for ecosystem services (Kareiva, 2010; Swift et al., 2004), tourism (Buckley, 2009; Pegas and Castley, 2014), and climate change mitigation (Heller and Zavaleta, 2009; Jantz et al., 2014). The implementation of conservation strategies in Cerrado private lands can be either mandatory, through the Forest Code legislation, or voluntary, through the implementation of private natural reserves (e.g. Private Reserves of Natural Heritage, or RPPNs – *Reserva Particular do Patrimônio Natural*).

The Forest Code is a land use regulation policy that requires all rural properties to set aside a percentage of their lands into Legal Reserves. In the Amazon region this percentage is 80% in forest lands and 35% in non-forest lands, while in other biomes, including the Cerrado, it is 20%. This policy also restricts land use within Areas of Permanent Preservation, which are designated to conserve water resources and prevent soil erosion in environmentally important areas, such as swamps, riparian areas, hilltops, and escarpments (Soares-Filho et al., 2014). In contrast, Private Reserves of Natural Heritage are privately protected areas of sustainable use defined by Brazil's National Protected Areas System (Sistema Nacional de Unidades de Conservação da Natureza – SNUC: MMA, 2003) aimed at protecting biodiversity while allowing for sustainable activities such as scientific research, education, recreation, and tourism. These areas are voluntarily established by landowners (i.e. individuals, corporations, or institutions) in private lands and, given that land ownership is fully maintained, the conservation status of RPPNs is non-revocable and there is no governmental cost for implementation (Rambaldi et al., 2005). The Cerrado holds ~300 RPPNs spanning an area of ~170,000 hectares; the number of RPPNs is projected to increase due to recent national program incentives of payments for ecosystem services such as the Floresta+ Program (established in July 2, 2020, ordinance n.288; http://reservasprivadasdocerrado.com.br/cerrado last accessed in 08/29/2020).

Ensuring landscape connectivity between protected areas is a major concern in biodiversity conservation (Heller and Zavaleta, 2009; Saura et al., 2019), and improving connectivity of Cerrado unprotected lands is a top priority in national (Grande et al., 2020) and global conservation studies (Saura et al., 2018, 2017). Similarly to the Atlantic Forest, where a dense network of small protected patches is key to increasing matrix permeability for forest bird species (Boscolo and Metzger, 2011; Uezu and Metzger, 2011), promoting sustainable management of Cerrado unprotected lands is needed to ensure connectivity between protected areas and increase biodiversity conservation (Magioli et al., 2016; Saura et al., 2017).

Most spatial planning approaches use coarse to medium (250 to 30 meters, respectively) spatial resolution datasets to identify broad-scale conservation priorities, and such datasets may not capture small heterogeneous habitat remnants in human-dominated landscapes (Paese et al., 2010). Because they are not explicitly accounted for in the planning process, these small habitat patches in agricultural landscapes may be converted to other land uses without regulatory restrictions (Wintle et al., 2019). To help overcome this challenge, readily available fine-scale Earth Observations (EO) offer a clear opportunity to improve conservation priorities in heterogeneous and fragmented tropical regions (Nagendra et al., 2013), such as the Cerrado (Ribeiro et al., 2020).

This study investigates alternative conservation priority-setting schemes for private agricultural landscapes in the Cerrado, aiming to improve habitat protection and increase landscape connectivity between protected areas. We analyze how the number and location of priority sites for conservation varies depending on biodiversity target levels and attention to habitat connectivity. Our analysis takes advantage of new fine-scale maps of vegetation types and derived species habitat maps that allow us to account for small habitat patches that might be important for achieving representation targets and maintaining structural connectivity. Specifically, we seek to answer the following research questions: a) Assuming efficient conservation, what proportion of unprotected land is required to adequately represent habitats in privately protected areas? b) How does a greater emphasis on connectivity affect conservation land requirements? c) How could projected land conversion impact conservation priority areas?

4.2 Material and Method

4.2.1 Study Site

The Matopiba region **(Figure 4.1**) is a relatively recent commodity-driven agricultural frontier that occupies ~73Mha over the Cerrado portion of the states of Maranhão, Tocantins, Piaui, and Bahia. The region accounts for approximately 10% of the national crop production (IBGE, 2015; Miranda et al., 2014) and is responsible for almost half of forest carbon emissions in the biome (Noojipady et al., 2017). One of the regions most modified by agricultural expansion is the western portion of Bahia State, where over 1Mha were cleared between 2002 and 2010 (Salmona et al., 2016). The western Bahia region is one of the most productive areas in the Cerrado, especially for soybeans, with most production going to China, Brazil's internal market, and Europe (https://trase.earth, last accessed July 12, 2020)

The western Bahia region is composed of plateaus/mesas located in the west, with maximum elevation of 808 m. In the eastern portion, the region features lowlands (known as *Depressão do Rio São Francisco*) with a minimum elevation of 433 m (Nou and Costa, 1994). Precipitation is positively correlated with elevation, ranging from ~800 mm in lowlands to ~1,600 mm in the plateaus/mesas (Brannstrom et al., 2008). Declining rainfall from west to east is a major determinant in land price and crop production, which also determines land conversion patterns in this region (Oliveira et al., 2014). Soils are diverse, including deep well-drained Oxisols (Latosols) of medium texture in the highest parts of the plateau, sandy texture (sandy quartz) on irregular terrain, rocky soils (Lithosols) of sandy to medium texture on steep slopes and escarpments, and hydromorphic/organic soils across floodplains (Nou and Costa, 1994). Our study area spans 9,409 km^2 in the western Bahia region (11°43'S 45°52'W to 12°36'S 44°59'W; **Figure 4.1**), covering parts of the municipalities of Barreiras, Luís Eduardo Magalhães, São Desidério, and Riachão das Neves. Together, these municipalities correspond to the highest Gross Domestic Product (IBGE, 2015) and one of the most productive areas in Matopiba for soybeans (3% of national productivity in 2018: https://sidra.ibge.gov.br/pesquisa/pam/tabelas, last accessed in 08/21/2020).

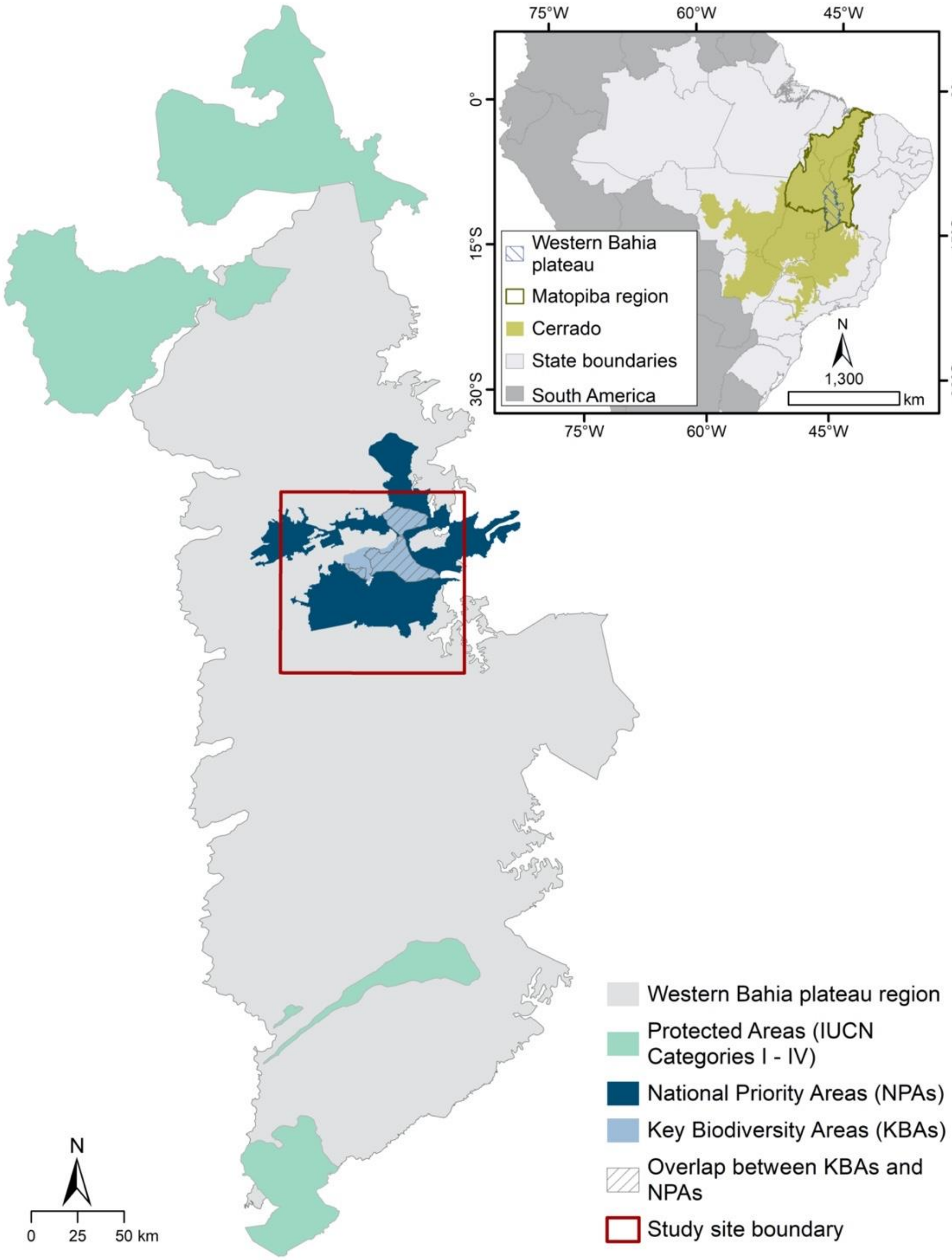

Figure 4.1. Map showing study site location in the western Bahia plateau region, areas identified as conservation priorities in regional prioritization exercises (i.e. NPAs and KBAs) within study area, and the existing surrounding protected areas.

The boundary of our study area was defined based on the availability of a fine-scale map featuring major Cerrado physiognomic types at 5m resolution (Ribeiro et al. 2020), used as one of the inputs in our prioritization scheme. At least 89% of the study site is privately owned and mostly managed for agricultural activities that presently cover ~43% of the total study area (Ribeiro et al., Chapter 3). Despite its agribusiness focus, our study area supports many ecosystems and species that are endangered due to habitat loss and climate change, such as Seasonally Dry Tropical Forests (Ferrer-Paris et al., 2019), the Hyacinth Macaw (*Anodorhynchus hyacinthinus*), the Brazilian merganser (*Mergus octosetaceus*) and the Wagler's Woodcreeper (*Lepidocolaptes wagleri*), with the latter also being an endemic species with a restricted distribution (IUCN, 2020). Moreover, the area features a wide range of Cerrado physiognomic types, including Semi-deciduous Forests, Riparian Forests, Swamps, Marshes, Woodlands, Savannas, and Scrub (Ribeiro et al. 2020). The Forest Code regulation is an important policy mechanism in the region, protecting ~24% of the total study site area and ~37% of its remaining native vegetation (Ribeiro et al., Chapter 3).

Our study area has featured in some broad-scale conservation planning exercises; for instance, it includes two Key Biodiversity Areas (KBAs: Sawyer et al., 2017), and overlaps with two other high conservation priority areas (WWF-Brazil, 2015) supported by the Brazilian Ministry of the Environment. These areas were selected as conservation priorities for ensuring biodiversity protection, especially for endangered species, as well as serving as potential corridors and stepping stones between some of the largest Cerrado protected areas, which despite their protective status also have high land clearing rates (Garcia et al., 2011; WWF-Brazil, 2015): *Serra Geral do Tocantins* Ecological Station and *Nascentes do Rio Parnaíba* National Park to the north of our study site; *Veredas do Oeste Baiano* Wildlife Reserve and *Grande Sertão Veredas* National Park in southern Bahia state.

4.2.2 Datasets

Given the lack of biological data available for our study site, we acquired the following datasets available at 5-meter spatial resolution: species-specific habitat suitability models for nine endemic and/or endangered bird species (developed in Chapter 2), and a land cover product for 2013 featuring 11 physiognomic types (Ribeiro et al. 2020, modified in Chapter 3). The datasets were used as biodiversity features in our priority-setting scheme to produce alternatives for conservation priorities using different biodiversity targets. For this spatial planning analysis, 5m grids of habitat suitability and land cover were aggregated to 30 meters in order to match the spatial resolution used by most decision-makers in Brazil.

We also used a 30-meter land cover dataset for 2011 developed by Oliveira et al. (2014) for the entire Western Bahia plateau region. This dataset, featuring seven classes (natural vegetation, converted areas, silviculture, water, urban, highways, and farming areas), was used to create a resistance raster for our regional connectivity analysis. For our local connectivity analysis, the 2013 5-meter land cover product was aggregated to 10 meters using a majority function, in order to speed computation and reduce noise in the connectivity analysis.

In addition, we obtained a land use projection dataset available for Brazil for the year 2030 at 500-meter resolution (Soares-Filho et al. 2016; available at maps.csr.ufmg.br/), which was used in our vulnerability analysis. This dataset was developed based on projections of agricultural expansion for 2024 (MAPA, 2014) and extrapolated to 2030 by using historical land clearing trends between 1994 and 2013 in the OTIMIZAGRO model (Soares-Filho et al., 2016). The model simulates the expansion of row crops and forest plantations based on crop suitability, future industry demands, and probabilities of land clearing, which were estimated through spatial determinants such as distance to roads and areas previously converted (Soares-Filho et al., 2006). Projected land clearing rates in the Cerrado were based on averages for the years 2009 – 2014 and constrained to areas legally permitted for land clearing (i.e., excluding areas under Forest Code protection). For further details, see Soares-Filho et al. (2016).

4.2.3 Fine-scale conservation priorities

We used the *Zonation* spatial prioritization framework and supporting software (Moilanen et al., 2005) to set conservation priorities across our study site. *Zonation* ranks cells in ascending priority based on a reverse stepwise removal heuristic algorithm and a cell marginal loss function. Spatial priorities are based on empirical conservation goals defined for each individual biodiversity feature (e.g., species, habitats, communities) while using a boundary length penalty that prioritizes contiguous rather scattered areas (Lehtomäki and Moilanen, 2013; Moilanen, 2007). The prioritization scheme is based on raster cells representing their biological value and accounts for complementarity between biodiversity features across a given region (i.e. balanced representation of features). The resulting priority-ranking ranges from 0 to 1: the lowest value (0) is the first cell removed from the landscape (i.e. the least important in terms of biodiversity complementarity) and the highest value (1) is the last cell removed (i.e. the most important for holding biodiversity representation in the landscape). The order of cell removal is documented and used for selecting landscape fractions (e.g., top or best 10%, lowest 10%) in a post-processing step. The landscape fractions represent areas of highest conservation values within a given percentage of the landscape, which in our case represents the fraction of the landscape left unprotected by the Forest Code (i.e. the top 10% represents the best solution within 10% of the unprotected portion of the study site).

Our priority-setting exercise explored alternative schemes of local conservation set-asides in private lands to examine differences in protection status and spatial distribution of priority sites using two biodiversity organizational levels: physiognomic types and species' habitats. Our conservation goal is to efficiently improve habitat representation and connectivity in an area already identified as having high conservation priority at the biome level. We evaluated changes in land requirements (i.e. by comparing the top/best 10%, 20%, and 30% sites with the highest conservation values) to maximize habitat representation for each landscape fraction of our *Zonation*-based conservation priority sites, as a measure of conservation success. We complemented our priority sites representation estimates with Forest Code protection estimates from Ribeiro et al. (Chapter 3) to estimate the total protection status of each biodiversity feature in a post-processing step. Grande et al. (2020) estimated that Cerrado landscapes must retain at least 37% of native vegetation to maintain landscape connectivity. To meet this connectivity threshold, we selected the 30% biodiversity target to perform further analyses (i.e. landscape connectivity, conservation hotspots, landscape metrics).

We calculated marginal loss using the Additive Benefit Function (ABF) cell removal rule, which sums the values across biodiversity features seeking to maximize their proportional representation (Moilanen, 2007). We developed three conservation alternative schemes using vegetation structural types and/or habitat suitability maps for birds as biodiversity features (**Table 4.1**). In order to avoid prioritization of competing land uses, we assigned negative weights to land use categories in two of the alternatives provided.

Table 4.1. Biodiversity features and weights used in different alternatives of conservation priority areas.

	Alternative 1	**Alternative 2**	**Alternative 3**
Biodiversity features	Land cover map	Habitat suitability models for endangered/ endemic avifauna	Habitat suitability models + Land cover map
Weights	Physiognomic types (sum to 1): endemics are weighted as 2x the weight of all other vegetation types; Land use (sum to -1): silviculture = -1.5; all other types = -12.5/5	Endangered = 2.0; Endemics = 1.0;	Habitat suitability model (sum to 1): endangered are weighted 2x all other species; Physiognomic types (sum to 1): endemics are weighted 2x all other vegetation types; Land use (sum to -2): -2/6

To avoid prioritizing sites under current policy protection, areas under Forest Code policy were masked out of our analysis using a dataset consolidated by Ribeiro et al. (Chapter 3).

4.2.4 Conservation hotspots: potential vulnerability of conservation priorities to land use expansion

To assess the vulnerability status of conservation priorities to projected land clearing, we aggregated the land-use projections developed by Soares-Filho et al. (2016) from seven into two categories - natural vegetation vs. converted areas. To match the spatial resolution between datasets, the 500m projection raster was resampled to 30 m before overlay with the top 30% conservation priorities. We then produced a vulnerability ranking based on the percentage of projected land conversion inside conservation priority areas (**Table 4.2**).

Table 4.2. Vulnerability ranking categories to identify conservation hotspots based on the percentage of projected land conversion in 2030 inside conservation priority areas.

Vulnerability ranking	Percentage of land conversion
Low	up to 25%
Medium	between 25% and 50%
High	between 50% and 75%
Very high	more than 75%

4.2.5 Identifying important areas for regional and local connectivity conservation

In addition to identifying local biodiversity conservation priorities, we also aimed to identify important areas for regional and local connectivity conservation. We developed a regional connectivity analysis to understand the contribution of our study site to maintain connectivity between surrounding protected areas at the regional scale. We further identified multiple important connective elements between local conservation priorities in our study site to understand how an emphasis on connectivity affects conservation land requirements.

We used *Circuitscape* 5.0 (Anantharaman et al., 2019) to explore structural connectivity patterns in two scenarios: a) regional landscape connectivity, performed among the largest Cerrado protected areas that surround the Western Bahia plateau, and b) local landscape connectivity, performed between conservation priority areas in our study site. *Circuitscape* uses electronic circuit theory to predict multiple pathways of landscape connectivity in heterogeneous landscapes and thus, provide measures of connectivity and isolation of elements (i.e. habitat patches, populations, protected areas) that can be used to identify priority corridors for conservation planning (McRae et al., 2008). The landscape is represented as a conductive surface characterizing resistance to movement for each raster grid cell, which may have values assigned as the inverse of habitat suitability (e.g., high resistance values assigned for unsuitable habitat or barrier to movement and vice versa). The circuit model then uses random walk functionality to calculate total landscape resistance and derive current flows between pairs of focal points (i.e. protected areas, core habitat patches), which are represented as sources and destinations for connectivity analysis (McRae et al., 2016, 2008)

Because we were interested in structural connectivity patterns across the matrix, our resistance raster for both analyses was created based on the assignment of lower values for natural vegetation and higher values for developed areas that do not hold enough habitat and would thus represent barriers for multiple species (i.e. highways, urban areas). All parameters used for both connectivity scenarios are shown in **Table 4.3**. Our circuit models were operated in "pairwise mode", meaning that current maps of each focal node pair are added, resulting in a cumulative current map that indicates the potential current density of a given cell. Pinch points, or areas of highest current density (i.e. highest cell values), were derived from the cumulative current maps (McRae et al. 2008) and were used to identify areas important for connectivity between focal points. As a post-processing analysis, we calculated the percentage of high current flow within our study site relative to the entire Western Bahia plateau to estimate the contribution of our study site to regional connectivity. In addition, we performed an overlay between the local connectivity priorities (i.e. areas of highest current density) and our conservation priority areas to identify the most important areas for biodiversity and connectivity conservation and to understand how conservation priorities contribute to landscape structural connectivity.

Table 4.3. Parameters used in each connectivity scenario performed: across protected areas (regional connectivity) and across conservation priorities (local connectivity). Numbers in parenthesis on the first row represent the resistance value attributed to each class in both scenarios.

Parameters	**Regional connectivity**	**Local connectivity**
Resistance raster	Natural vegetation (1) Silviculture (25) Farming and Converted areas (50) Urban and Highways (100)	Natural vegetation (1) Silviculture (25) Farming (50) Urban and Highways (100)
Cell size	30 m	10 m
Circuitscape mode	Pairwise	Pairwise
Focal points	Southernmost point in the Protected Areas to the North of the study region (*Serra Geral do Tocantins* Ecological Station, *Nascentes do Rio Parnaíba* National Park); Northernmost point in the Protected Area to the South of the study region (*Grande Sertão Veredas* National Park); Centroid point in the *Veredas do Oeste Baiano* Wildlife Refuge	Centroid points within the five most contiguous (> 1,000 ha) conservation priority areas that were at least 50 km apart to ensure areas in each border of the site extent

4.2.6 Analysis of alternative conservation designs

We quantified the landscape patterns of all resulting conservation designs (i.e. *Zonation* priority sites, Forest Code areas, and landscape connectivity) using FRAGSTATS (McGarigal and Marks, 1994) to calculate the following metrics: percentage of landscape, core area, edge density, core area percentage of landscape, and mean core area size. These metrics were calculated separately for the western (i.e. plateau) and eastern (i.e. valley) landscapes, and divided into: 1) portion of Forest Code sites that do not contribute to connectivity, 2) total area of Forest Code sites, 3) portion of Forest Code and *Zonation* priority sites that do not contribute to connectivity, and 4) total areas covered by Forest

Code policy, *Zonation* priority sites, and areas of high current flow. We used an edge depth of 100 meters to calculate core area metrics. Patches smaller than 10 ha were excluded from conservation priorities and areas of high current flow in our analysis under the assumption that small isolated habitat patches would have lower conservation value. We opted for not excluding small patches from Forest Code sites given that these areas are already under protection.

4.3 Results

4.3.1 Fine-scale conservation priorities

In alternative 1 (**Figure 4.2**), conservation priority sites (within 30% of the unprotected portion of our study area) were mostly concentrated towards the eastern side of the study site, scattered between north-south, with the largest concentration of contiguous patches in the northeastern portion. In contrast, priority sites in alternatives 2 and 3 (**Figure 4.2**) were mostly concentrated in the northwest and southeast portions of the landscape. In all conservation alternatives provided, the largest blocks are located in the northern and southern parts of the study site.

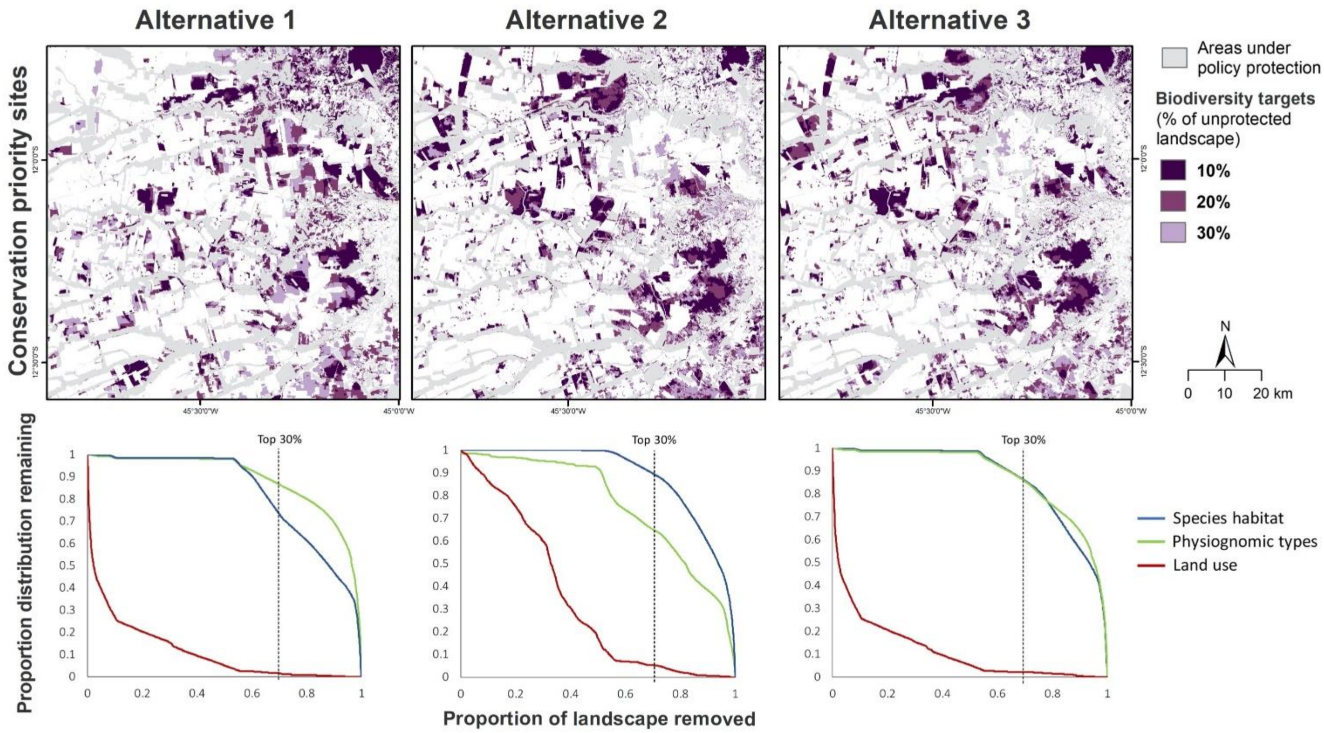

Figure 4.2. Alternative schemes of local conservation set-asides in private lands featuring the spatial distribution and performance curves of *Zonation* conservation priority outputs featuring conservation priority sites for up to the top 30% unprotected landscape. The alternatives differ in terms of their biodiversity features used as inputs in the *zonation* model: alternative 1 uses physiognomic types, alternative 2 uses habitat suitability models for nine endemic or endangered birds, and alternative 3 accounts for both previous biodiversity features. The performance curves from each respective *zonation* output (alternative schemes 1, 2, and 3) are at the bottom, showing the average of biodiversity features (i.e. habitat suitability models for 9 bird species, 11 physiognomic types, and 6 land use types) for the top 30% solution.

To understand differences between biodiversity features, we compared the *Zonation* model performance curves (i.e. average representation of biodiversity features) for the top 30% of our prioritization solutions for alternative schemes 1, 2, and 3 (**Figure 4.2**). Alternative 3 features the best solution for maximizing biodiversity, retaining 86% of both biodiversity features while featuring a low proportion of alternative land use (~2% retained). Alternatives 1 and 2 represent trade-offs in terms of feature protection given that both alternatives protected a similar proportion (~70%) of each preferred biodiversity feature within the top 30% solution (i.e. alternative 1 protected a larger fraction of physiognomic types and alternative 2 protected a larger fraction of species habitat). Alternative 3 is the most efficient for equally retaining 86% of both biodiversity features, and thus representing the best alternative for protecting two different organizational levels. For the same landscape fraction, intensive land uses (i.e. agriculture, silviculture, urban areas, major highways, barren) were avoided in all alternatives provided (< 6% retained). Despite being more efficient, alternative scheme 3 requires spatially explicit information for multiple species. Given that biological knowledge is still very limited for the Cerrado regarding species-habitat relationships (Colli et al. 2020), this alternative is not easily reproducible in other Cerrado regions and for different biological groups. At the same time, the other two alternatives provide similar protection status to alternative 3. Because alternative scheme 1 retained the lowest proportion of land use in the top 30% solution while prioritizing more than 70% of both biodiversity features, we chose this result to perform the next analyses: identifying conservation hotspots and evaluating local structural connectivity.

We used alternative scheme 1 to compare the protection status of both biodiversity features considering different landscape fractions as biodiversity targets (i.e. top 10%, 20%, and 30% of the landscape portion left unprotected by the Forest Code policy). Areas under Forest Code policy, combined with our priority sites considering the top 10%, 20%, and 30% fractions of the unprotected landscape, collectively protect 27.9%, 35.1%, and 42.2% of our study area. Regarding the protection status of individual physiognomic types (**Table C.1**), the top 10% solution alone increased protection by at least 6%, with a significant increase for Cerrado Woodland (40%), Shrubby Grassland (45%), Scrub Cerrado (59%), and Semi-deciduous Forest (80%). Together with Forest Code sites, the top 10% solution increased the protection status to more than 90% for 4 physiognomic types: Riparian Forest

(96%), Tropical Dry Forest (92%), Semi-deciduous Forest (99%), and Scrub Cerrado (93%). Priority sites within the top 20% solution contribute to adding complementary habitat, increasing protection status by at least 20% for Savanna and Open Savanna physiognomies while also increasing representation for Cerrado Woodland and Shrubby Grassland. The top 30% solution increases protection of physiognomic types by at least 32%. For this target, our priority sites and areas under Forest Code policy collectively increase the protection status of all individual physiognomic types to > 90%, except for Open Savanna and Shrubby Grassland, which achieved a total protection of 65% and 84%, respectively.

Regarding the conservation status of species habitats (**Table C.2**), comparing the top 10%, 20%, or 30% of the unprotected landscape results in protecting an average of 10%, 23%, or 37% of all species habitats, respectively. The top 10% solution alone increases protection by at least 10% for 4 species: *Anodorhynchus hyacinthinus* (~10%), *Neothraupis fasciata* (~11%), *Herpsilochmus longirostris* (~13%), and *Lepidocolaptes wagleri* (~55%). For these same species, the top 10% solution and Forest Code sites would protect a total of: ~50%, ~44%, 91.8%, and ~95.3% of their potential habitats, respectively. Moreover, when accounting for sites under Forest Code policy, the top 10% solution provides at least 90% habitat protection for the following species: *Herpsilochmus longirostris* (91.8%), *Lepidocolaptes wagleri* (95.3%), *Myiothlypis leucophrys* (97.5%), and *Syndactila dimidiata* (94.3%). The top 20% solution is complementary to the Forest Code for increasing representation of species habitats underrepresented in policy. The species for which habitat protection increased by at least 20% include *Euscarthmus rufomarginatus* (23%), *Melanopareia torquata* (23%), *Cypsnagra hyrundinacea* (24%), and *Neothraupis fasciata* (25%). The top 30% increases habitat protection for these same species by at least 32%. For this target, our priority sites and areas under Forest Code policy all together result in a total protection status of at least 70% of all species potential habitats, reaching more than 90% protection for *Syndactyla dimidiata* (96%), *Myiothlypis leucophrys* (98%), *Lepidocolaptes wagleri* (96%), and *Herpsilochmus longirostris* (94%).

4.3.2 Conservation hotspots

The conservation hotspot analysis (**Figure 4.3**) revealed that, in total, an area of 65,884 ha is projected to be cleared by 2030 in our site, of which 31% (~20,703 ha) is within our priority sites. 55% of the area covered by our priority sites is vulnerable to land conversion by 2030. The majority of the hotspots (~55%) carry low vulnerability to land-use expansion, with remaining hotspots having medium (~17%), high (~9%), and very high (~19%) vulnerabilities.

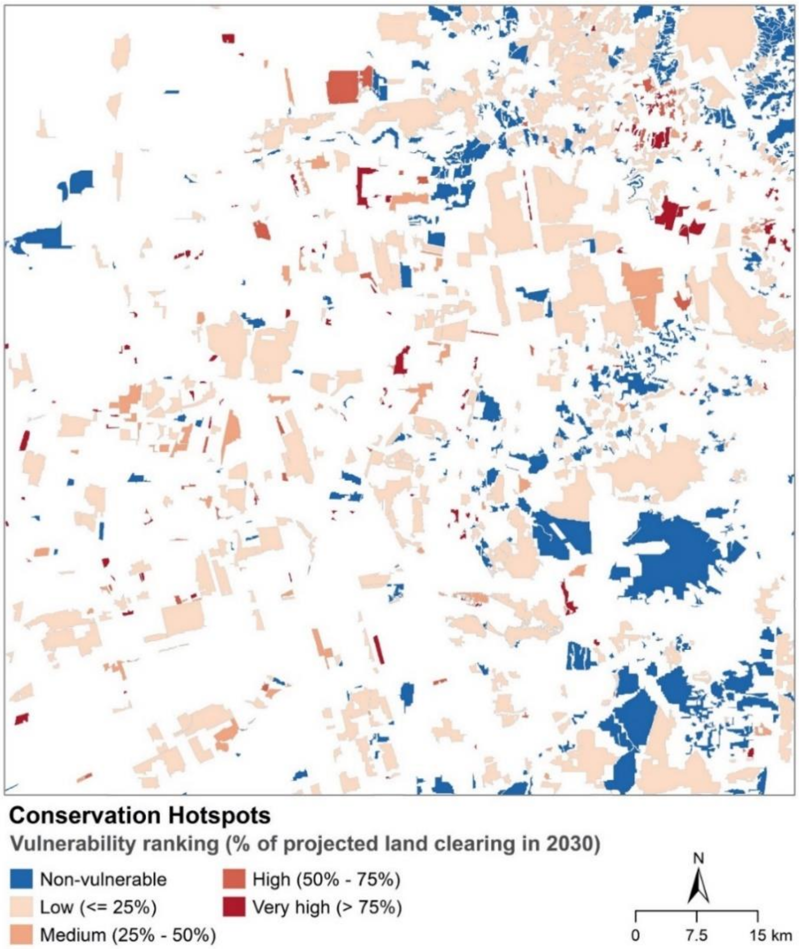

Figure 4.3. Map showing conservation hotspots based on vulnerabilities of conservation priorities to projected land clearing in 2030.

Despite carrying low vulnerability due to smaller proportion of land clearing relative to total size, these sites concentrate the largest area coverage of projected cleared lands (i.e. ~8,059 ha projected to be cleared by 2030 in areas of low vulnerability). The remainder of land clearing is projected to happen in areas of very high (~5,481 ha converted), medium (~4,379 ha converted), and high (~2,784 ha converted) vulnerabilities. Large conservation hotspots (> 1,000 ha) represent 39 areas spanning 6,600 ha under threat of being cleared. Most of these areas (30 hotspots) are ranked as low vulnerability representing a total of

~4,402 ha projected to be cleared, whereas two hotspots are ranked as medium (~1,103ha cleared) and high (~1,094ha cleared) vulnerabilities.

4.3.3 Landscape connectivity

The regional landscape connectivity analysis (**Figure 4.4**) revealed that the eastern portion of the western Bahia plateau plays a key role in maintaining North-South structural connectivity between major protected areas in the Cerrado.

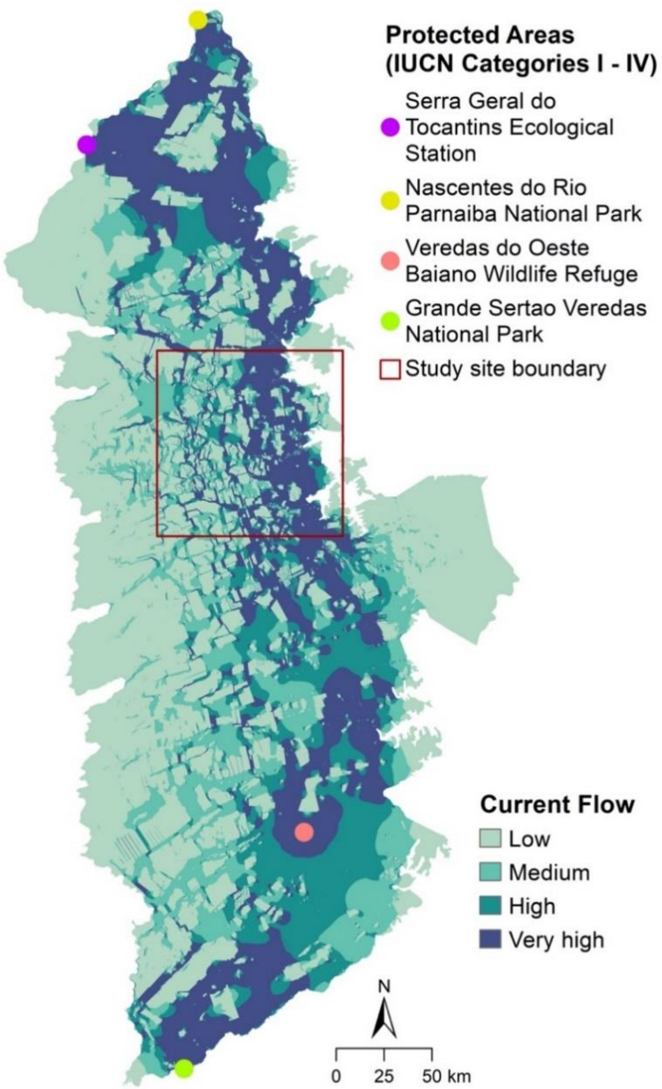

Figure 4.4. Map showing Circuitscape cumulative current flow for the Western Bahia plateau region.

Our study site alone holds ~17% of high current flow areas of the total western Bahia plateau, indicating that the site is important for connectivity conservation between protected areas. The local connectivity analysis (**Figure 4.5**) shows that areas of high current flow correspond to ~10% of our study site. Moreover, ~75% of all areas important for local connectivity overlap with areas under Forest Code policy or with our *Zonation* priority sites. In terms of connectivity, Forest Code sites overlap by ~19% with areas of high current flow, while our priority sites have ~15% overlap with areas of high current flow. In terms of spatial distribution, East-West connectivity is mostly maintained through the Forest Code policy while our priority sites are responsible for increasing North-South connectivity.

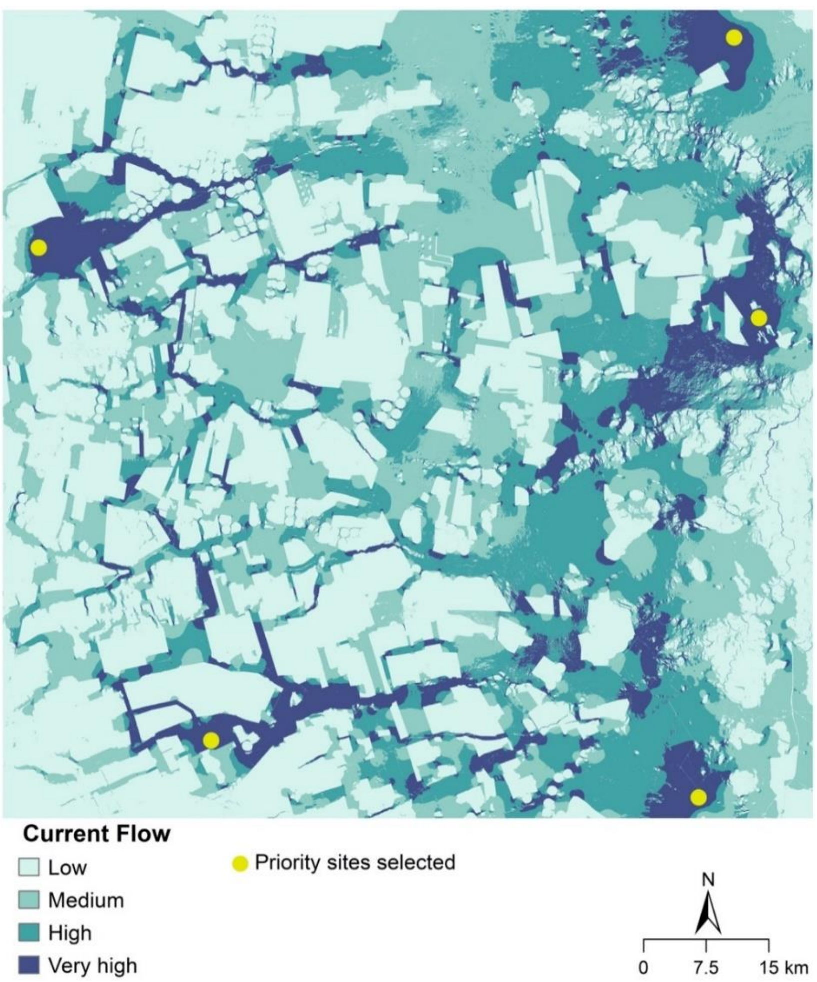

Figure 4.5. Circuitscape map showing cumulative current flow, which indicates areas of structural connectivity.

4.3.4 Alternative conservation designs

Based on the landscape metrics calculated (**Table 4.4**, **Figure 4.6**), the plateau in our study site holds a larger portion (i.e. class area, ha) of the different conservation designs, larger core areas, and lower edge density for all scenarios evaluated. The valley has a smaller extent, and thus retains a larger proportion of the conservation scenarios (i.e. percentage of landscape) and core areas. Considering only the plateau landscape, the core area size substantially increases when accounting for connectivity. For instance, the portion of Forest Code sites that do not contribute to connectivity have a mean core area size of 26.8 ha, increasing to 217.7 ha in the scenario considering connectivity. Edge density also decreases, but less dramatically, in conservation designs considering connectivity.

Table 4.4. Landscape metrics (in bold) calculated for the West (plateau) and East (valley) landscapes.

Landscape	Scenario	Type	**Class Area (ha)**	**% of landscape**	**Edge Density**	**Core area % of landscape**	**Mean core area size (ha)**
West	Forest Code only	Matrix	618,132.15	83%	12.6	76%	148.9
	Forest Code only	Conservation	127,363.23	17%	**61.2**	10%	26.8
East	Forest Code only	Matrix	138,150.18	72%	51.8	48%	27.7
	Forest Code only	Conservation	52,623.54	28%	**136.1**	11%	6.7
West	Forest Code + Connectivity	Matrix	544,883.58	73%	16.1	66%	86.8
	Forest Code + Connectivity	Conservation	200,611.80	27%	**43.7**	19%	217.7
East	Forest Code + Connectivity	Matrix	113,155.65	59%	70.4	36%	12.1
	Forest Code + Connectivity	Conservation	77,618.07	41%	**102.7**	19%	22.9
West	Forest Code + Conservation Priorities	Matrix	486,680.67	65%	23.6	55%	66.4

Landscape	Scenario	Type	**Class Area (ha)**	**% of landscape**	**Edge Density**	**Core area % of landscape**	**Mean core area size (ha)**
	Forest Code + Conservation Priorities	Conservation	258,814.71	35%	**44.4**	24%	52.5
East	Forest Code + Conservation Priorities	Matrix	94,183.47	49%	83.5	27%	9.1
	Forest Code + Conservation Priorities	Conservation	96,590.25	51%	**81.4**	27%	22.3
West	Forest Code + Conservation Priorities + Connectivity	Matrix	413,432.10	55%	27.7	46%	42.9
	Forest Code + Conservation Priorities + Connectivity	Conservation	332,063.28	45%	**34.5**	33%	413.1
East	Forest Code + Conservation Priorities + Connectivity	Matrix	69,188.94	36%	117.0	17%	4.3
	Forest Code + Conservation Priorities + Connectivity	Conservation	121,584.78	64%	**66.6**	36%	96.2

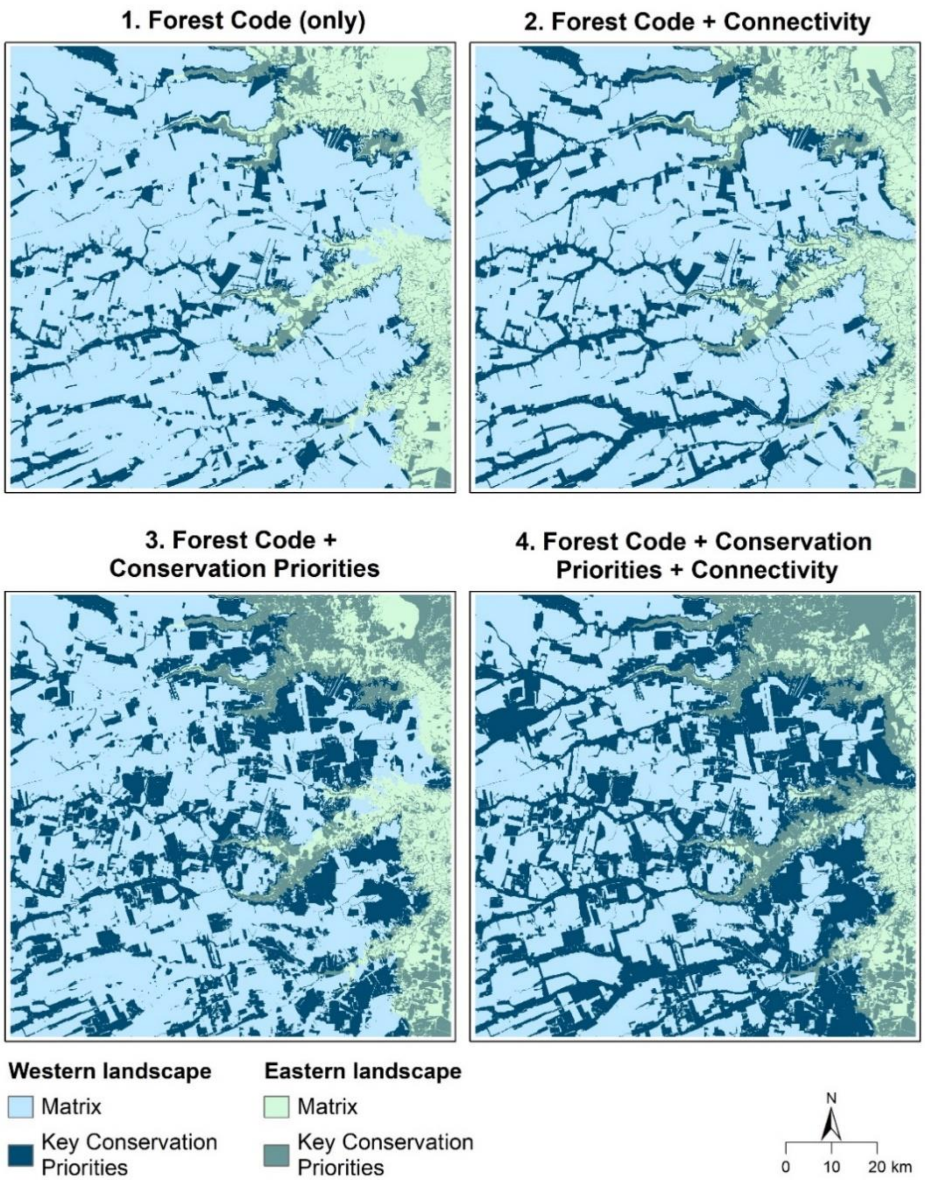

Figure 4.6. Inset maps showing areas under current conservation policy (i.e. Forest Code) and/or priorities for connectivity and biodiversity conservation for Western (plateau) and Eastern (valley) landscapes in study site.

4.4 Discussion

Our conservation priority-setting scheme demonstrates that expanding conservation efforts in Cerrado private agricultural lands is necessary in order to increase representation of essential habitats for endangered and endemic bird species and vegetation communities while providing complementarity to areas under Forest Code policy. Differences in biodiversity targets revealed that our *Zonation* priority sites combined with areas under land-use regulation would increase the total area protected in the study site to 27.9%, 35.1%, and 42.2%, regarding sites within the top 10%, 20%, 30% solutions, respectively. Sites within the top 10% of the landscape are effective for increasing protection by at least 10% for four species: *Anodorhynchus hyacinthinus* (~10%), *Neothraupis fasciata* (~11%), *Herpsilochmus longirostris* (~13%), and *Lepidocolaptes wagleri* (~55%). Accounting for Forest Code sites, these same species would have their habitat protection increased to ~50%, ~44%, ~91.8%, and ~95.3%, respectively. When additional landscape fractions are added for protection, habitat representation is substantially increased for other species not well-represented in areas under Forest Code policy, such as *Cypsnagra hyrundinacea* and *Melanopareia torquata*. Our results revealed that sites within the top 30% combined with Forest Code sites would collectively protect at least 70% of all species potential habitats. Private conservation areas also showed opportunities for complementing and maximizing habitat representation in other countries relative to existing public lands and protected areas. In the U.S., for instance, conservation easements contribute to added protection for 10% of the ecosystems in the Rocky Mountains (Graves et al., 2019), and in South Africa, these areas were especially important for protecting endangered habitat types (Gallo et al., 2009).

Our priority-setting results were similar to findings reported by Harris et al. (2005) for the Atlantic Forest, a neighboring and highly fragmented biome in Brazil, demonstrating that not all remaining habitat in fragmented landscapes is equally important and sites must be prioritized to complement current conservation efforts. In contrast to policy-driven targets (i.e. CBD's Aichi Target 11), science-based estimates recommend a range of 25 – 75% of a given landscape to be managed for biodiversity conservation as the primary goal in order to maintain biodiversity and ecological processes (Butchart et al., 2015; Noss et al., 2012). Specifically for highly modified agricultural regions, Belote et al. (2020) recommend conserving 90% of the remaining natural vegetation. The Forest Code policy alone covers ~24% of our study site and protect ~36% of the remaining natural vegetation (Ribeiro et al., Chapter 3). Compared to current efforts, we found that setting aside ~40% of the study site for protection (i.e. top 30% target + Forest Code sites) would increase the total protected vegetation by 38.5%. These combined efforts would protect 397,391 ha (~75% of the total remaining vegetation), which corresponds to >90% habitat protection for 10 biodiversity features (6 physiognomic types and habitats of 4 species). These protection estimates, however, are not based on the original extent of biodiversity features given such information is unavailable for our study area.

Current discussions for the CBD post-2020 biodiversity framework (https://www.cbd.int/sbstta/sbstta-24/post2020-monitoring-en.pdf last accessed on 07/30/2020) suggest that the biodiversity target will be increased to 30% landscape protection; this new policy target corresponds to the area covered by Forest Code sites and the top 10% solution of our priority sites. In our analysis, a total of 30% landscape protection leads to protecting at least 40% of biodiversity features underrepresented in areas under Forest Code policy and reaches >90% protection for 8 biodiversity features (i.e. 4 physiognomic types and habitats of 4 species). This result suggests that increasing policy targets to 30% is effective for increasing habitat representation and their protection status. However, previous findings from Grande et al. (2020) revealed that Cerrado landscapes must have a minimum of 37% of protected native habitat to maintain landscape connectivity. Sites within our top 30% solution combined with Forest Code sites represent ~40% of the landscape, which would be effective for meeting this connectivity threshold.

Land conversion comes at the expense of Cerrado biodiversity, with high land clearing rates occurring both in and outside protected areas (Matricardi et al., 2019). Given the fast rate of commodity-driven land conversion undertaken in the Western Bahia plateau, protected areas nearby this region are at risk of becoming isolated (de Oliveira et al., 2017b; Salmona et al., 2016). Our regional connectivity analysis demonstrated that the study area makes an important contribution in maintaining structural connectivity between the surrounding protected areas in the western Bahia. Moreover, our local connectivity analysis supports findings from Oliveira et al. (2017a) for demonstrating that Forest Code sites have an important role in maintaining structural connectivity within our study area, especially in an East-West direction. These results complement findings from Grande et al. (2020), Magioli et al. (2016), Metzger et al. (2019), Oliveira et al. (2014), Ribeiro et al. (Chapter 3), and Soares-Filho et al. (2014), in which enforcing protection of Forest Code areas is crucial to guaranteeing biodiversity and connectivity conservation in agricultural landscapes.

Our priority sites provide an important complement to the Forest Code policy in maintaining North-South structural connectivity and thus, providing connectivity between surrounding protected areas at the regional scale. However, only ~15% of our priority sites and ~19% of the Forest Code areas are important for connectivity conservation. This result indicates that the remaining vegetation available in our study area, regardless of protection status, might not be well connected throughout the landscape. Given that our study area contributes significantly to regional connectivity, immediate protection of its remaining prioritized sites is recommended for maintaining structural landscape connectivity between large Cerrado protected areas surrounding the Western Bahia plateau.

In-situ datasets of biological observations and measurements are mostly unavailable and difficult to be collected in Cerrado private lands due to accessibility restrictions and funding availability (Rosa, 2020). In this context, maps featuring detailed vegetation physiognomic types may serve as biodiversity surrogates for Cerrado conservation priority-setting exercises (Monteiro et al., 2020). Medium to coarse spatial resolution (> 30 m) imagery has traditionally failed to differentiate a wide range of heterogeneous Cerrado physiognomic types, and high spatial resolution imagery proved necessary to map these detailed vegetation categories (Ribeiro et al., 2020). Such fine-scale maps might be particularly useful to priority-setting exercises in Cerrado agricultural landscapes. Vegetation remnants in agricultural lands are often constrained to small habitat patches, which are usually not featured in broad-scale prioritization exercises (Paese et al., 2010). This hypothesis can guide future studies to test the feasibility of using fine-scale vegetation maps as reliable biodiversity surrogates in Cerrado conservation priority exercises.

Accounting for land use projections to identify conservation hotspots is an effective strategy to inform decision-makers in prioritizing areas when resources are limited (Monteiro et al., 2020; Munang et al., 2013). Recent land-use projections show that up to 34% of Cerrado natural lands may be cleared by 2050, which could potentially drive ~480 endemic plants to extinction and alter ecosystem functioning (Strassburg et al., 2017). The Western Bahia region in particular contributes to increasing the isolation of some of the largest protected areas in the Cerrado. The plateau region alone increased its converted areas from 795,503 ha to over 2.8 Mha from 1988 to 2011. Moreover, between 2002 and 2010, the average land clearing rate in the Western Bahia region 1.2% per year - twice as much as the average observed for the entire Cerrado (de Oliveira et al., 2017a). Projections specifically developed for this region show that land clearing rates may continue around 1% per year until 2050 (Salmona et al. 2016). In this context, we estimated the vulnerability of our conservation priorities to land use change and found that most of them are not under a high level of vulnerability, according to land use projections from Soares-Filho et al. (2016).

Given that areas of low vulnerability to land use are more prone to be included by decision-makers involving the agricultural sector (Brum et al., 2019; Lemes et al., 2019), our hotspot results indicate that the western Bahia landscape has great potential to reconcile conservation and agriculture. Despite the positive overall result, we have identified that over 20,000 ha (~10%) of priority sites are under some threat of potential land use expansion by 2030. These areas hold primary habitats of many endemic birds, such as the White-rumped Tanager (*Cypsnagra hirundinacea*), a seed disperser (Bagno and Marinho-Filho, 2001), as well as the Wagler's Woodcreeper (*Lepidocolaptes wagleri*) and the Hyacinth Macaw (*Anodorhynchus hyacinthinus*:), which are both endangered and have small populations inhabiting the Cerrado (Dornas et al., 2013; ICMBio, 2018).

Although conservation and agricultural conflicts remain a serious issue, agricultural activities in the Cerrado do not entirely depend on land clearing given its potential to increase commodity productivity without opening new lands (Soterroni et al., 2019; Strassburg et al., 2014). Establishing zero-deforestation commitments from Cerrado commodity supply chains (such as the Cerrado Manifesto: Gibbs et al., 2015; Soterroni et al., 2019) provides an opportunity to improve landscape sustainability and reduce land clearing rates. For instance, by restricting soybean expansion, the Cerrado Manifesto could prevent ~3.6 Mha of land clearing by 2050 (Soterroni et al., 2019). Additional set-asides in private lands are important for maximizing biodiversity protection and maintaining connectivity beyond the Forest Code policy, which in many instances is the only conservation strategy implemented to date within Cerrado agricultural landscapes. As a result of a large extent of natural vegetation remaining in private lands, the Cerrado offers new opportunities to improve conservation efforts within its agricultural areas (Klink, 2019; Strassburg et al., 2017). We identified important areas for biodiversity and connectivity that can be implemented through well-established policy mechanisms in Brazil, such as RPPNs, which are recognized by IUCN as privately protected areas and can be counted towards international agreements, such as the CBD and REDD+ (Stolton et al., 2014). Incentives to implement RPPNs include tax exemption over the area protected, priority on bank loans and agricultural credits, and transferable property rights (Rambaldi et al., 2005; Wiedmann and Guagliardi, 2018). Given that there are no limits on their size, RPPNs generally protect small patches that are important for biodiversity and connectivity conservation, whereas large areas are favored by natural reserves of strict protection (Mittermeier et al., 2005). For this reason, these privately-owned natural reserves have great potential to serve as one of the implementation mechanisms for conservation set asides in Cerrado agricultural landscapes. These strategies might be financed by the private sector through payments for carbon sequestration, ecosystem services, and incentives to zero-deforestation commitments. However, innovative arrangements of policy-setting involving the private sector and a greater engagement of producers, industry, and consumers are necessary for an effective implementation of conservation strategies in Cerrado private lands (Klink, 2019).

Caveats and recommendations for future studies

Our approach considered fine-scale EO data consolidated by Ribeiro et al. (Chapter 3), such as a land cover map discriminating detailed vegetation physiognomic types (Ribeiro et al. 2020) updated to 2013 and species-specific habitat suitability maps based on land cover information available from the literature. Biological observations and measurements were not available to our study site, and we were unable to collect new data to validate our species habitat models. Future studies should incorporate *in-situ* data to assess habitat quality and actual patterns of species distribution and persistence in this landscape. It is also important to evaluate functional connectivity across different taxa at fine scales in order to advance our understanding of whether structural connectivity in the Cerrado can be used as a proxy for species movement across the landscape.

Protection status estimated in this study are based on the remaining habitat extent and should thus be interpreted with caution. Information regarding the original extent of each habitat and physiognomic type is unavailable for our study site and not possible to be estimated at the time this study was conducted. Additional studies focusing on estimating the conservation value of priority sites and their habitat quality are important ways to complement our protection estimates.

The large difference in spatial resolution between our priority sites and land-use projections acquired for this study (i.e. 30m and 500m, respectively) represent a source of uncertainty in our analysis. In the absence of finer resolution projections, our results represent the best available estimates for conservation hotspots in the region. Fine-scale land use projections are urgently needed in the Cerrado for improving local vulnerability and land use planning estimates. We suggest that future studies should refine this analysis using finer resolution land use projections when available.

The Western Bahia region features one of the largest savanna coverage extents in the Cerrado (Alencar et al., 2020) and, at the same time, is under high threat of agricultural expansion (Salmona et al. 2016). Future studies examining the entire Matopiba region are crucial to investigate and improve opportunity assessments for conservation strategies in Cerrado agricultural landscapes. Understanding how conservation priorities are related to land tenure is also a top concern to prioritize land management and implementation actions.

4.5 Conclusion

This study demonstrates that additional conservation efforts within Cerrado private agricultural lands are important for complementing habitat representation and increasing landscape connectivity beyond efforts implemented by current policies (i.e. Forest Code sites). Our approach of combining a formal conservation priority-setting scheme with landscape connectivity conservation yielded new insights regarding conservation in agricultural lands. Agricultural landscapes tend to be neglected in most conservation prioritization exercises given their profitable nature and also because they lack large and well-connected blocks of natural vegetation. However, their remaining vegetation patches provide important habitat for several endangered and endemic species and have a key role in maintaining regional structural connectivity between large protected areas.

In commodity-driven landscapes such as the western Bahia region, a combination of strategies such as zero-deforestation commitments in soybean production, adopting sustainable land use practices (i.e. direct seeding), and implementing additional conservation measures are crucial for maintaining biodiversity and landscape connectivity beyond the Forest Code policy. Landowners can benefit from conservation strategies in private agricultural landscapes through the provision of key ecosystem services, which can increase their agricultural production. They may also receive explicit economic returns for meeting international market demands for sustainable production, especially for commodity crops. With most of the remaining Cerrado vegetation located in private lands, Cerrado landowners have an opportunity to lead such an effort and become a global example of reconciling conservation and agricultural production.

Acknowledgments

This research was funded by the Coordenação de Aperfeiçoamento de Pessoal de Nível Superior (CAPES) Foundation, Ministry of Education of Brazil, through the Science Without Borders program. We would like to thank Braulio Ferreira Dias for valuable and insightful feedback on this manuscript; Ramiro Crego for providing assistance in the landscape connectivity models; Peter Leimgruber and the Smithsonian Conservation Biology Institute for collaborating with resources to produce this research.

Chapter 5. Conclusions

The goal of this study is to contribute to the understanding of strategies to improve conservation and sustainability efforts in the Neotropical savannas of Brazil through fine-scale geospatial analysis. In Chapter 2, I developed a systematic GEOBIA framework to map Cerrado physiognomic types at fine spatial scale. I found that GEOBIA is a robust method to differentiate a wide range of heterogeneous Cerrado physiognomic types. The GEOBIA framework proposed using spectral and spatial features was effective to map 13 land cover categories with an 87.6% overall accuracy. Results demonstrated that 5-meter spatial resolution imagery is adequate for mapping land cover types of savanna structural elements. The GEOBIA framework is essential for refining land cover categories to ecological classes (physiognomic types), leading to a higher number of vegetation classes while improving overall accuracy. To the best of our knowledge, this is the first map to feature a wide range of detailed physiognomic types with high map accuracy at high spatial resolution.

In Chapter 3, I found that Legal Reserves and Areas of Permanent Preservation are both essential for ensuring protection of a wide diversity of physiognomic types, including the endangered Tropical Dry Forests, and essential habitats for endemic and endangered species. I demonstrated that Legal Reserves can be allocated to maximize biodiversity by considering the representativeness of unprotected physiognomic types from Areas of Permanent Preservation. This result can be used to improve the Forest Code policy guidelines for allocation of mandatory set asides, which should include further engagement with stakeholders at the municipality level following policy specifications for land parcel categories. Moreover, the results suggest that land property size might be a reliable indicator to target illegal land clearing within areas under Forest Code regulation. Thus, efforts to enforce compliance with the Forest Code can be improved by targeting land property size.

In Chapter 4, I demonstrated that combining a formal conservation priority-setting scheme with landscape connectivity conservation yielded to new insights regarding conservation in Cerrado private agricultural lands. The results suggest that unprotected vegetation in Cerrado private lands is critical to maintaining regional structural connectivity between large protected areas (i.e. national parks, ecological stations). I found that additional conservation set asides are important for complementing habitat representation and increasing habitat protection and landscape connectivity beyond efforts implemented by current policies (i.e. Forest Code sites). Landowners can benefit from conservation strategies in private lands through the provision of key ecosystem services, by obtaining payment incentives for zero-deforestation agriculture, and receiving expressive economic returns for meeting international market demands for sustainable production. Thus, conservation in private lands represents an opportunity to reconcile conservation and agricultural production.

5.1 Future Research

The results of my dissertation research present an opportunity to improve our understanding of several topics in Cerrado ecology and conservation. In Chapter 2, I proposed a systematic GEOBIA framework for mapping Cerrado physiognomic types at high spatial resolution. Fine-scale maps of Cerrado physiognomic types are necessary to improve the understanding of species habitat requirements and conditions, as well as our ability to assess ecosystem services and biodiversity. They also provide improved inputs for species suitability modeling, fire risk modeling, carbon accounting, landscape restoration, land-use management, and conservation planning.

This GEOBIA framework can be immediately tested using other publicly available multispectral imagery such as the Sentinel-2 Multispectral Imager (MSI) data, which is promising to improve discrimination of Cerrado physiognomic types due to its combination of fine spatial and spectral properties and larger areal coverage allowing for regional scale analysis. Although our framework was able to map 15 land cover categories, additional work is necessary to differentiate classes of similar structure for which edaphic conditional drivers were not available in our dataset. For instance, Tropical Dry Forests were not differentiated from Sclerophyll Forests located on flat terrain; Open Savannas were not differentiated from Rocky Savannas; and Savannas were not differentiated from denser Caatinga (a deciduous xerophyte type) enclaves. A potential solution for overcoming these remaining class issues would be the availability of a detailed (<1:10,000) soil types map or a combination of LiDAR and hyperspectral imagery. Future research needs to be directed towards expanding the proposed GEOBIA framework to other Cerrado ecoregions featuring additional physiognomic types not present in the test sites used in this study. The framework also has great potential to be adapted to other tropical savannas, depending on ancillary data availability. Moreover, we believe there is a need for adapting this framework using open source software to improve its accessibility. Future studies should investigate other open source segmentation algorithms and create geospatial rules in open source platforms such as a consolidated script in Python, Google Earth Engine, or RStudio.

Regional maps of vegetation physiognomic types can be used as biodiversity surrogates and are necessary for monitoring land use policy compliance, effectiveness, and planning. In Chapter 3, I developed a fine-scale spatially explicit biodiversity gap analysis, assessed the potential compliance status of rural properties with the Forest Code policy by land property size, and proposed a quantitative approach to support the Forest Code implementation at the landscape scale. The results from this chapter can be used to improve policy guidelines for an optimized Legal Reserve allocation considering representativeness of biodiversity. To advance our understanding of the contribution of the Forest Code policy to biodiversity and how to improve landowner compliance, future research should employ the methods used in this chapter to the entire Matopiba region and link results to information on supply chains, exporters, and consumers. Moreover, significant information regarding biodiversity within and outside Cerrado protected areas is still unknown. A combination of fine-scale physiognomic type maps, species occurrences and richness, and habitat suitability maps can be used in a gap analysis to estimate species and habitat protection status for individual protected areas. Species-specific habitat suitability maps should also be improved in future studies by validating with species occurrence data, where possible. Additional habitat suitability maps are also needed for other vertebrates, such as mammals, of which there are several endangered species in the Cerrado.

The results of this dissertation can assist in the creation of new public policies supporting conservation and sustainability in the Cerrado. In Chapter 4, I demonstrated that additional conservation efforts within Cerrado private agricultural lands are critical for guaranteeing biodiversity and connectivity conservation and are essential to complement efforts implemented by current policies. Future research should expand methods used in this chapter to the entire Matopiba region accounting for information on individual land parcels to estimate the conservation potential of unprotected lands at the regional scale. Results from this chapter reveal the benefits of alternative targets for biodiversity and connectivity conservation. Upcoming studies should test the feasibility of setting targets varying accordingly to each municipality and their compliance with the Forest Code policy. Moreover, such studies should incorporate datasets on species occurrence and richness, when possible, to calibrate and validate models.

A. Appendix: Supplementary materials for Chapter 2: Geographic Object-Based Image Analysis framework for mapping vegetation physiognomic types at fine scales in Neotropical savannas

Table A.1. RapidEye tiles, acquisition dates, and sensor viewing angles, for each study site.

Taquara watershed site

RapidEye tile	Date (mm/dd/yyyy)	sensor viewing angle
2331702	08/11/2013	+3.4°

Western Bahia site

RapidEye tile	Date (mm/dd/yyyy)	sensor viewing angle
2333511, 2333512, 2333611, 2333612	08/11/2011	-3.2°
2333311, 2333312, 2333411, 23333412	10/07/2011	+6.6°
2333313, 2333413, 2333513, 2333514, 2333613, 2333614	06/25/2011	+6.5°
2333414	09/13/2011	+6.7°
2333314	09/16/2011	+0.3°

Table A.2. Number of training data (polygons) collected for each image and study site.

Classes	Number of training data collected					
	Tile 2333314	Tile 2333414	June mosaic	August mosaic	October mosaic	Taquara Watershed
Closed-canopy	84	76	250	72	94	97
Dense shrub	90	91	232	110	86	97
Herbaceous (wet)	11	-	96	69	74	11
Herbaceous (dry)	-	-	-	-	-	35
Herbaceous	-	-	-	-	-	26
Open Canopy	43	57	123	60	78	33
Open shrub	80	82	232	100	52	95
Scrub-herb	-	-	88	41	-	-
Shrub-herb	24	66	100	49	41	-
Soil, NPV, impervious (bright)	87	72	128	57	78	118
Soil, NPV, impervious (dark)	111	110	205	93	126	88
Water	70	46	88	49	34	-
Total	600	600	1,542	700	675	600

Table A.3. Error matrix (reported as ha), overall accuracy, producer's accuracy, and user's accuracy, Taquara watershed Level 1 classification. The number in parentheses corresponds to the number of testing samples used for validation.

	Closed canopy (50)	Dense shrub (95)	Herbaceous (51)	Herbaceous (dry) (49)	Herbaceous (wet) (52)	Open canopy (101)	Open shrub (60)	Soil, NPV, impervious (bright) (34)	Soil, NPV, impervious (dark) (33)
Closed canopy	**3.9**	0.1	0.2	0.0	0.2	0.3	0.0	0.0	0.0
Dense shrub	0.0	**37.1**	0.0	0.0	0.0	0.7	3.5	0.0	0.0
Herbaceous	0.1	0.0	**5.1**	0.0	0.0	0.1	0.0	0.0	0.0
Herbaceous (dry)	0.0	0.0	0.0	**7.2**	0.0	0.0	1.0	0.0	0.0
Herbaceous (wet)	0.0	0.9	0.3	0.0	**3.5**	0.1	0.6	0.0	0.1
Open canopy	0.3	1.7	0.0	0.0	0.2	**14.0**	0.1	0.0	0.0
Open shrub	0.0	2.3	0.0	0.6	0.0	0.0	**16.0**	0.0	0.1
Soil, NPV, impervious (bright)	0.0	0.1	0.0	0.0	0.0	0.0	0.0	**3.6**	0.9
Soil, NPV, impervious (dark)	0.0	1.0	0.0	0.0	0.0	0.0	0.6	0.2	**5.4**
Overall accuracy (%)	85.5								
Producer's accuracy (%)	90.7	86.1	90.9	92	90	92	73.4	95	82.8
User's accuracy (%)	84.1	89.7	97.2	87.7	64	85.9	84	78.2	75.3

Table A.4. Error matrix (reported as ha), overall accuracy, producer's accuracy, and user's accuracy, Taquara watershed Level 2 classification. The number in parenthesis corresponds to the number of testing samples used for validation.

	Cerrado Woodland (50)	Grassland (50)	Invasive (50)	Marsh (50)	Non-vegetated/ Barren (69)	Open savanna (60)	Palm swamp (50)	Riparian forest (50)	Savanna (96)	Shrub swamp (0)
Cerrado woodland	**11.6**	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.0	0.0
Grassland	0.0	**7.2**	0.1	0.0	0.0	1.0	0.0	0.0	0.0	0.0
Invasive	0.0	0.0	**5.0**	0.0	0.0	0.0	0.0	0.1	0.0	0.0
Marsh	0.1	0.0	0.3	**3.5**	0.0	0.0	0.0	0.0	0.3	1.1
Non-vegetated/barren	0.0	0.0	0.1	0.0	**10.2**	0.6	0.0	0.0	1.0	0.0
Open savanna	0.0	0.6	0.0	0.0	0.1	**16.0**	0.0	0.0	2.3	0.0
Palm swamp	0.4	0.0	0.0	0.2	0.0	0.0	**2.0**	0.4	0.0	0.4
Riparian forest	0.3	0.0	0.2	0.2	0.0	0.0	0.0	**3.9**	0.1	0.0
Savanna	0.7	0.0	0.0	0.0	0.0	3.5	0.0	0.0	**37.5**	0.0
Overall accuracy (%)	86.4									
Producer's accuracy (%)	88.5	92	88.6	90	98.9	75.6	100	88.1	88.8	
User's accuracy (%)	92.3	86.4	97.2	66	85.4	84	59	84.1	89.8	

Table A.5. Error matrix (reported as count of polygons), overall accuracy, producer's accuracy, and user's accuracy, Taquara watershed Level 1 classification. The number in parentheses corresponds to the number of testing samples used for validation.

	Closed canopy (50)	Dense shrub (95)	Herbaceous (51)	Herbaceous (dry) (49)	Herbaceous (wet) (52)	Open canopy (101)	Open shrub (60)	Soil, NPV, impervious (bright) (34)	Soil, NPV, impervious (dark) (33)
Closed canopy	**41**	1	3	0	2	3	0	0	0
Dense shrub	0	**88**	0	0	0	2	5	0	0
Herbaceous	1	0	**49**	0	0	1	0	0	0
Herbaceous (dry)	0	0	0	**43**	0	0	6	0	0
Herbaceous (wet)	0	7	4	0	**35**	1	4	0	1
Open canopy	6	11	0	0	1	**81**	2	0	0
Open shrub	0	4	0	3	0	0	**52**	0	1
Soil, NPV, impervious (bright)	0	1	0	1	0	0	0	**28**	4
Soil, NPV, impervious (dark)	0	3	0	0	0	0	4	2	**24**
Overall accuracy (%)	84								
Producer's accuracy (%)	85.4	76.5	87.5	91.5	92.1	92	71.2	93.3	80
User's accuracy (%)	82	92.6	96.1	87.8	67.3	80.2	86.7	82.4	72.7

Table A.6. Error matrix (reported as count of polygons), overall accuracy, producer's accuracy, and user's accuracy, Taquara watershed Level 2 classification. The number in parentheses corresponds to the number of testing samples used for validation.

	Cerrado Woodland (50)	Grassland (50)	Invasive (50)	Marsh (50)	Non-vegetated/ Barren (69)	Open savanna (60)	Palm swamp (50)	Riparian forest (50)	Savanna (96)	Shrub swamp (0)
Cerrado woodland	**45**	0	0	0	0	0	0	0	5	0
Grassland	0	**43**	1	0	0	6	0	0	0	0
Invasive	0	0	**48**	0	0	0	0	2	0	0
Marsh	1	0	3	**35**	0	0	0	0	3	8
Non-vegetated/barren	0	1	1	0	**59**	4	0	0	4	0
Open savanna	0	3	0	0	1	**52**	0	0	4	0
Palm swamp	3	0	0	1	0	0	**32**	7	0	7
Riparian forest	3	0	3	2	0	0	0	**41**	1	0
Savanna	2	0	0	0	0	5	0	0	**89**	0
Overall accuracy (%)	84.6									
Producer's accuracy (%)	83.3	91.5	85.7	92.1	98.3	77.6	100	82	84	
User's accuracy (%)	90.0	86	96	70	85.5	86.7	64	82	92.7	

Table A.7. Error matrix (reported as count of polygons), overall accuracy, producer's accuracy, and user's accuracy, Western Bahia Level 1 classification. The number in parentheses corresponds to the number of testing samples used for validation.

	Closed canopy (125)	Dense shrub (90)	Herbaceous (wet) (50)	Open canopy (125)	Open shrub (100)	Scrub-shrub (59)	Shrub-herb (50)	Soil, NPV, impervious (bright) (30)	Soil, NPV, impervious (dark) (46)	Water (50)
Closed canopy	**102**	2	0	17	0	4	0	0	0	0
Dense shrub	0	**64**	1	7	14	2	2	0	0	0
Herbaceous (wet)	0	0	**46**	1	3	0	0	0	0	0
Open canopy	7	11	4	**101**	0	2	0	0	0	0
Open shrub	0	8	7	0	**78**	1	6	0	0	0
Scrub-shrub	10	0	0	0	4	**44**	1	0	0	0
Shrub-herb	0	0	0	0	6	0	**42**	2	0	0
Soil, NPV, impervious (bright)	0	0	0	0	0	0	3	**16**	11	0
Soil, NPV, impervious (dark)	0	0	0	0	0	0	6	1	**39**	0
Water	0	0	5	0	0	0	0	0	0	**45**
Overall accuracy (%)	79.6									
Producer's accuracy (%)	85.7	75.3	73	80.2	74.3	83	70	84.2	78	100
User's accuracy (%)	81.6	71.1	92	80.8	78	74.6	84	53.3	84.8	90

Table A.8. Error matrix (reported as count of polygons), overall accuracy, producer's accuracy, and user's accuracy, Western Bahia Level 2 classification. Number in parentheses corresponds to the number of testing samples used for validation.

	Cerrado woodland (20)	Marsh (20)	Non-vegetated/ Barren (78)	Open savanna (74)	Palm swamp (20)	Riparian forest (20)	Savanna (75)	Scrub cerrado (20)	Sd-forest (20)	Shrub swamp (20)	Shrubby grassland (23)	Tropical dry forest (20)	Water (20)
Cerrado woodland	**40**	0	0	0	0	3	3	2	2	0	0	0	0
Marsh	0	**46**	0	0	1	0	0	0	0	3	0	0	0
Non-vegetated/Barren	0	0	**67**	0	0	0	0	0	0	0	9	0	0
Open savanna	0	0	0	**61**	0	0	6	1	0	0	6	0	0
Palm swamp	1	4	0	0	**37**	3	0	0	0	5	0	0	0
Riparian forest	3	0	0	0	0	**44**	0	0	0	3	0	0	0
Savanna	1	0	0	11	0	0	**58**	0	3	0	2	0	0
Scrub cerrado	0	0	0	4	0	0	0	**36**	9	0	1	0	0
Sd-forest	4	0	0	0	0	0	1	1	**44**	0	0	0	0
Shrub swamp	1	8	0	0	2	1	1	0	0	**37**	0	0	0
Shrubby grassland	0	0	2	6	0	0	0	0	0	0	**42**	0	0
Tropical dry forest	3	0	0	0	0	0	4	0	0	0	0	**43**	0
Water	0	5	0	0	0	0	0	0	0	0	0	0	**45**
Overall accuracy (%)	82.8												
Producer's accuracy (%)	75.5	73	97.1	74.4	92.5	86.3	79.5	90	75.9	77.1	70	100	100
User's accuracy (%)	80	92	88.2	82.4	74	88	77.3	72	88	74	84	86	90

Table A.3. Estimate of landscape composition for the Taquara site considering the proportion of the mapped area of each physiognomic type, reported in percentage, with respect to the total mapped area (i.e. entire study site extent).

	Total (%)
Cerrado Woodland	4.9
Clouds	3.4
Grassland	3.4
Invasive forbs	0.9
Marsh	0.7
Non-natural / Barren	23.9
Open savanna	20.6
Palm swamp	0.6
Riparian forest	4.2
Savanna	33.2
Semi-deciduous forest	0.3
Shade	1.8
Shrub swamp	0.3
Crop, orchard, and silviculture mask	1.7
Water	0.03

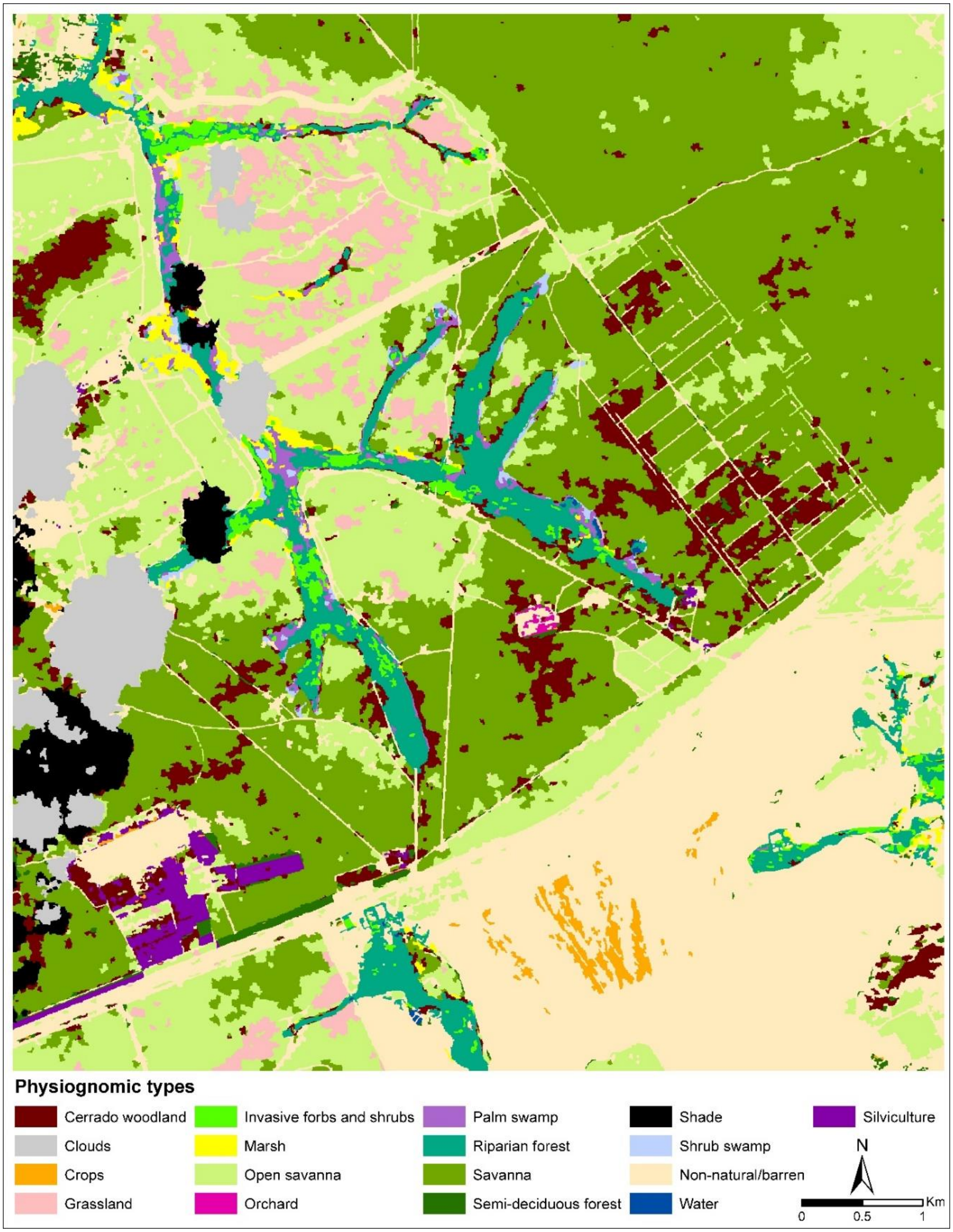

Figure A.1. Map of vegetation physiognomic types (Level 2) for the Taquara watershed study site

B. Appendix: Supplementary materials for Chapter 3: Is the Brazilian Forest Code protecting Neotropical savanna habitats? A fine-scale gap analysis of a Brazilian Cerrado agricultural landscape

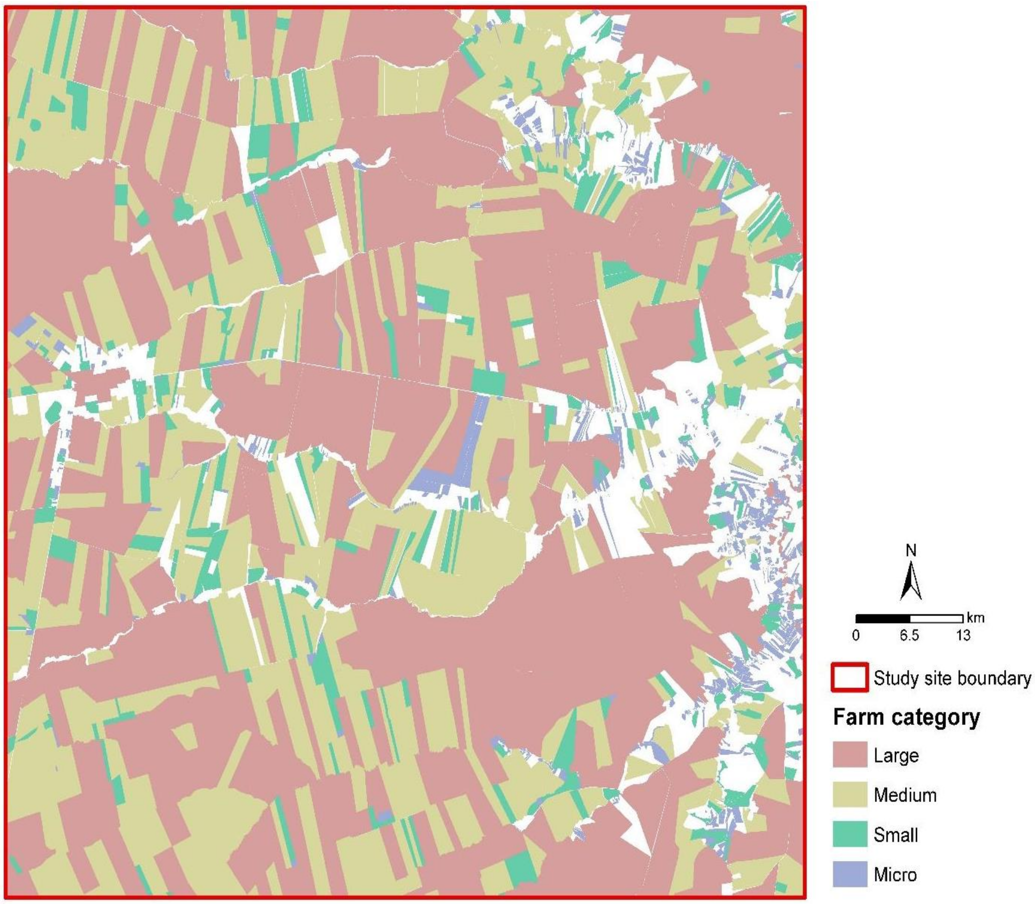

Figure B.1. Map showing the spatial distribution of farm size categories in the study region.

Table B.1. Detailed description of species-specific habitat suitability models.

Species (Family)	Home Range	Habitat	Life Habits	Description
Myiothlypis leucophrys (Parulidae)	2 ha (Marini and Cavalcanti, 1993)	Riparian forest (seasonally or permanently wet)	Primary habitat	Area equal or larger than home range
		Riparian forest (upland)	Suitable habitat	Area equal or larger than home range
		Riparian forest (wet and upland)	Marginal habitat	Area lower than home range
		Wetlands, Tropical Dry Forest, and Semi-deciduous Forest	Marginal habitat	Located within 100m buffer from primary and suitable habitats
Neothraupis fasciata	3.7 ha (Duca and Marini, 2014)	Open Savanna, Savanna	Primary habitat	Area equal or larger than home range

Species (Family)	Home Range	Habitat	Life Habits	Description
(Thraupidae)		Woodland, Shrubby Grassland	Suitable habitat	Area equal or larger than home range
		Open Savanna, Savanna, Woodland, Shrubby Grassland	Marginal habitat	Area lower than home range
Melanopareia torquata (Melanopareiidae)	0.78 ha (Kanegae et al., 2012)	Shrubby Grassland, Open Savanna	Primary habitat	Area equal or larger than home range
		Savanna	Suitable habitat	Area equal or larger than home range
		Shrubby Grassland, Open Savanna, Savanna	Marginal habitat	Area lower than home range
		Marsh	Marginal habitat	
Herpsilochmus longirostris (Thamnophilidae)	0.91 ha (Kennedy et al., 2016b)	Riparian Forest	Primary habitat	Area equal or larger than home range
		Tropical Dry Forest, Semi-deciduous Forest	Suitable habitat	Area within 60m to riparian forest, not accounting for home range. This is to increase home range foraging area (Tubelis et al., 2014)
		Riparian Forest	Marginal habitat	Area lower than home range
		Tropical Dry Forest, Semi-deciduous Forest	Marginal habitat	Adjacent (100m) to primary riparian habitat (40m from the areas classified as suitable)
		Marsh	Marginal habitat	Adjacent (100m) to primary riparian habitat
Syndactyla dimidiata (Furnariidae)	2.73 ha (Kennedy et al., 2016b)	Riparian forest (seasonally or permanently wet)	Primary habitat	Area equal or larger than home range
		Riparian forest (upland)	Suitable habitat	Area equal or larger than home range
		Tropical Dry Forest, Semi-deciduous Forest	Suitable habitat	Area within 60m to riparian forest, not accounting for home range. This is to increase home range foraging area (Tubelis et al., 2014)
		Palm Swamp, Shrub Swamp	Suitable habitat	Area equal or larger than home range
		Riparian forest (wet and upland), Palm Swamp, Shrub Swamp	Marginal habitat	Area lower than home range

Species (Family)	Home Range	Habitat	Life Habits	Description
Syndactyla dimidiata (Furnariidae)	2.73 ha (Kennedy et al., 2016b)	Tropical Dry Forest, Semi-deciduous Forest, Savanna, Woodland	Marginal habitat	Adjacent (100m) to primary riparian habitat
Euscarthmus rufomarginatus (Tyrannidae)	0.34 ha (Kennedy et al., 2016b)	Shrubby Grassland, Open Savanna	Primary habitat	Area equal or larger than home range
		Savanna	Suitable habitat	Area equal or larger than home range
		Shrubby Grassland, Open Savanna, Savanna	Marginal habitat	Area lower than home range
Cypsnagra hirundinacea (Thraupidae)	0.20 ha (Kennedy et al., 2016b)	Shrubby Grassland, Open Savanna	Primary habitat	Area equal or larger than home range
		Savanna	Suitable habitat	Area equal or larger than home range
		Shrubby Grassland, Open Savanna, Savanna	Marginal habitat	Area lower than home range
Anodorhynchus hyacinthinus (Psittacidae)	3.45 ha (Kennedy et al., 2016b)	Palm Swamp	Primary habitat	Area equal or larger than home range
		Riparian Forest	Suitable habitat	Did not account for home range
		Escarpments	Suitable habitat	Areas classified as "shade" with slope > 45% (mountainous terrain) could be escarpment cavities used for breeding
		Open Savanna	Suitable habitat	Area equal or larger than home range
		Shrubby Grassland	Suitable habitat	Area equal or larger than home range
		Shrubby Swamp, Marsh	Marginal habitat	Did not account for home range
		Palm Swamp, Shrubby Grassland, Open Savanna	Marginal habitat	Area lower than home range
Lepidocolaptes wagleri (Dendrocolaptidae)	2.65 ha (Kennedy et al., 2016b)	Tropical Dry Forest, Semi-deciduous Forest	Primary habitat	Area equal or larger than home range
		Riparian Forest	Suitable habitat	Area equal or larger than home range; Adjacent 60m to primary habitats
		Tropical Dry Forest, Semi-deciduous Forest, Riparian Forest	Marginal habitat	Area lower than home range
		Cerrado Woodland	Marginal habitat	Adjacent 100m to primary habitats

Table B.2. Weight given to biodiversity features (land cover classes) used in the *Zonation* analysis

Biodiversity features	Weights
Semi-deciduous forest, Open savanna, Savanna, Cerrado Woodland, Scrub Cerrado	4.0
Riparian Forest, Tropical Dry Forest, Marsh, Palm Swamp, Shrub Swamp, Water	2.0
Shrubby grassland	1.0
Non-natural/barren, Silviculture	-1.0
Paved roads, Farming, Crop pivots, Urban areas	-2.0

Table B.3. Proportion of physiognomic types protected by Legal Reserves (LRs) and Areas of Permanent Preservation (APPs).

Classes	LRs – Area (ha)	APPs – Area (ha)	LRs – Representation (%)	APPs – Representation (%)
Barren	5,977	3,432	4.4%	3.8%
Cerrado Woodland	7,703	5,662	5.7%	6.3%
Farming	10,165	7,482	7.5%	8.4%
Marsh	12.40	10,787	0.0%	12.1%
Open Savanna	50,766	12,100	37.5%	13.5%
Palm Swamp	-	7,059	-	7.9%
Irrigated Crops	57	68	0.0%	0.1%
Riparian Forest	228	7,857	0.2%	8.8%
Savanna	50,259	19,880	37.1%	22.2%
Scrub Cerrado	2,004	1,810	1.5%	2.0%
Semi-deciduous Forest	1,325	240	1.0%	0.3%
Shade	57	646	0.0%	0.7%
Shrub Swamp	-	4,900	-	5.5%
Shrubby Grassland	6,422	1,290	4.7%	1.4%
Silviculture	84	8	0.1%	0.0%
Tropical Dry Forest	449	3,592	0.3%	4.0%
Urban Areas	-	59	0.0%	0.1%
Water	3	2,515	0.0%	2.8%
Total	135,523	89,401		

Table B.4. Protection status considering only the species primary habitat and its proportion within categories of Areas of Permanent Preservation (APPs) and Legal Reserves (LRs). The largest amount of primary habitat protection is highlighted in bold.

	Hydromorphic Soil APP (%)	Plateau APP (%)	10m Stream APP (%)	50m Stream APP (%)	LRs (%)	Total protection of primary habitats (%)
Anodorhynchus hyacinthinus	**78.8%**	0.0%	15.1%	6.1%	0.0%	100.0%
Cypsnagra hirundinacea	1.3%	2.4%	1.8%	0.1%	**26.0%**	31.7%
Euscarthmus rufomarginatus	1.3%	2.4%	1.7%	0.1%	**26.2%**	31.7%
Herpsilochmus longirostris	**43.5%**	1.9%	28.1%	9.9%	1.7%	85.1%
Lepidocolaptes Wagleri	0.2%	**19.2%**	4.8%	0.0%	12.4%	36.6%
Melanopareia torquata	1.2%	2.4%	1.7%	0.1%	**26.4%**	31.7%
Myiothlypis leucophrys	**65.1%**	0.0%	22.0%	12.9%	0.0%	100.0%
Neothraupis fasciata	1.0%	4.1%	2.2%	0.1%	**25.9%**	33.2%
Syndactyla dimidiata	**65.4%**	0.0%	21.9%	12.7%	0.0%	100.0%

Table B.5. Number of rural properties in the study area separated into size categories (large, medium, small, and micro), the total area (in hectares) by property size category, as well as average and standard deviation of area (in hectares) occupied by rural properties according to their size category.

Property size category	Number of properties	Total area (ha)	Average of property area (ha)	Standard Deviation of property area (ha)
Large	212	385,504	1,818	1,182.8
Medium	504	268,413	533	193.3
Small	333	52,294	157	56.1
Micro	3,762	21,814	6	11.1
Total	4,811	728,025	153	475.4

C. Appendix: Supplementary materials for Chapter 4: Private lands as opportunities for improving biodiversity and connectivity conservation across Cerrado protected areas

Table C.1. Conservation status of physiognomic types estimated for Alternative scheme 1, for each landscape fraction. Fractions top 20% and top 30% represents the cumulative percentages for all landscape fractions (e.g. top 30% includes both top 20% and top 10% fractions of the landscape).

Landscape fraction (cumulative)	Vegetation and land use types	Area (ha)	Conservation Status	Total Conservation Status (Zonation + Forest Code sites)
Top 10%	Barren	367	1%	-
Top 10%	Cerrado Woodl:	14,398	40%	76%
Top 10%	Farming	128	0%	-
Top 10%	Open Savanna	10,911	6%	39%
Top 10%	Riparian Veget:	1,324	14%	96%
Top 10%	Savanna	12,287	6%	39%
Top 10%	Scrub Cerrado	6,515	59%	93%
Top 10%	Semi-deciduous Forest	6,842	80%	99%
Top 10%	Shrubby Grassl:	14,590	45%	69%
Top 10%	Tropical Dry Forest	1,165	21%	92%
Top 20%	Barren	1,130	3%	-
Top 20%	Cerrado Woodl:	19,207	53%	89%
Top 20%	Farming	252	0%	-
Top 20%	Open Savanna	37,955	20%	53%
Top 20%	Riparian Veget:	1,333	14%	96%
Top 20%	Savanna	43,255	21%	54%
Top 20%	Scrub Cerrado	6,632	60%	94%
Top 20%	Semi-deciduous Forest	6,850	80%	99%
Top 20%	Shrubby Grassl:	19,085	59%	83%
Top 20%	Tropical Dry Forest	1,168	21%	92%
Top 30%	Barren	2,230	6%	-
Top 30%	Cerrado Woodl:	19,704	54%	91%
Top 30%	Farming	404	0%	-
Top 30%	Open Savanna	60,461	32%	65%
Top 30%	Riparian Veget:	1,338	14%	96%
Top 30%	Savanna	87,221	41%	93%

Landscape fraction (cumulative)	Vegetation and land use types	Area (ha)	Conservation Status	Total Conservation Status (Zonation + Forest Code sites)
Top 30%	Scrub Cerrado	6,655	60%	94%
Top 30%	Semi-deciduous Forest	6,855	81%	99%
Top 30%	Shrubby Grassla	19,645	61%	84%
Top 30%	Tropical Dry Forest	1,169	21%	92%

Table C.2. Conservation status of potential habitats for endemic and endangered avifauna estimated for Scenario 1, for each landscape fraction. Fractions top 20% and top 30% represents the cumulative percentages for all landscape fractions (e.g. top 30% includes both top 20% and top 10% fractions of the landscape). Total efforts in the last column represent the sum of conservation status (%) of the current conservation efforts in the landscape (areas protected by land use regulation) and areas protected by our conservation strategy.

Landscape fraction (cumulative)	Species	Habitat types	Protection (ha)	Conservation status (%) - zonation; total efforts
Top 10%	*Anodorhynchus hyacinthinus*	*Total*	25,600	10.1; 49.6
Top 10%		Primary	-	
Top 10%		Suitable	22,277	
Top 10%		Marginal	3,322	
Top 10%	*Cypsnagra hyrundinacea*	*Total*	37,789	9.2; 41.4
Top 10%		Primary	24,387	
Top 10%		Suitable	11,658	
Top 10%		Marginal	1,744	
Top 10%	*Euscarthmus rufomarginatus*	*Total*	37,789	8.7; 41.2
Top 10%		Primary	23,930	
Top 10%		Suitable	11,318	
Top 10%		Marginal	2,540	
Top 10%	*Herpsilochmus longirostris*	*Total*	3,336	13.1; 91.8
Top 10%		Primary	943	
Top 10%		Suitable	89	
Top 10%		Marginal	2,305	

Landscape fraction (cumulative)	**Species**	**Habitat types**	**Protection (ha)**	**Conservation status (%) - zonation; total efforts**
Top 10%	*Lepidocolaptes wagleri*	*Total*	9,734	54.9; 95.3
Top 10%		Primary	7,025	
Top 10%		Suitable	33	
Top 10%		Marginal	2,676	
Top 10%	*Melanopareia torquata*	*Total*	37,814	8.5; 42.6
Top 10%		Primary	23,274	
Top 10%		Suitable	10,774	
Top 10%		Marginal	3,767	
Top 10%	*Myiothlypis leucophrys*	*Total*	1,444	8.2; 97.5
Top 10%		Primary	-	
Top 10%		Suitable	834	
Top 10%		Marginal	610	
Top 10%	*Neothraupis fasciata*	*Total*	52,187	11.1; 43.9
Top 10%		Primary	18,251	
Top 10%		Suitable	24,120	
Top 10%		Marginal	9,816	
Top 10%	*Syndactyla dimidiata*	*Total*	1,977	6.5; 94.3
Top 10%		Primary	-	
Top 10%		Suitable	63	
Top 10%		Marginal	1,913	
Top 20%	*Anodorhynchus hyacinthinus*	*Total*	57,145	22.6; 62.1
Top 20%		Primary	-	
Top 20%		Suitable	48,333	
Top 20%		Marginal	8,812	
Top 20%	*Cypsnagra hyrundinacea*	*Total*	100,295	24.3; 56.6
Top 20%		Primary	54,160	

Landscape fraction (cumulative)	Species	Habitat types	Protection (ha)	Conservation status (%) - zonation; total efforts
Top 20%		Suitable	41,958	
Top 20%		Marginal	4,178	
Top 20%	*Euscarthmus rufomarginatus*	*Total*	100,295	23.1; 55.6
Top 20%		Primary	52,952	
Top 20%		Suitable	41,262	
Top 20%		Marginal	6,081	
Top 20%	*Herpsilochmus longirostris*	*Total*	3,808	15.0; 93.6
Top 20%		Primary	947	
Top 20%		Suitable	89	
Top 20%		Marginal	2,772	
Top 20%	*Lepidocolaptes wagleri*	*Total*	9,836	55.5; 95.8
Top 20%		Primary	7,027	
Top 20%		Suitable	-	
Top 20%		Marginal	2,776	
Top 20%	*Melanopareia torquata*	*Total*	100,320	22.6; 56.7
Top 20%		Primary	51,168	
Top 20%		Suitable	40,150	
Top 20%		Marginal	9,003	
Top 20%	*Myiothlypis leucophrys*	*Total*	1,454	8.2; 97.5
Top 20%		Primary	-	
Top 20%		Suitable	837	
Top 20%		Marginal	617	
Top 20%	*Neothraupis fasciata*	*Total*	119,502	25.4; 58.2
Top 20%		Primary	70,494	
Top 20%		Suitable	27,103	
Top 20%		Marginal	21,905	

Landscape fraction (cumulative)	**Species**	**Habitat types**	**Protection (ha)**	**Conservation status (%) - zonation; total efforts**
Top 20%	*Syndactyla dimidiata*	*Total*	2,332	7.7; 95.5
Top 20%		Primary	-	
Top 20%		Suitable	64	
Top 20%		Marginal	2,268	
Top 30%	*Anodorhynchus hyacinthinus*	*Total*	80,215	31.7; 71.2
Top 30%		Primary	-	
Top 30%		Suitable	68,366	
Top 30%		Marginal	11,849	
Top 30%	*Cypsnagra hyrundinacea*	*Total*	167,327	40.6; 72.9
Top 30%		Primary	76,317	
Top 30%		Suitable	85,574	
Top 30%		Marginal	5,436	
Top 30%	*Euscarthmus rufomarginatus*	*Total*	167,327	38.6; 71.1
Top 30%		Primary	74,720	
Top 30%		Suitable	84,667	
Top 30%		Marginal	7,940	
Top 30%	*Herpsilochmus longirostris*	*Total*	3,962	15.6; 94.2
Top 30%		Primary	948	
Top 30%		Suitable	89	
Top 30%		Marginal	2,924	
Top 30%	*Lepidocolaptes wagleri*	*Total*	9,849	55.5; 95.9
Top 30%		Primary	7,029	
Top 30%		Suitable	-	
Top 30%		Marginal	2,787	
Top 30%	*Melanopareia torquata*	*Total*	167,353	37.7; 71.8
Top 30%		Primary	72,308	

Landscape fraction (cumulative)	**Species**	**Habitat types**	**Protection (ha)**	**Conservation status (%) - zonation; total efforts**
Top 30%		Suitable	83,187	
Top 30%		Marginal	11,858	
Top 30%	*Myiothlypis leucophrys*	*Total*	1,459	8.2; 97.5
Top 30%		Primary	-	
Top 30%		Suitable	838	
Top 30%		Marginal	620	
Top 30%	*Neothraupis fasciata*	*Total*	187,031	39.8; 72.6
Top 30%		Primary	132,617	
Top 30%		Suitable	27,332	
Top 30%		Marginal	27,082	
Top 30%	*Syndactyla dimidiata*	*Total*	2,445	8.1; 95.8
Top 30%		Primary	-	
Top 30%		Suitable	64	
Top 30%		Marginal	2,382	

References

Alencar, A., Z. Shimbo, J., Lenti, F., Balzani Marques, C., Zimbres, B., Rosa, M., Arruda, V., Castro, I., Fernandes Márcico Ribeiro, J.P., Varela, V., Alencar, I., Piontekowski, V., Ribeiro, V., M. C. Bustamante, M., Eyji Sano, E., Barroso, M., 2020. Mapping Three Decades of Changes in the Brazilian Savanna Native Vegetation Using Landsat Data Processed in the Google Earth Engine Platform. Remote Sensing 12, 924. https://doi.org/10.3390/rs12060924

Anantharaman, R., Hall, K., Shah, V., Edelman, A., 2019. Circuitscape in Julia: High Performance Connectivity Modelling to Support Conservation Decisions. arXiv:1906.03542 [q-bio].

Arroyo, L.A., Johansen, K., Armston, J., Phinn, S., 2010a. Integration of LiDAR and QuickBird imagery for mapping riparian biophysical parameters and land cover types in Australian tropical savannas. Forest Ecology and Management 259, 598–606. https://doi.org/10.1016/j.foreco.2009.11.018

Arroyo, L.A., Johansen, K., Armston, J., Phinn, S., 2010b. Integration of LiDAR and QuickBird imagery for mapping riparian biophysical parameters and land cover types in Australian tropical savannas. Forest Ecology and Management 259, 598–606. https://doi.org/10.1016/j.foreco.2009.11.018

Asner, G.P., Levick, S.R., Kennedy-Bowdoin, T., Knapp, D.E., Emerson, R., Jacobson, J., Colgan, M.S., Martin, R.E., 2009. Large-scale impacts of herbivores on the structural diversity of African savannas. PNAS 106, 4947–4952. https://doi.org/10.1073/pnas.0810637106

Assunção, Chiavari, 2015. Towards Efficient Land Use in Brazil. The New Climate Economy.

Azevedo, A.A., Rajão, R., Costa, M.A., Stabile, M.C.C., Macedo, M.N., Reis, T.N.P. dos, Alencar, A., Soares-Filho, B.S., Pacheco, R., 2017. Limits of Brazil's Forest Code as a means to end illegal deforestation. PNAS 114, 7653–7658. https://doi.org/10.1073/pnas.1604768114

Azevedo, A.A., Stabile, M.C.C., Reis, T.N.P., 2015. Commodity production in Brazil: Combining zero deforestation and zero illegality. Elem Sci Anth 3, 000076. https://doi.org/10.12952/journal.elementa.000076

Baatz, M., Schäpe, A., 2000. Multiresolution segmentation: an optimization approach for high quality multi-scale image segmentation. Presented at the Angewandte Geographische Informationsverarbeitung XII 58, Wichmann-Verlag, Heidelberg, pp. 12–23.

Bagno, M.A., Marinho-Filho, J., 2001. A avifauna do Distrito Federal: uso de ambientes abertos e florestais e ameaças, in: Cerrado: Caracterização e Recuperação de Matas de Galeria. Embrapa, Brasilia, DF, pp. 495–528.

Baldeck, C.A., Asner, G.P., 2013. Estimating Vegetation Beta Diversity from Airborne Imaging Spectroscopy and Unsupervised Clustering. Remote Sensing 5, 2057–2071. https://doi.org/10.3390/rs5052057

Baldeck, C.A., Colgan, M.S., Féret, J.-B., Levick, S.R., Martin, R.E., Asner, G.P., 2014. Landscape-scale variation in plant community composition of an African savanna from airborne species mapping. Ecological Applications 24, 84–93. https://doi.org/10.1890/13-0307.1

Banks-Leite, C., Pardini, R., Tambosi, L.R., Pearse, W.D., Bueno, A.A., Bruscagin, R.T., Condez, T.H., Dixo, M., Igari, A.T., Martensen, A.C., Metzger, J.P., 2014. Using ecological thresholds to evaluate the costs and benefits of set-asides in a biodiversity hotspot. Science 345, 1041–1045. https://doi.org/10.1126/science.1255768

Bastin, J.-F., Berrahmouni, N., Grainger, A., Maniatis, D., Mollicone, D., Moore, R., Patriarca, C., Picard, N., Sparrow, B., Abraham, E.M., Aloui, K., Atesoglu, A., Attore, F., Bassüllü, Ç., Bey, A., Garzuglia, M., García-Montero, L.G., Groot, N., Guerin, G., Laestadius, L., Lowe, A.J., Mamane, B., Marchi, G., Patterson, P., Rezende, M., Ricci, S., Salcedo, I., Diaz, A.S.-P., Stolle, F., Surappaeva, V., Castro, R., 2017. The extent of forest in dryland biomes. Science 356, 635–638. https://doi.org/10.1126/science.aam6527

Belote, R.T., Beier, P., Creech, T., Wurtzebach, Z., Tabor, G., 2020. A Framework for Developing Connectivity Targets and Indicators to Guide Global Conservation Efforts. Bioscience 70, 122–125. https://doi.org/10.1093/biosci/biz148

Beuchle, R., Grecchi, R.C., Shimabukuro, Y.E., Seliger, R., Eva, H.D., Sano, E., Achard, F., 2015. Land cover changes in the Brazilian Cerrado and Caatinga biomes from 1990 to 2010 based on a systematic remote sensing sampling approach. Applied Geography 58, 116–127. https://doi.org/10.1016/j.apgeog.2015.01.017

Bingham, H., Fitzsimons, J., A., Redford, K., Mitchell, B.A., Bezaury-Creel, J., Cumming, T.L., 2017. Privately protected areas: advances and challenges in guidance, policy and documentation. PARKS 23, 13–28. https://doi.org/10.2305/IUCN.CH.2017.PARKS-23-1HB.en

Blaschke, T., 2010. Object based image analysis for remote sensing.pdf. ISPRS Journal of Photogrammetry and Remote Sensing 65, 2–16.

Blaschke, T., Hay, G.J., Kelly, M., Lang, S., Hofmann, P., Addink, E., Queiroz Feitosa, R., van der Meer, F., van der Werff, H., van Coillie, F., Tiede, D., 2014. Geographic Object-Based Image Analysis – Towards a new paradigm. ISPRS Journal of Photogrammetry and Remote Sensing 87, 180–191. https://doi.org/10.1016/j.isprsjprs.2013.09.014

Boggs, G.S., 2010. Assessment of SPOT 5 and QuickBird remotely sensed imagery for mapping tree cover in savannas. International Journal of Applied Earth Observation and Geoinformation 12, 217–224. https://doi.org/10.1016/j.jag.2009.11.001

Bond, W.J., Parr, C.L., 2010. Beyond the forest edge: Ecology, diversity and conservation of the grassy biomes. Biological Conservation, Conserving complexity: Global change and community-scale interactions 143, 2395–2404. https://doi.org/10.1016/j.biocon.2009.12.012

Borges, F.J.A., Ribeiro, B.R., Lopes, L.E., Loyola, R., 2019. Bird vulnerability to climate and land use changes in the Brazilian Cerrado. Biological Conservation 236, 347–355. https://doi.org/10.1016/j.biocon.2019.05.055

Borrini-Feyerabend, G., Dudley, N., Jaeger, T., Lassen, B., Broome, N.P., Phillips, A., Sandwith, T., 2013. Governance of Protected Areas: From understanding to action (No. 20), Best Practice Protected Area Guidelines. IUCN, Glad, Switzerland.

Boscolo, D., Metzger, J.P., 2011. Isolation determines patterns of species presence in highly fragmented landscapes. Ecography 34, 1018–1029. https://doi.org/10.1111/j.1600-0587.2011.06763.x

Brannstrom, C., Jepson, W., Filippi, A.M., Redo, D., Xu, Z., Ganesh, S., 2008. Land change in the Brazilian Savanna (Cerrado), 1986–2002: Comparative analysis and implications for land-use policy. Land Use Policy 25, 579–595. https://doi.org/10.1016/j.landusepol.2007.11.008

Breiman, L., 2001. Random forests. Machine learning 45, 5–32.

Bridgewater, S., Ratter, J.A., Felipe Ribeiro, J., 2004. Biogeographic patterns, β-diversity and dominance in the cerrado biome of Brazil. Biodiversity and Conservation 13, 2295–2317. https://doi.org/10.1023/B:BIOC.0000047903.37608.4c

Brum, F.T., Pressey, R.L., Bini, L.M., Loyola, R., 2019. Forecasting conservation impact to pinpoint spatial priorities in the Brazilian Cerrado. Biological Conservation 240, 108283. https://doi.org/10.1016/j.biocon.2019.108283

Buckley, R., 2009. Evaluating the net effects of ecotourism on the environment: a framework, first assessment and future research. Journal of Sustainable Tourism 17, 643–672. https://doi.org/10.1080/09669580902999188

Butchart, S.H.M., Clarke, M., Smith, R.J., Sykes, R.E., Scharlemann, J.P.W., Harfoot, M., Buchanan, G.M., Angulo, A., Balmford, A., Bertzky, B., Brooks, T.M., Carpenter, K.E., Comeros-Raynal, M.T., Cornell, J., Ficetola, G.F., Fishpool, L.D.C., Fuller, R.A., Geldmann, J., Harwell, H., Hilton-Taylor, C., Hoffmann, M., Joolia, A., Joppa, L., Kingston, N., May, I., Milam, A., Polidoro, B., Ralph, G., Richman, N., Rondinini, C., Segan, D.B., Skolnik, B., Spalding, M.D., Stuart, S.N., Symes, A., Taylor, J., Visconti, P., Watson, J.E.M., Wood, L., Burgess, N.D., 2015. Shortfalls and Solutions for Meeting National and Global Conservation Area Targets. Conservation Letters 8, 329–337. https://doi.org/10.1111/conl.12158

Butchart, S.H.M., Walpole, M., Collen, B., Strien, A. van, Scharlemann, J.P.W., Almond, R.E.A., Baillie, J.E.M., Bomhard, B., Brown, C., Bruno, J., Carpenter, K.E., Carr, G.M., Chanson, J., Chenery, A.M., Csirke, J., Davidson, N.C., Dentener, F., Foster, M., Galli, A., Galloway, J.N., Genovesi, P., Gregory, R.D., Hockings, M., Kapos, V., Lamarque, J.-F., Leverington, F., Loh, J., McGeoch, M.A., McRae, L., Minasyan, A., Morcillo, M.H., Oldfield, T.E.E., Pauly, D., Quader, S., Revenga, C., Sauer, J.R., Skolnik, B., Spear, D., Stanwell-Smith, D., Stuart, S.N., Symes, A., Tierney, M., Tyrrell, T.D., Vié, J.-C., Watson, R., 2010. Global Biodiversity: Indicators of Recent Declines. Science 328, 1164–1168. https://doi.org/10.1126/science.1187512

Cardinale, B.J., Duffy, J.E., Gonzalez, A., Hooper, D.U., Perrings, C., Venail, P., Narwani, A., Mace, G.M., Tilman, D., Wardle, D.A., Kinzig, A.P., Daily, G.C., Loreau, M., Grace, J.B., Larigauderie, A., Srivastava, D.S., Naeem, S., 2012. Biodiversity loss and its impact on humanity. Nature 486, 59–67. https://doi.org/10.1038/nature11148

Carmignotto, A.P., Vivo, M., Langguth, A., 2012. Mammals of the Cerrado and Caatinga, in: Bones, Clones, and Biomes: The History and Geography of Recent Neotropical Mammals. University of Chicago Press.

Carranza, T., Balmford, A., Kapos, V., Manica, A., 2014. Protected Area Effectiveness in Reducing Conversion in a Rapidly Vanishing Ecosystem: The Brazilian Cerrado. Conservation Letters 7, 216–223. https://doi.org/10.1111/conl.12049

Carvalho, D.L. de, Sousa-Neves, T., Cerqueira, P.V., Gonsioroski, G., Silva, S.M., Silva, D.P., Santos, M.P.D., 2017. Delimiting priority areas for the conservation of endemic and threatened Neotropical birds using a niche-based gap analysis. PLOS ONE 12, e0171838. https://doi.org/10.1371/journal.pone.0171838

Chiarello, A.G., 2000. Conservation value of a native forest fragment in a region of extensive agriculture. Revista Brasileira de Biologia 60, 237–247. https://doi.org/10.1590/S0034-71082000000200007

Cho, M.A., Mathieu, R., Asner, G.P., Naidoo, L., van Aardt, J., Ramoelo, A., Debba, P., Wessels, K., Main, R., Smit, I.P.J., Erasmus, B., 2012. Mapping tree species composition in South African savannas using an integrated airborne spectral and LiDAR system. Remote Sensing of Environment 125, 214–226. https://doi.org/10.1016/j.rse.2012.07.010

Cochrane, M.A., 2003. Fire science for rainforests. Nature 421, 913–919. https://doi.org/10.1038/nature01437

Cole, M., 1986. The Savannas: Biogeography and Geobotany. Academic Press, London.

Colgan, M.S., Baldeck, C.A., Féret, J.-B., Asner, G.P., 2012. Mapping Savanna Tree Species at Ecosystem Scales Using Support Vector Machine Classification and BRDF Correction on Airborne Hyperspectral and LiDAR Data. Remote Sensing 4, 3462–3480. https://doi.org/10.3390/rs4113462

Congalton, R.G., 1991. A review of assessing the accuracy of classifications of remotely sensed data. Remote sensing of environment 37, 35–46.

Cord, A., Conrad, C., Schmidt, M., Dech, S., 2010. Standardized FAO-LCCS land cover mapping in heterogeneous tree savannas of West Africa. Journal of Arid Environments 74, 1083–1091. https://doi.org/10.1016/j.jaridenv.2010.03.012

Cortés Capano, G., Toivonen, T., Soutullo, A., Di Minin, E., 2019. The emergence of private land conservation in scientific literature: A review. Biological Conservation 237, 191–199. https://doi.org/10.1016/j.biocon.2019.07.010

Coulter, L.L., Stow, D.A., Tsai, Y.-H., Ibanez, N., Shih, H., Kerr, A., Benza, M., Weeks, J.R., Mensah, F., 2016. Classification and assessment of land cover and land use change in southern Ghana using dense stacks of Landsat 7 ETM+ imagery. Remote Sensing of Environment 184, 396–409. https://doi.org/10.1016/j.rse.2016.07.016

Coutinho, L.M., 1978. O conceito de Cerrado. Revista Brasileira de Botânica 11, 17–23.

Crippen, R., Buckley, S., Agram, P., Belz, E., Gurrola, E., Hensley, S., Kobrick, M., Lavalle, M., Martin, J., Neumann, M., Nguyen, Q., Rosen, P., Shimada, J., Simard, M., Tung, W., 2016. NASADEM Global Elevation Model: Methods and Progress. ISPRS - International Archives of the Photogrammetry, Remote Sensing and Spatial Information Sciences XLI-B4, 125–128. https://doi.org/10.5194/isprsarchives-XLI-B4-125-2016

Dave, R., Saint-Laurent, C., Murray, L., Antunes Daldegan, G., Brouwer, R., de Mattos Scaramuzza, C.A., Raes, L., Simonit, S., Catapan, M., García Contreras, G., Ndoli, A., Karangwa, C., Perera, N., Hingorani, S., Pearson, T., 2019. Second Bonn Challenge progress report: application of the Barometer in 2018. IUCN, International Union for Conservation of Nature. https://doi.org/10.2305/IUCN.CH.2019.06.en

Davis, F.W., Stoms, D.M., 1996. Sierran Vegetation: A Gap Analysis (Sierra Nevada Ecosystem Project: Final report to Congress No. 2), Assessments and scientific basis for management options. University of California, Centers for Water and Wildland Resources.

de Oliveira, S.N., Abílio de Carvalho Júnior, O., Trancoso Gomes, R.A., Fontes Guimarães, R., McManus, C.M., 2017a. Deforestation analysis in protected areas and scenario simulation for structural corridors in the agricultural frontier of Western Bahia, Brazil. Land Use Policy 61, 40–52. https://doi.org/10.1016/j.landusepol.2016.10.046

de Oliveira, S.N., de Carvalho Júnior, O.A., Gomes, R.A.T., Guimarães, R.F., McManus, C.M., 2017b. Landscape-fragmentation change due to recent agricultural expansion in the Brazilian Savanna, Western Bahia, Brazil. Reg Environ Change 17, 411–423. https://doi.org/10.1007/s10113-016-0960-0

DeFries, R., Hansen, A., Turner, B.L., Reid, R., Liu, J., 2007. Land Use Change Around Protected Areas: Management to Balance Human Needs and Ecological Function. Ecological Applications 17, 1031–1038. https://doi.org/10.1890/05-1111

Di Gregorio, A., Jansen, L.J.M., 2000. Land cover classification system (LCCS): Classification concepts and user manual (Environment and Natural Resources Service), GCP/RAF/287/ITA Africover—East Africa Project and Soil Resources, Management and Conservation Service. Food and Agriculture Organization.

Dornas, T., Barbosa, M. de O., Leite, G., Pinheiro, R.T., Prado, A.D., Crozariol, M.A., Carrano, E., 2013. Ocorrências da Arara-azul-grande (Anodorhynchus hyacinthinus) no Estado do Tocantins: distribuição, implicações biogeográficas e conservação 14.

Duca, C., Marini, M.Â., 2014. Territorial system and adult dispersal in a cooperative-breeding tanagerTerritory and dispersal of a tanager. Auk 131, 32–40. https://doi.org/10.1642/AUK-13-005.1

Eiten, G., 1982. Brazilian "Savannas," in: Ecology of Tropical Savannas. Springer, Berlin, Heidelberg, pp. 25–47.

Eiten, G., 1978. Delimitation of the cerrado concept. Plant Ecol 36, 169–178. https://doi.org/10.1007/BF02342599

Eiten, G., 1972. The cerrado vegetation of Brazil. The Botanical Review 38, 201–341. https://doi.org/10.1007/BF02859158

Embrapa, 1979. Serviço Nacional de Levantamento e Conservação de Solos (Súmula No. 10), Reunião Técnica de Levantamento de Solos. Empresa Brasileira de Pesquisa Agropecuária, Rio de Janeiro.

Fearnside, P.M., Ferraz, J., 1995. A Conservation Gap Analysis of Brazil's Amazonian Vegetation. Conservation Biology 9, 1134–1147. https://doi.org/10.1046/j.1523-1739.1995.9051127.x-i1

Felfili, M.C., Felfili, J.M., 2001. Diversidade alfa e beta no Cerrado Sensu Strictu da Chapada Pratinha, Brasil. Acta bot. bras. 12.

Ferreira, L.G., Huete, A.R., 2004. Assessing the seasonal dynamics of the Brazilian Cerrado vegetation through the use of spectral vegetation indices. International Journal of Remote Sensing 25, 1837–1860. https://doi.org/10.1080/0143116031000101530

Ferreira, L.G., Yoshioka, H., Huete, A., Sano, E.E., 2003. Seasonal landscape and spectral vegetation index dynamics in the Brazilian Cerrado: An analysis within the Large-Scale Biosphere–Atmosphere Experiment in Amazônia (LBA). Remote Sensing of Environment, Large Scale Biosphere Atmosphere Experiment in Amazonia 87, 534–550. https://doi.org/10.1016/j.rse.2002.09.003

Ferreira, M.E., Ferreira, L.G., Sano, E.E., Shimabukuro, Y.E., 2007. Spectral linear mixture modelling approaches for land cover mapping of tropical savanna areas in Brazil. International Journal of Remote Sensing 28, 413–429. https://doi.org/10.1080/01431160500181507

Ferrer-Paris, J.R., Zager, I., Keith, D.A., Oliveira-Miranda, M.A., Rodríguez, J.P., Josse, C., González-Gil, M., Miller, R.M., Zambrana-Torrelio, C., Barrow, E., 2019. An ecosystem risk assessment of temperate and tropical forests of the Americas with an outlook on future conservation strategies. Conservation Letters 12, e12623. https://doi.org/10.1111/conl.12623

Foley, J.A., DeFries, R., Asner, G.P., Barford, C., Bonan, G., Carpenter, S.R., Chapin, F.S., Coe, M.T., Daily, G.C., Gibbs, H.K., Helkowski, J.H., Holloway, T., Howard, E.A., Kucharik, C.J., Monfreda, C., Patz, J.A., Prentice, I.C., Ramankutty, N., Snyder, P.K., 2005. Global Consequences of Land Use. Science 309, 570–574. https://doi.org/10.1126/science.1111772

Françoso, R.D., Brandão, R., Nogueira, C.C., Salmona, Y.B., Machado, R.B., Colli, G.R., 2015b. Habitat loss and the effectiveness of protected areas in the Cerrado Biodiversity Hotspot. Natureza & Conservação 13, 35–40. https://doi.org/10.1016/j.ncon.2015.04.001

Franklin, J., 1991. Land cover stratification using Landsat Thematic Mapper data in Sahelian and Sudanian woodland and wooded grassland. Journal of Arid Environments 20, 141–163. https://doi.org/10.1016/S0140-1963(18)30705-5

Franklin, J., Davis, F.W., Lefebvre, P., 1991. Thematic mapper analysis of tree cover in semiarid woodlands using a model of canopy shadowing. Remote Sensing of Environment 36, 189–202. https://doi.org/10.1016/0034-4257(91)90056-C

Furley, P.A., Ratter, J.A., 1988. Soil Resources and Plant Communities of the Central Brazilian Cerrado and Their Development. Journal of Biogeography 15, 97–108. https://doi.org/10.2307/2845050

Galetti, M., Giacomini, H.C., Bueno, R.S., Bernardo, C.S.S., Marques, R.M., Bovendorp, R.S., Steffler, C.E., Rubim, P., Gobbo, S.K., Donatti, C.I., Begotti, R.A., Meirelles, F., Nobre, R. de A., Chiarello, A.G., Peres, C.A., 2009. Priority areas for the conservation of Atlantic forest large mammals. Biological Conservation, Conservation Issues in the Brazilian Atlantic Forest 142, 1229–1241. https://doi.org/10.1016/j.biocon.2009.01.023

Gallo, J.A., Pasquini, L., Reyers, B., Cowling, R.M., 2009. The role of private conservation areas in biodiversity representation and target achievement within the Little Karoo region, South Africa. Biological Conservation 142, 446–454. https://doi.org/10.1016/j.biocon.2008.10.025

Garcia, F.N., Ferreira, L.G., Leite, J.F., 2011. Áreas Protegidas no Bioma Cerrado: fragmentos vegetacionais sob forte pressão. Anais do XV Simpósio Brasileiro de Sensoriamento Remoto - SBSR, Curitiba, PR, Brasil 4086.

García-Moreno, J., Clay, R.P., Ríos-Muñoz, C.A., 2007. The importance of birds for conservation in the Neotropical region. J Ornithol 148, 321–326. https://doi.org/10.1007/s10336-007-0194-5

Gibbes, C., Adhikari, S., Rostant, L., Southworth, J., Qiu, Y., 2010. Application of Object Based Classification and High Resolution Satellite Imagery for Savanna Ecosystem Analysis. Remote Sensing 2, 2748–2772. https://doi.org/10.3390/rs2122748

Gibbs, H.K., Rausch, L., Munger, J., Schelly, I., Morton, D.C., Noojipady, P., Soares-Filho, B., Barreto, P., Micol, L., Walker, N.F., 2015. Brazil's Soy Moratorium. Science 347, 377–378. https://doi.org/10.1126/science.aaa0181

Girolamo-Neto, C.D., Fonseca, L.M.G., Körting, S., Soares, A.R., 2018. Mapping Brazilian Savanna Physiognomies using WorldView-2 Imagery and Geographic Object Based Image Analysis. GEOBIA 2018-From pixels to ecosystems and global sustainability 15.

Girolamo-Neto, C.D., Fonseca, L.M.G., Körting, T.S., 2017. Assessment of texture features for Brazilian Savanna classification: a case study in Brasília National Park. Revista Brasileira de Cartografia 69.

Goettsch, B., Durán, A.P., Gaston, K.J., 2019. Global gap analysis of cactus species and priority sites for their conservation. Conservation Biology 33, 369–376. https://doi.org/10.1111/cobi.13196

Gomes, L., Miranda, H.S., Silvério, D.V., Bustamante, M.M.C., 2020. Effects and behaviour of experimental fires in grasslands, savannas, and forests of the Brazilian Cerrado. Forest Ecology and Management 458, 117804. https://doi.org/10.1016/j.foreco.2019.117804

Grace, J., José, J.S., Meir, P., Miranda, H.S., Montes, R.A., 2006. Productivity and carbon fluxes of tropical savannas. Journal of Biogeography 33, 387–400. https://doi.org/10.1111/j.1365-2699.2005.01448.x

Grande, T.O., Aguiar, L.M.S., Machado, R.B., 2020. Heating a biodiversity hotspot: connectivity is more important than remaining habitat. Landscape Ecol 35, 639–657. https://doi.org/10.1007/s10980-020-00968-z

Graves, R.A., Williamson, M.A., Belote, R.T., Brandt, J.S., 2019. Quantifying the contribution of conservation easements to large-landscape conservation. Biological Conservation 232, 83–96. https://doi.org/10.1016/j.biocon.2019.01.024

Grimm, A.M., 2011. Interannual climate variability in South America: impacts on seasonal precipitation, extreme events, and possible effects of climate change. Stochastic Environmental Research and Risk Assessment 25, 537–554. https://doi.org/10.1007/s00477-010-0420-1

Haddad, N.M., Brudvig, L.A., Clobert, J., Davies, K.F., Gonzalez, A., Holt, R.D., Lovejoy, T.E., Sexton, J.O., Austin, M.P., Collins, C.D., Cook, W.M., Damschen, E.I., Ewers, R.M., Foster, B.L., Jenkins, C.N., King, A.J., Laurance, W.F., Levey, D.J., Margules, C.R., Melbourne, B.A., Nicholls, A.O., Orrock, J.L., Song, D.-X., Townshend, J.R., 2015. Habitat fragmentation and its lasting impact on Earth's ecosystems. Science Advances 1, e1500052. https://doi.org/10.1126/sciadv.1500052

Hansen, A.J., DeFries, R., 2007. Ecological Mechanisms Linking Protected Areas to Surrounding Lands. Ecological Applications 17, 974–988. https://doi.org/10.1890/05-1098

Harris, G.M., Jenkins, C.N., Pimm, S.L., 2005. Refining Biodiversity Conservation Priorities. Conservation Biology 19, 1957–1968. https://doi.org/10.1111/j.1523-1739.2005.00307.x

Hay, G. J., Castilla, G., 2008. Geographic Object-Based Image Analysis (GEOBIA): A new name for a new discipline, in: Blaschke, T., Lang, S., Hay, Geoffrey J. (Eds.), Object-Based Image Analysis: Spatial Concepts for Knowledge-Driven Remote Sensing Applications, Lecture Notes in Geoinformation and Cartography. Springer Berlin Heidelberg, Berlin, Heidelberg, pp. 75–89. https://doi.org/10.1007/978-3-540-77058-9_4

Hay, G.J., Niemann, K.O., McLean, G.F., 1996. An object-specific image-texture analysis of H-resolution forest imagery. Remote Sensing of Environment 55, 108–122. https://doi.org/10.1016/0034-4257(95)00189-1

Heller, N.E., Zavaleta, E.S., 2009. Biodiversity management in the face of climate change: A review of 22 years of recommendations. Biological Conservation 142, 14–32. https://doi.org/10.1016/j.biocon.2008.10.006

Herold, M., Mayaux, P., Woodcock, C.E., Baccini, A., Schmullius, C., 2008. Some challenges in global land cover mapping: An assessment of agreement and accuracy in existing 1 km datasets. Remote Sensing of Environment, Earth Observations for Terrestrial Biodiversity and Ecosystems Special Issue 112, 2538–2556. https://doi.org/10.1016/j.rse.2007.11.013

Hill, M.J., Zhou, Q., Sun, Q., Schaaf, C.B., Palace, M., 2017. Relationships between vegetation indices, fractional cover retrievals and the structure and composition of Brazilian Cerrado natural vegetation. International Journal of Remote Sensing 38, 874–905. https://doi.org/10.1080/01431161.2016.1271959

Hilty, J., Merenlender, A.M., 2003. Studying Biodiversity on Private Lands. Conservation Biology 17, 132–137. https://doi.org/10.1046/j.1523-1739.2003.01361.x

Huntley, B.J., Walker, B.H., 1982. Ecology of Tropical Savannas. Springer, Berlin, Heidelberg.

IBGE, 2015. Producao Agricola Municipal: culturas temporarias e permanentes.

IBGE, 2012. Instituto Brasileiro de Geografia e Estatística. Manual técnico da vegetação brasileira: sistema fitogeográfico, inventário das formações florestais e campestres, técnicas e manejo de coleções botânicas, procedimentos para mapeamentos, Manuais Técnicos de Geociências. IBGE- Diretoria de Geociências, Rio de Janeiro.

ICMBio, 2018. Livro vermelho da fauna brasileira ameaçada de extinção.

IUCN, 2020. The IUCN Red List of Threatened Species. Version 2020-2.

Jansen, L.J.M., Gregorio, A.D., 2002. Parametric land cover and land-use classifications as tools for environmental change detection. Agriculture, Ecosystems & Environment 91, 89–100. https://doi.org/10.1016/S0167-8809(01)00243-2

Jantz, P., Goetz, S., Laporte, N., 2014. Carbon stock corridors to mitigate climate change and promote biodiversity in the tropics. Nature Climate Change 4, 138–142. https://doi.org/10.1038/nclimate2105

Jin, C., Xiao, X., Merbold, L., Arneth, A., Veenendaal, E., Kutsch, W.L., 2013. Phenology and gross primary production of two dominant savanna woodland ecosystems in Southern Africa. Remote Sensing of Environment 135, 189–201. https://doi.org/10.1016/j.rse.2013.03.033

Johansen, K., Phinn, S., Taylor, M., 2015. Mapping woody vegetation clearing in Queensland, Australia from Landsat imagery using the Google Earth Engine. Remote Sensing Applications: Society and Environment 1, 36–49. https://doi.org/10.1016/j.rsase.2015.06.002

Joppa, L.N., O'Connor, B., Visconti, P., Smith, C., Geldmann, J., Hoffmann, M., Watson, J.E.M., Butchart, S.H.M., Virah-Sawmy, M., Halpern, B.S., Ahmed, S.E., Balmford, A., Sutherland, W.J., Harfoot, M., Hilton-Taylor, C., Foden, W., Minin, E.D., Pagad, S., Genovesi, P., Hutton, J., Burgess, N.D., 2016. Filling in biodiversity threat gaps. Science 352, 416–418. https://doi.org/10.1126/science.aaf3565

Kamal, S., Grodzinska-Jurczak, M., 2014. Should conservation of biodiversity involve private land? A Q methodological study in Poland to assess stakeholders' attitude. Biodivers Conserv 23, 2689–2704. https://doi.org/10.1007/s10531-014-0744-0

Kamal, S., Grodzińska-Jurczak, M., Brown, G., 2015. Conservation on private land: a review of global strategies with a proposed classification system. Journal of Environmental Planning and Management 58, 576–597. https://doi.org/10.1080/09640568.2013.875463

Kanegae, M.F., Levy, G., Freitas, S.R., 2012. Habitat use by Collared Crescentchest (Melanopareia torquata) in a Cerrado in southeastern Brazil: implications for management. Brazilian Journal of Biology 72, 865–871. https://doi.org/10.1590/S1519-69842012000500013

Kareiva, P., 2010. Trade-in to trade-up. Nature 466, 322–323. https://doi.org/10.1038/466322a

Kaszta, Ż., Van De Kerchove, R., Ramoelo, A., Cho, M.A., Madonsela, S., Mathieu, R., Wolff, E., 2016. Seasonal Separation of African Savanna Components Using Worldview-2 Imagery: A Comparison of Pixel- and Object-Based Approaches and Selected Classification Algorithms. Remote Sensing 8, 763. https://doi.org/10.3390/rs8090763

Kennedy, C. M., Miteva, D.A., Baumgarten, L., Hawthorne, P.L., Sochi, K., Polasky, S., Oakleaf, J.R., Uhlhorn, E.M., Kiesecker, J., 2016a. Bigger is better: Improved nature conservation and economic returns from landscape-level mitigation. Science Advances 2, e1501021–e1501021. https://doi.org/10.1126/sciadv.1501021

Kennedy, C. M., Hawthorne, P.L., Miteva, D.A., Baumgarten, L., Sochi, K., Matsumoto, M., Evans, J.S., Polasky, S., Hamel, P., Vieira, E.M., Develey, P.F., Sekercioglu, C.H., Davidson, A.D., Uhlhorn, E.M., Kiesecker, J., 2016b. Optimizing land use decision-making to sustain Brazilian agricultural profits, biodiversity and ecosystem services. Biological Conservation 204, 221–230. https://doi.org/10.1016/j.biocon.2016.10.039

Klink, C.A., 2019. Tropical Savannas and Conciliating Production with Conservation Strategies: The Case of Brazil, in: Leal Filho, W., Azul, A.M., Brandli, L., Özuyar, P.G., Wall, T. (Eds.), Life on Land. Springer International Publishing, Cham, pp. 1–9. https://doi.org/10.1007/978-3-319-71065-5_57-1

Klink, C.A., Machado, R.B., 2005. Conservation of the Brazilian cerrado. Conservation biology 19, 707–713.

Knight, R.L., 1999. Private Lands: The Neglected Geography. Conservation Biology 13, 223–224. https://doi.org/10.1046/j.1523-1739.1999.013002223.x

Kosmidou, V., Petrou, Z., Bunce, R.G.H., Mücher, C.A., Jongman, R.H.G., Bogers, M.M.B., Lucas, R.M., Tomaselli, V., Blonda, P., Padoa-Schioppa, E., Manakos, I., Petrou, M., 2014. Harmonization of the Land Cover Classification System (LCCS) with the General Habitat Categories (GHC) classification system. Ecological Indicators 36, 290–300. https://doi.org/10.1016/j.ecolind.2013.07.025

Kremen, C., Merenlender, A.M., 2018. Landscapes that work for biodiversity and people. Science 362. https://doi.org/10.1126/science.aau6020

Lang, S., 2008. Object-based image analysis for remote sensing applications: modeling reality – dealing with complexity, in: Blaschke, T., Lang, Stefan, Hay, G.J. (Eds.), Object-Based Image Analysis: Spatial Concepts for Knowledge-Driven Remote Sensing Applications, Lecture Notes in Geoinformation and Cartography. Springer, Berlin, Heidelberg, pp. 3–27. https://doi.org/10.1007/978-3-540-77058-9_1

Lehtomäki, J., Moilanen, A., 2013. Methods and workflow for spatial conservation prioritization using Zonation. Environmental Modelling & Software 47, 128–137. https://doi.org/10.1016/j.envsoft.2013.05.001

Lemes, L., de Andrade, A.F.A., Loyola, R., 2019. Spatial priorities for agricultural development in the Brazilian Cerrado: may economy and conservation coexist? Biodivers Conserv. https://doi.org/10.1007/s10531-019-01719-6

Liaw, A., Wiener, M., 2002. Classification and Regression by randomForest. R news 2, 18–22.

Lopes, L.E., 2009. Taxonomia, biogeografia e conservação das aves endêmicas do Cerrado.

L'Roe, J., Rausch, L., Munger, J., Gibbs, H.K., 2016. Mapping properties to monitor forests: Landholder response to a large environmental registration program in the Brazilian Amazon. Land Use Policy 57, 193–203. https://doi.org/10.1016/j.landusepol.2016.05.029

MacLean, M.G., Congalton, D.R.G., 2012. Map accuracy assessment issues when using an object-oriented approach, in: Proceedings of the American Society for Photogrammetry and Remote Sensing 2012 Annual Conference. Presented at the ASPRS 2012 Annual Conference, Sacramento, California, pp. 1–5.

Magioli, M., Ferraz, K.M.P.M. de B., Setz, E.Z.F., Percequillo, A.R., Rondon, M.V. de S.S., Kuhnen, V.V., Canhoto, M.C. da S., dos Santos, K.E.A., Kanda, C.Z., Fregonezi, G. de L., do Prado, H.A., Ferreira, M.K., Ribeiro, M.C., Villela, P.M.S., Coutinho, L.L., Rodrigues, M.G., 2016. Connectivity maintain mammal assemblages functional diversity within agricultural and fragmented landscapes. Eur J Wildl Res 62, 431–446. https://doi.org/10.1007/s10344-016-1017-x

MAPA, 2014. Ministério da Agricultura Pecuária e Abastecimento. Projeções do Agronegócio: Brasil 2013/2014 a 2023/2024—Projeções de Longo Prazo. 5nd ed. Brasília: Ministério da Agricultura, Pecuária e Abastecimento. Assessoria de Gestão Estratégica. 100 pg.

Margules, C.R., Pressey, R.L., 2000. Systematic Conservation Planning. Nature, insight review articles 405.

Marini, M.Â., Barbet-Massin, M., Lopes, L.E., Jiguet, F., 2009. Major current and future gaps of Brazilian reserves to protect Neotropical savanna birds. Biological Conservation 142, 3039–3050. https://doi.org/10.1016/j.biocon.2009.08.002

Marini, M.A., Cavalcanti, R.B., 1993. Habitat and foraging substrate use of three Basileuterus Warblers from central Brazil. Ornitologia Neotropical 4, 69–76.

Matricardi, E.A.T., Mendes, T.J., Pereira, E.M., Vasconcelos, P.G. de A., Ângelo, H., Costa, O.B. da, 2019. Dinamica no uso e cobertura da terra na regiao do MATOBIPAentre 200 e 2016. Nativa: Pesquisas Agrárias e Ambientais. https://doi.org/10.31413/nativa.v7i5.7391

Mayes, M.T., Mustard, J.F., Melillo, J.M., 2015. Forest cover change in Miombo Woodlands: modeling land cover of African dry tropical forests with linear spectral mixture analysis. Remote Sensing of Environment 165, 203–215. https://doi.org/10.1016/j.rse.2015.05.006

McCarthy, M.J., Radabaugh, K.R., Moyer, R.P., Muller-Karger, F.E., 2018. Enabling efficient, large-scale high-spatial resolution wetland mapping using satellites. Remote Sensing of Environment 208, 189–201. https://doi.org/10.1016/j.rse.2018.02.021

McFeeters, S.K., 1996. The use of the Normalized Difference Water Index (NDWI) in the delineation of open water features. International Journal of Remote Sensing 17, 1425–1432. https://doi.org/10.1080/01431169608948714

McGarigal, K., Marks, B.J., 1994. FRAGSTATS: spatial pattern analysis program for quantifying landscape structure. Version 2.0. Forest Science Department, Oregon State University, Corvallis.

McRae, B.H., Dickson, B.G., Keitt, T.H., Shah, V.B., 2008. Using Circuit Theory to Model Connectivity in Ecology, Evolution, and Conservation. Ecology 89, 2712–2724. https://doi.org/10.1890/07-1861.1

McRae, B.H., Viral B. Shah, Edelman, A., 2016. Circuitscape: modeling landscape connectivity to promote conservation and human health. https://doi.org/10.13140/RG.2.1.4265.1126

Medeiros, R., Young, C.E.F., 2011. Contribuição das unidades de conservação brasileiras para a economia nacional: Relatório Final. UNEP-WCMC, Brasília, DF.

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

Mendonça, R., Felfili, J., Walter, B., Silva-Junior, M., Rezende, A., Filgueiras, T., Nogueira, P., Fagg, C., 2008. Vascular flora of the Cerrado biome: Checklist with 12,356 species, in: Cerrado: Ecologia e Flora. Embrapa Cerrados, pp. 422–442.

Metzger, J.P., Bustamante, M.M.C., Ferreira, J., Fernandes, G.W., Librán-Embid, F., Pillar, V.D., Prist, P.R., Rodrigues, R.R., Vieira, I.C.G., Overbeck, G.E., 2019. Why Brazil needs its Legal Reserves. Perspectives in Ecology and Conservation 17, 91–103. https://doi.org/10.1016/j.pecon.2019.07.002

Michalski, F., Metzger, J.P., Peres, C.A., 2010. Rural property size drives patterns of upland and riparian forest retention in a tropical deforestation frontier. Global Environmental Change, 20th Anniversary Special Issue 20, 705–712. https://doi.org/10.1016/j.gloenvcha.2010.04.010

Miranda, E., Magalhaes, L., de Carvalho, C.A., 2014. Proposta de Delimitacao Territorial do MATOPIBA. Embrapa.

Mittermeier, R.A., Fonseca, G.A.B. da, Rylands, A.B., Brandon, K., 2005. A Brief History of Biodiversity Conservation in Brazil. Conservation Biology 19, 601–607.

MMA, 2003. Sistema Nacional de Unidades de Conservação – SNUC: lei n° 9.985, de 18 de Julho de 2000; decreto n° 4.340, de 22 de agosto de 2002. 3 ed. aum. MMA/SBF, Brasília.

Moilanen, A., 2007. Landscape Zonation, benefit functions and target-based planning: Unifying reserve selection strategies. Biological Conservation 134, 571–579. https://doi.org/10.1016/j.biocon.2006.09.008

Moilanen, A., Arponen, A., 2011. Administrative regions in conservation: Balancing local priorities with regional to global preferences in spatial planning. Biological Conservation, Ecoregional-scale monitoring within conservation areas, in a rapidly changing climate 144, 1719–1725. https://doi.org/10.1016/j.biocon.2011.03.007

Moilanen, A., Franco, A.M.A., Early, R.I., Fox, R., Wintle, B., Thomas, C.D., 2005. Prioritizing multiple-use landscapes for conservation: methods for large multi-species planning problems. Proceedings of the Royal Society B: Biological Sciences 272, 1885–1891. https://doi.org/10.1098/rspb.2005.3164

Monteiro, L.M., Brum, F.T., Pressey, R.L., Morellato, L.P.C., Soares-Filho, B., Lima-Ribeiro, M.S., Loyola, R., 2020. Evaluating the impact of future actions in minimizing vegetation loss from land conversion in the Brazilian Cerrado under climate change. Biodivers Conserv 29, 1701–1722. https://doi.org/10.1007/s10531-018-1627-6

Müller, H., Rufin, P., Griffiths, P., Barros Siqueira, A.J., Hostert, P., 2015. Mining dense Landsat time series for separating cropland and pasture in a heterogeneous Brazilian savanna landscape. Remote Sensing of Environment 156, 490–499. https://doi.org/10.1016/j.rse.2014.10.014

Munang, R., Thiaw, I., Alverson, K., Mumba, M., Liu, J., Rivington, M., 2013. Climate change and Ecosystem-based Adaptation: a new pragmatic approach to buffering climate change impacts. Current Opinion in Environmental Sustainability, Terrestrial systems 5, 67–71. https://doi.org/10.1016/j.cosust.2012.12.001

Myint, S.W., Gober, P., Brazel, A., Grossman-Clarke, S., Weng, Q., 2011. Per-pixel vs. object-based classification of urban land cover extraction using high spatial resolution imagery. Remote Sensing of Environment 115, 1145–1161. https://doi.org/10.1016/j.rse.2010.12.017

Nagendra, H., Lucas, R., Honrado, J.P., Jongman, R.H.G., Tarantino, C., Adamo, M., Mairota, P., 2013. Remote sensing for conservation monitoring: Assessing protected areas, habitat extent, habitat condition, species diversity, and threats. Ecological Indicators, Biodiversity Monitoring 33, 45–59. https://doi.org/10.1016/j.ecolind.2012.09.014

Nagendra, H., Rocchini, D., 2008. High resolution satellite imagery for tropical biodiversity studies: the devil is in the detail. Biodiversity and Conservation 17, 3431–3442. https://doi.org/10.1007/s10531-008-9479-0

Naidoo, L., Cho, M.A., Mathieu, R., Asner, G., 2012. Classification of savanna tree species, in the Greater Kruger National Park region, by integrating hyperspectral and LiDAR data in a Random Forest data mining environment. ISPRS Journal of Photogrammetry and Remote Sensing 69, 167–179. https://doi.org/10.1016/j.isprsjprs.2012.03.005

Naidoo, R., Balmford, A., Ferraro, P.J., Polasky, S., Ricketts, T.H., Rouget, M., 2006. Integrating economic costs into conservation planning. Trends in Ecology & Evolution 21, 681–687. https://doi.org/10.1016/j.tree.2006.10.003

Nogueira, C., Ribeiro, S., Costa, G.C., Colli, G.R., 2011. Vicariance and endemism in a Neotropical savanna hotspot: distribution patterns of Cerrado squamate reptiles. Journal of Biogeography 38, 1907–1922. https://doi.org/10.1111/j.1365-2699.2011.02538.x

Noojipady, P., Morton, C.D., Macedo, N.M., Victoria, C.D., Huang, C., Gibbs, K.H., Bolfe, L.E., 2017. Forest carbon emissions from cropland expansion in the Brazilian Cerrado biome. Environ. Res. Lett. 12, 025004. https://doi.org/10.1088/1748-9326/aa5986

Noss, R.F., Dobson, A.P., Baldwin, R., Beier, P., Davis, C.R., Dellasala, D.A., Francis, J., Locke, H., Nowak, K., Lopez, R., Reining, C., Trombulak, S.C., Tabor, G., 2012. Bolder Thinking for Conservation. Conservation Biology 26, 1–4. https://doi.org/10.1111/j.1523-1739.2011.01738.x

Nou, E., Costa, N., 1994. Diagnóstico da qualidade ambiental da bacia do rio São Francisco: sub-bacias do Oeste Baiano e Sobradinho. IBGE, Primeira Divisão de Geociências do Nordeste, Rio de Janeiro - Brazil.

Oliveira, S.N. de, Júnior, O.A. de C., Gomes, R.A.T., Guimarães, R.F., Martins, É. de S., 2014. Detecção de mudança do uso e cobertura da terra usando o método de pós-classificação na fronteira agrícola do oeste da Bahia sobre o Grupo Urucuia durante o período 1988 - 2011. Revista Brasileira de Cartografia 5.

Oliveira-Filho, A.T., Ratter, J.A., 2002. Vegetation physiognomies and woody flora of the Cerrado biome, in: The Cerrados of Brazil: Ecology and Natural History of a Neotropical Savanna. Columbia University Press, New York.

Olofsson, P., Foody, G.M., Herold, M., Stehman, S.V., Woodcock, C.E., Wulder, M.A., 2014. Good practices for estimating area and assessing accuracy of land change. Remote Sensing of Environment 148, 42–57. https://doi.org/10.1016/j.rse.2014.02.015

Orozco Filho, J., 2017. Avaliação do Uso da Abordagem Orientada-Objeto com Imagens de Alta Resolução RapidEye na Classificação das Fitofisionomias do Cerrado. Universidade de Brasilia, UnB.

Overbeck, G.E., Vélez-Martin, E., Scarano, F.R., Lewinsohn, T.M., Fonseca, C.R., Meyer, S.T., Müller, S.C., Ceotto, P., Dadalt, L., Durigan, G., Ganade, G., Gossner, M.M., Guadagnin, D.L., Lorenzen, K., Jacobi, C.M., Weisser, W.W., Pillar, V.D., 2015. Conservation in Brazil needs to include non-forest ecosystems. Diversity and Distributions 21, 1455–1460. https://doi.org/10.1111/ddi.12380

Paese, A., Paglia, A., Pinto, L.P., Foster, M.N., Fonseca, M., Sposito, R., 2010. Fine-scale sites of global conservation importance in the Atlantic forest of Brazil. Biodivers Conserv 19, 3445–3458. https://doi.org/10.1007/s10531-010-9906-x

Parr, C.L., Lehmann, C.E.R., Bond, W.J., Hoffmann, W.A., Andersen, A.N., 2014. Tropical grassy biomes: misunderstood, neglected, and under threat. Trends in Ecology & Evolution 29, 205–213. https://doi.org/10.1016/j.tree.2014.02.004

Pegas, F. de V., Castley, J.G., 2014. Ecotourism as a conservation tool and its adoption by private protected areas in Brazil. Journal of Sustainable Tourism 22, 604–625. https://doi.org/10.1080/09669582.2013.875550

Pereira, B.A.S., Furtado, P.P., 2011. Vegetação da Bacia do Córrego Taquara: Coberturas Naturais e Antrópicas, in: Reserva Ecológica Do IBGE: Biodiversidade Terrestre. IBGE, Coordenacao de Recursos Naturais e Estudos Ambientais, Rio de Janeiro.

Powell, R.L., Matzke, N., de Souza, C., Clark, M., Numata, I., Hess, L.L., Roberts, D.A., 2004. Sources of error in accuracy assessment of thematic land-cover maps in the Brazilian Amazon. Remote Sensing of Environment 90, 221–234. https://doi.org/10.1016/j.rse.2003.12.007

Radoux, J., Bogaert, P., 2014. Accounting for the area of polygon sampling units for the prediction of primary accuracy assessment indices. Remote Sensing of Environment 142, 9–19. https://doi.org/10.1016/j.rse.2013.10.030

Radoux, J., Bogaert, P., Radoux, J., Bogaert, P., 2017. Good Practices for Object-Based Accuracy Assessment. Remote Sensing 9, 646. https://doi.org/10.3390/rs9070646

Rambaldi, D.M., Fernandes, R.V., Schmidt, M.A.R., 2005. Private protected areas and their key role in the conservation of the Atlantic Forest biodiversity hotspot, Brazil. Parks 15.

Ratana, P., Huete, A.R., Ferreira, L., 2005. Analysis of Cerrado Physiognomies and Conversion in the MODIS Seasonal–Temporal Domain. Earth Interact. 9, 1–22.

Ratter, J.A., Ribeiro, J.F., Bridgewater, S., 1997. The Brazilian cerrado vegetation and threats to its biodiversity. Annals of botany 80, 223–230.

Reynolds, J., Wesson, K., Desbiez, A.L.J., Ochoa-Quintero, J.M., Leimgruber, P., 2016. Using Remote Sensing and Random Forest to Assess the Conservation Status of Critical Cerrado Habitats in Mato Grosso do Sul, Brazil. Land 5, 12. https://doi.org/10.3390/land5020012

Rezende, C.L., Fraga, J.S., Sessa, J.C., de Souza, G.V.P., Assad, E.D., Scarano, F.R., 2018. Land use policy as a driver for climate change adaptation: A case in the domain of the Brazilian Atlantic forest. Land Use Policy 72, 563–569. https://doi.org/10.1016/j.landusepol.2018.01.027

Ribeiro, F.F., Roberts, D.A., Hess, L.L., W. Davis, F., Caylor, K.K., Antunes Daldegan, G., 2020. Geographic Object-Based Image Analysis Framework for Mapping Vegetation Physiognomic Types at Fine Scales in Neotropical Savannas. Remote Sensing 12, 1721. https://doi.org/10.3390/rs12111721

Ribeiro, J.F., Walter, B.M.T., 2008. As principais fitofisionomias do bioma Cerrado, in: Cerrado: Ecologia e Flora. Embrapa Cerrados, Brasilia, DF, pp. 151–199.

Ribeiro, M.L., 2011. Reserva Ecologica do IBGE: Biodiversidade Terrestre. IBGE, Coordenacao de Recursos Naturais e Estudos Ambientais, Rio de Janeiro.

Richards, J.A., 1996. Classifier performance and map accuracy. Remote Sensing of Environment 57, 161–166. https://doi.org/10.1016/0034-4257(96)00038-7

Richards, P.D., VanWey, L., 2016. Farm-scale distribution of deforestation and remaining forest cover in Mato Grosso. Nature Clim Change 6, 418–425. https://doi.org/10.1038/nclimate2854

Roberts, D.A., Keller, M., Soares, J.V., 2003. Studies of land-cover, land-use, and biophysical properties of vegetation in the Large Scale Biosphere Atmosphere experiment in Amazônia. Remote Sensing of Environment 87, 377–388. https://doi.org/10.1016/j.rse.2003.08.012

Rocha, R.G., Ferreira, E., Martins, I.C.M., Costa, L.P., Fonseca, C., 2014. Seasonally flooded stepping stones: emerging diversity of small mammal assemblage in the Amazonia-Cerrado ecotone, central Brazil. Zool. Stud. 53, 60. https://doi.org/10.1186/s40555-014-0060-0

Rosa, C.A., 2020. Savannah for sale: is there hope for neotropical biodiversity on private, Brazilian properties? Animal Conservation n/a. https://doi.org/10.1111/acv.12617

Rouse, J.W., Hass, H.R., Schell, J.A., Deering, D.W., 1973. Monitoring vegetation systems in the Great Plains with ERTS, in: Proceedings of the Third ERTS Symposium. Goddard Space Flight Center, NASA SP-351, NASA, Washington, DC.

Salmona, Y.B., Paiva, A.O., Matricardi, E.A.T., 2016. Estimativas futuras de desmatamento e emmisões de CO^2 equivalente no Oeste Baiano. Revista Brasileira de Cartografia 68.

Sano, E.E., Ferreira, L.G., Asner, G.P., Steinke, E.T., 2007. Spatial and temporal probabilities of obtaining cloud-free Landsat images over the Brazilian tropical savanna. International Journal of Remote Sensing 28, 2739–2752. https://doi.org/10.1080/01431160600981517

Sano, E.E., Rosa, R., Brito, J.L.S., Ferreira, L.G., 2010. Land cover mapping of the tropical savanna region in Brazil. Environmental Monitoring and Assessment 166, 113–124. https://doi.org/10.1007/s10661-009-0988-4

Santana, O.A., Júnior, O.A. de C., Gomes, R.A.T., Cardoso, W. dos S., Martins, É. de S., Passo, D.P., Guimarães, R.F., 2010. Distribuição de espécies vegetais nativas em distintos macroambientes na Região do oeste da Bahia. Revista Espaço e Geografia 13.

Saura, S., Bastin, L., Battistella, L., Mandrici, A., Dubois, G., 2017. Protected areas in the world’s ecoregions: How well connected are they? Ecological Indicators 76, 144–158. https://doi.org/10.1016/j.ecolind.2016.12.047

Saura, S., Bertzky, B., Bastin, L., Battistella, L., Mandrici, A., Dubois, G., 2019. Global trends in protected area connectivity from 2010 to 2018. Biological Conservation 238, 108183. https://doi.org/10.1016/j.biocon.2019.07.028

Saura, S., Bertzky, B., Bastin, L., Battistella, L., Mandrici, A., Dubois, G., 2018. Protected area connectivity: Shortfalls in global targets and country-level priorities. Biological Conservation 219, 53–67. https://doi.org/10.1016/j.biocon.2017.12.020

Sawyer, D., Mesquita, B., Coutinho, B., Almeida, F., Figueiredo, I., Lamas, I., Pereira, L., Pinto, L., Pires, M., Kasecker, T., 2017. Ecosystem Profile Cerrado Biodiversity Hotspot. Critical Ecosystem Partnership Fund.

Scholes, R.J., Archer, S.R., 1997. Tree-Grass Interactions in Savannas. Annual Review of Ecology and Systematics 28, 517–544. https://doi.org/10.1146/annurev.ecolsys.28.1.517

Schuster, C., Förster, M., Kleinschmit, B., 2012. Testing the red edge channel for improving land-use classifications based on high-resolution multi-spectral satellite data. International Journal of Remote Sensing 33, 5583–5599. https://doi.org/10.1080/01431161.2012.666812

Schuster, R., Law, E.A., Rodewald, A.D., Martin, T.G., Wilson, K.A., Watts, M., Possingham, H.P., Arcese, P., 2018. Tax Shifting and Incentives for Biodiversity Conservation on Private Lands. Conservation Letters 11, e12377. https://doi.org/10.1111/conl.12377

Schwieder, M., Leitão, P.J., da Cunha Bustamante, M.M., Ferreira, L.G., Rabe, A., Hostert, P., 2016. Mapping Brazilian savanna vegetation gradients with Landsat time series. International Journal of Applied Earth Observation and Geoinformation 52, 361–370. https://doi.org/10.1016/j.jag.2016.06.019

Scott, J.M., Davis, F., Csuti, B., Noss, R., Butterfield, B., Groves, C., Anderson, H., Caicco, S., D'Erchia, F., Edwards, T.C., Ulliman, J., Wright, R.G., 1993. Gap Analysis: A Geographic Approach to Protection of Biological Diversity. Wildlife Monographs 3–41.

Scott, J.M., Murray, M., Wright, R.G., Csuti, B., Morgan, P., Pressey, R.L., 2001. Representation of natural vegetation in protected areas: capturing the geographic range. Biodiversity and Conservation 10, 1297–1301. https://doi.org/10.1023/A:1016647726583

Silva, A.D., Bergamini, L.L., In prep. Biodiversidade, in: Reserva Ecológica Do IBGE, Compromisso Com a Biodiversidade. IBGE, Coordenacao de Recursos Naturais e Estudos Ambientais, Brasilia, DF, Brazil.

Silva, J.M.C. da, Straube, F.C., 1996. Systematics and Biogeography of Scaled Woodcreepers (Aves: Dendrocolaptidae). Studies on Neotropical Fauna and Environment 31, 3–10. https://doi.org/10.1076/snfe.31.1.3.13321

Silva, J.F., Farinas, M.R., Felfili, J.M., Klink, C.A., 2006. Spatial heterogeneity, land use and conservation in the cerrado region of Brazil. Journal of Biogeography 33, 536–548. https://doi.org/10.1111/j.1365-2699.2005.01422.x

Silva, J.M., Bates, J.M., 2002. Biogeographic Patterns and Conservation in the South American Cerrado: A Tropical Savanna Hotspot. BioScience 52, 225. https://doi.org/10.1641/0006-3568(2002)052[0225:BPACIT]2.0.CO;2

Silva, J.M., 1997. Endemic bird species and conservation in the Cerrado Region, South America. Biodiversity & Conservation 6, 435–450.

Silva, J.M., 1997. Endemic bird species and conservation in the Cerrado Region, South America. Biodiversity and Conservation 6, 435–450. https://doi.org/10.1023/A:1018368809116

Silva, L.R., Sano, E.E., 2016. Análise das imagens do satélite RapidEye para discriminação da cobertura vegetal do bioma Cerrado. Revista Brasileira de Cartografia 68.

Smith, J.H., Wickham, J.D., Stehman, S.V., Yang, L., 2002. Impacts of Patch Size and Land-Cover Heterogeneity on Thematic Image Classification Accuracy. Photogrammetric Engineering 6.

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

Soares-Filho, B., Rajão, R., Merry, F., Rodrigues, H., Davis, J., Lima, L., Macedo, M., Coe, M., Carneiro, A., Santiago, L., 2016. Brazil's Market for Trading Forest Certificates. PLOS ONE 11, e0152311. https://doi.org/10.1371/journal.pone.0152311

Soares-Filho, B.S., Nepstad, D.C., Curran, L.M., Cerqueira, G.C., Garcia, R.A., Ramos, C.A., Voll, E., McDonald, A., Lefebvre, P., Schlesinger, P., 2006. Modelling conservation in the Amazon basin. Nature 440, 520–523. https://doi.org/10.1038/nature04389

Soterroni, A.C., Ramos, F.M., Mosnier, A., Fargione, J., Andrade, P.R., Baumgarten, L., Pirker, J., Obersteiner, M., Kraxner, F., Câmara, G., Carvalho, A.X.Y., Polasky, S., 2019. Expanding the Soy Moratorium to Brazil's Cerrado. Science Advances 5, eaav7336. https://doi.org/10.1126/sciadv.aav7336

Spera, S.A., Galford, G.L., Coe, M.T., Macedo, M.N., Mustard, J.F., 2016. Land-use change affects water recycling in Brazil's last agricultural frontier. Global Change Biology 22, 3405–3413. https://doi.org/10.1111/gcb.13298

Stefanes, M., Roque, F. de O., Lourival, R., Melo, I., Renaud, P.C., Quintero, J.M.O., 2018. Property size drives differences in forest code compliance in the Brazilian Cerrado. Land Use Policy 75, 43–49. https://doi.org/10.1016/j.landusepol.2018.03.022

Stehman, S.V., 1997. Selecting and interpreting measures of thematic classification accuracy. Remote Sensing of Environment 62, 77–89. https://doi.org/10.1016/S0034-4257(97)00083-7

Stehman, S.V., Wickham, J.D., Fattorini, L., Wade, T.D., Baffetta, F., Smith, J.H., 2009. Estimating accuracy of land-cover composition from two-stage cluster sampling. Remote Sensing of Environment 113, 1236–1249. https://doi.org/10.1016/j.rse.2009.02.011

Stickler, C.M., Nepstad, D.C., Azevedo, A.A., McGrath, D.G., 2013. Defending public interests in private lands: compliance, costs and potential environmental consequences of the Brazilian Forest Code in Mato Grosso. Philosophical Transactions of the Royal Society B: Biological Sciences 368, 20120160. https://doi.org/10.1098/rstb.2012.0160

Stolton, S., Redford, K.H., Dudley, N., 2014. The Futures of Privately Protected Areas. IUCN.

Strassburg, B.B.N., Brooks, T., Feltran-Barbieri, R., Iribarrem, A., Crouzeilles, R., Loyola, R., Latawiec, A.E., Oliveira Filho, F.J.B., Scaramuzza, C.A. de M., Scarano, F.R., Soares-Filho, B., Balmford, A., 2017. Moment of truth for the Cerrado hotspot. Nature Ecology & Evolution 1. https://doi.org/10.1038/s41559-017-0099

Strassburg, B.B.N., Latawiec, A.E., Barioni, L.G., Nobre, C.A., da Silva, V.P., Valentim, J.F., Vianna, M., Assad, E.D., 2014. When enough should be enough: Improving the use of current agricultural lands could meet production demands and spare natural habitats in Brazil. Global Environmental Change 28, 84–97. https://doi.org/10.1016/j.gloenvcha.2014.06.001

Swift, B., Arias, V., Bass, S., Chacon, C.M., 2004. Private Lands Conservation in Latin America: The Need for Enhanced Legal Tools and Incentives. J. Envtl. L. & Litig. 19, 85.

Symeonakis, E., Higginbottom, T.P., Petroulaki, K., Rabe, A., 2018. Optimisation of Savannah Land Cover Characterisation with Optical and SAR Data. Remote Sensing 10, 499. https://doi.org/10.3390/rs10040499

Teixeira, L.R., Nunes, G.M., Finger, Z., Siqueira, A.J.B., 2015. Potencialidades da Classificação Orientada a Objetos em Imagens SPOT5 no Mapeamento de Fitofisionomias do Cerrado. Revista ESPACIOS | Vol. 36 (Nº 20) Año 2015.

Tilman, D., Clark, M., Williams, D.R., Kimmel, K., Polasky, S., Packer, C., 2017. Future threats to biodiversity and pathways to their prevention. Nature 546, 73–81. https://doi.org/10.1038/nature22900

Tilman, D., Socolow, R., Foley, J.A., Hill, J., Larson, E., Lynd, L., Pacala, S., Reilly, J., Searchinger, T., Somerville, C., Williams, R., 2009. Beneficial Biofuels—The Food, Energy, and Environment Trilemma. Science 325, 270–271. https://doi.org/10.1126/science.1177970

Trancoso, R., Sano, E.E., Meneses, P.R., 2014. The spectral changes of deforestation in the Brazilian tropical savanna. Environ Monit Assess 187, 4145. https://doi.org/10.1007/s10661-014-4145-3

Uezu, A., Metzger, J.P., 2011. Vanishing bird species in the Atlantic Forest: relative importance of landscape configuration, forest structure and species characteristics. Biodivers Conserv 20, 3627–3643. https://doi.org/10.1007/s10531-011-0154-5

Valdujo, P.H., Silvano, D.L., Colli, G., Martins, M., 2012. Anuran Species Composition and Distribution Patterns in Brazilian Cerrado, a Neotropical Hotspot. sajh 7, 63–78. https://doi.org/10.2994/057.007.0209

Vaughn, N.R., Asner, G.P., Smit, I.P.J., Riddel, E.S., 2015. Multiple Scales of Control on the Structure and Spatial Distribution of Woody Vegetation in African Savanna Watersheds. PLOS ONE 10, e0145192. https://doi.org/10.1371/journal.pone.0145192

Vieira, R.R.S., Ribeiro, B.R., Resende, F.M., Brum, F.T., Machado, N., Sales, L.P., Macedo, L., Soares-Filho, B., Loyola, R., 2018. Compliance to Brazil's Forest Code will not protect biodiversity and ecosystem services. Diversity and Distributions 24, 434–438. https://doi.org/10.1111/ddi.12700

Whiteside, T.G., Boggs, G.S., Maier, S.W., 2011. Comparing object-based and pixel-based classifications for mapping savannas. International Journal of Applied Earth Observation and Geoinformation 13, 884–893. https://doi.org/10.1016/j.jag.2011.06.008

Wiedmann, S.M.P., Guagliardi, R., 2018. A Reserva Particular do Patrimônio Natural (RPPN): unidade de conservação particular, in: Programa Estadual de Reservas Paticulares do Patrimônio Natural - RPPNs: 10 anos de apoio à conservação da biodiversidade. Instituto Estadual do Ambiente, Rio de Janeiro - Brazil, pp. 12–39.

Wintle, B.A., Kujala, H., Whitehead, A., Cameron, A., Veloz, S., Kukkala, A., Moilanen, A., Gordon, A., Lentini, P.E., Cadenhead, N.C.R., Bekessy, S.A., 2019. Global synthesis of conservation studies reveals the importance of small habitat patches for biodiversity. Proc Natl Acad Sci USA 116, 909–914. https://doi.org/10.1073/pnas.1813051115

WWF-Brazil, 2015. Áreas Prioritárias para Conservação da Biodiversidade no Cerrado e Pantanal. Brasília, DF.

Zimbres, B., Machado, R.B., Peres, C.A., 2018. Anthropogenic drivers of headwater and riparian forest loss and degradation in a highly fragmented southern Amazonian landscape. Land Use Policy 72, 354–363. https://doi.org/10.1016/j.landusepol.2017.12.062