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The role of environmental legislation and land use patterns on riparian deforestation dynamics in an Amazonian agricultural frontier (MT, Brazil)

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ABSTRACT

Agribusiness expansion and intensification in Brazil have prompted an abrupt change in land use and occupation in the Amazonian agricultural frontier since the 1980s. Considering the increasing suppression of native vegetation, riparian areas represent an important tool for protecting ecosystem services and biodiversity. Although the effects of land tenure and land use on large-scale deforestation in the Amazon have been widely assessed, their roles on riparian deforestation remains poorly explored. Here we assessed two municipalities – Querência (QRC) and São José do Xingu (SJX) – located in an agricultural frontier of the Brazilian Amazon to explore the relationship of illegal deforestation in riparian areas and different types of land use and property-sizes, as well as the impact of the Native Vegetation Protection Law (NVPL, Federal Law 12,651/2012) on environmental compliance. Therefore, we developed a robust geodatabase using hydrographic, land use and land tenure data. Riparian areas protected as Permanent Preservation Areas (PPAs) were delimited and their land cover mapped for 2012 and 2018 using high spatial resolution satellite images and unsupervised K-means classification method. We also applied landscape metrics to analyze riparian PPA structure and dynamics. Our results indicated that NVPL was followed by a downward trend in the riparian vegetation deficits in all land use types and property-sizes, but it did not stop new clearings. Although riparian PPAs in minifundios ($\leq$ 80 ha) and agrarian reform settlements tended to concentrate higher relative deforestation amounts, large-sized farms were responsible for most of the absolute extent of riparian deforestation in both years, accounting for 76–78% in QRC and 93% in SJX. They were also the main drivers of new clearings, for which account for 71% in QRC and 86% in SJX. The impact of land use on riparian deforestation was not homogeneous among properties, possibly reflecting different levels of technological investment and management techniques. In the so-called consolidated areas, in which the riparian PPA minimum width was reduced by NVPL, decreases in deforestation between 2012 and 2018 were lower. In these areas, vegetation coverage did not exceed 23% in any of the study areas. In the riparian PPAs that was not under consolidated areas, the vegetation coverage was of at least 85% of the area. Local environmental governance may also have affected the riparian deforestation dynamics, in which stricter monitoring and law enforcement lead to lower deforestation extents. Finally, landscape metrics revealed the importance of managing riparian areas at the landscape level, as local improvements did not necessarily result in connectivity gains.

1. Introduction

Functioning as interfaces between terrestrial and aquatic ecosystems, riparian areas maintain a natural disturbance gradient that supports a sensitive mosaic of environments almost unparalleled in other systems (Naiman and Décamps, 1997). Land use changes and agricultural management practices are usually associated with some kind of impact on riparian areas which, in turn, mitigates such effects on water bodies (Leal et al., 2016). Suppression of riparian vegetation can result in significant impacts on hydrological and biogeochemical processes (Sweeney et al., 2004; Deegan et al., 2011; Bleich et al., 2014), and also lead to structural and functional changes in biological communities, considering their role as habitat and as ecological corridors (Nagy et al., 2015; Elliott and Vose, 2016). Moreover, these systems are of particular

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interest for biodiversity conservation, as well as to maintain ecological processes and ecosystem services (Naiman and Décamps, 1997; Burdon et al., 2013; Bleich et al., 2014).

Many countries have national or regional environmental regulations to protect riparian areas (Chiavari and Lopes, 2017). In Brazil, vegetation strips associated with water bodies and headwaters are protected as Permanent Preservation Areas (PPAs). Restrictions on the use and occupation of riparian areas for conservation purposes have existed since the 1920s. However, the consolidation of norms that guide the definition, execution and monitoring of these areas took decades until the implementation of the Forest Code of 1965 (Federal Law no. 4771/1965) and its numerous amendments. Currently, the PPA protection regime is dictated by Federal Law no. 12,651/2012, also known as the Native Vegetation Protection Law (NVPL).

NVPL was the result of a great pressure, especially from the agribusiness sector, in favor of the reformulation of the Forest Code of 1965. The main arguments were based on the need to correct legal insecurities arising from the various amendments to the Forest Code, and on the difficulties of regularizing the legal environmental aspects of rural properties with the introduction of these amendments. There was also an implicit interest in the amnesty of penalties related to deforestation and in legalizing the maintenance of agricultural activities in protected areas illegally occupied (Brancalion et al., 2016). The approval of the NVPL brought significant flexibilities to the PPA protection regime, many of which were received with objections by the scientific community (Silva et al., 2012) and by the Brazilian Federal Prosecution Office (PGR) (2013).

In the PPA protection regime implemented since the Forest Code of 1965, here denoted as 'regular regime', vegetation strips must be maintained at a minimum width of 50 m for headwaters, and 30–500 m for fluvial channels, depending on their width. Deforestation within these limits implies mandatory full restoration, in addition to criminal penalties (Brazil, 2012a). However, NVPL created an exception to this regular regime, which enabled a set of more flexible requirements for riparian areas that were illegally deforested before July 22, 2008. In these so-called 'consolidated rural areas', instead of fully restore the PPA, the law requires only minor buffer strips to be recovered, ranging from 5 to 100 m in width, depending on the property size. In the remaining portions of the consolidated PPAs, where there is no restoration requirement, the maintenance of activities such as farming, forestry, ecotourism, and rural tourism are allowed, including all associated infrastructure (Brazil, 2012a, 2012b).

Despite their importance, studies that assess the impact of these changes on riparian protection are still scarce (Nunes et al., 2019). In addition to the lack of accurate mapping of water bodies and their widths for the whole country, access to high spatial resolution images, which is essential for mapping land cover in the narrow riparian PPAs, is quite limited (Taniwaki et al., 2018; Nunes et al., 2019). Furthermore, problems related to delimitation and overlapping of rural properties in the national environmental registries increases uncertainties related to PPA delineation (Freitas et al., 2017). Currently, it is estimated that 4.5 million hectares of PPAs must be restored throughout the country (Soares-Filho et al., 2014), which reinforces the importance of understand the impacts of NVPL on the protection of riparian areas.

The role of land tenure and land use on riparian deforestation dynamics also remains poorly understood. Several studies suggest a correlation between land concentration and large-scale deforestation in tropical forests (Souza et al., 2013; Assunção et al., 2017). In Brazil, despite a trend of deforestation, measured in absolute terms, being concentrated on large and medium-sized farms (Godar et al., 2014; L'Roe et al., 2016; Richards and VanWey, 2016), small-sized farms may have higher percentages of relative deforestation within their properties (Godar et al., 2012; Richards and VanWey, 2016). Environmental certifications and demands from some agricultural commodity supply chains may also play an important role in reducing illegal large-scale deforestation (Nepstad et al., 2014; Assunção et al., 2017). However, the maintenance of these relationships observed between land tenure and land use and the large-scale deforestation in the riparian PPAs is still uncertain. Understanding how property-size and agricultural activities can affect riparian deforestation is paramount to advancing and directing protection efforts considering the wide range of possible pressures and responses to the fulfilling of legal requirements.

Here we present an analysis of NVPL impact on riparian PPA deforestation dynamics, considering property size, land use and PPA protection regime (regular and consolidated). Riparian deforestation was assessed in two different periods: 2012, the year that NVPL was approved, and 2018, six years after its implementation. This is the first study of this kind, based on a robust set of hydrographic and land data, combined with high spatial resolution satellite images. Focusing on two representative areas of the Upper Xingu River Basin (UXRB), located in an Amazonian agricultural frontier, we sought to answer three important questions in the scope of environmental governance: (i) what was the impact of NVPL on deforestation and restoration dynamics in the riparian PPAs within the study areas?; (ii) were these effects observed similarly among different property-sizes and land use types?; and (iii) are deforestation and restoration dynamics homogeneous across PPAs in regular and consolidated regimes?

2. Materials and methods

2.1. Study area

The Xingu River is one of the main tributaries of the Amazon River. Its headwater region is located in Mato Grosso State in Brazil, where the basin is named Upper Xingu River Basin (UXRB), draining an area of 170,000 km^2 in the ecotone between the Amazon rainforest and the Cerrado biome (neotropical savannas) (ANA, 2006; IBGE, 2019b), two highly important biomes for global biodiversity and ecosystem service conservation (Mittermeier, 2004; Fearnside, 2008). The basin also contains important ethnic diversity, including 16 indigenous groups and other traditional communities (Velasquez et al., 2010). In one of the most active agricultural frontiers in the world in terms of vegetation suppression (FAO, 2006) and burning (Giglio et al., 2006), it is estimated that 8356 km^2 of Cerrado and 20,971 km^2 of Amazon rainforest were cleared in UXRB from 1995 to 2015 (Garcia et al., 2019).

The agricultural frontier in Mato Grosso stands out as one of a few to reach the final intensive stage in less than 40 years (Arvor et al., 2018). Until the early 2000s, the UXRB transitioned from a pioneer landscape to a consolidated frontier, switching from labor-intensive agricultural systems to one based on financial capital and integration into market. Agriculture intensification started to become more significant, increasing cultivated areas over pasturelands, reducing deforestation rates, and spreading double cropping as a common practice (Garcia et al., 2019). Between 2010 and 2019, soybean planted area in the main municipalities of the UXRB increased from 1,503,653 to 3,322,844 ha, followed by an increase in the average soybean yield from 2928 to 3392 kg/ha (Supplementary material 1). This intensification process may have been favored by several factors, including environmental regulations, technological changes, economic disincentives for deforestation, and/or market regulations (Gasparri and de Waroux, 2015). Meanwhile, the average effective of the herd fell by 15% in the basin, decreasing in 16 of the 25 municipalities considered (Supplementary material 1). Currently, UXRB supplies international and regional markets (Garcia et al., 2019), accounting for about 2% and 0.4% of the world's soy and livestock production, respectively (IBGE, 2017a; FAO, 2019).

Considering its socio-environmental importance, two municipalities in the UXRB were chosen for this study, representing the main socio-economic scenarios found in the basin (Fig. 1). Querência has a population of 17,479 inhabitants and an area of 17,786 km^2, while São José do Xingu is considerably smaller, with 5595 inhabitants and an area of 7459 km^2 (IBGE, 2018, 2019a). Both areas are covered mainly by Amazon rainforest, and Evergreen Seasonal Forest is the dominant

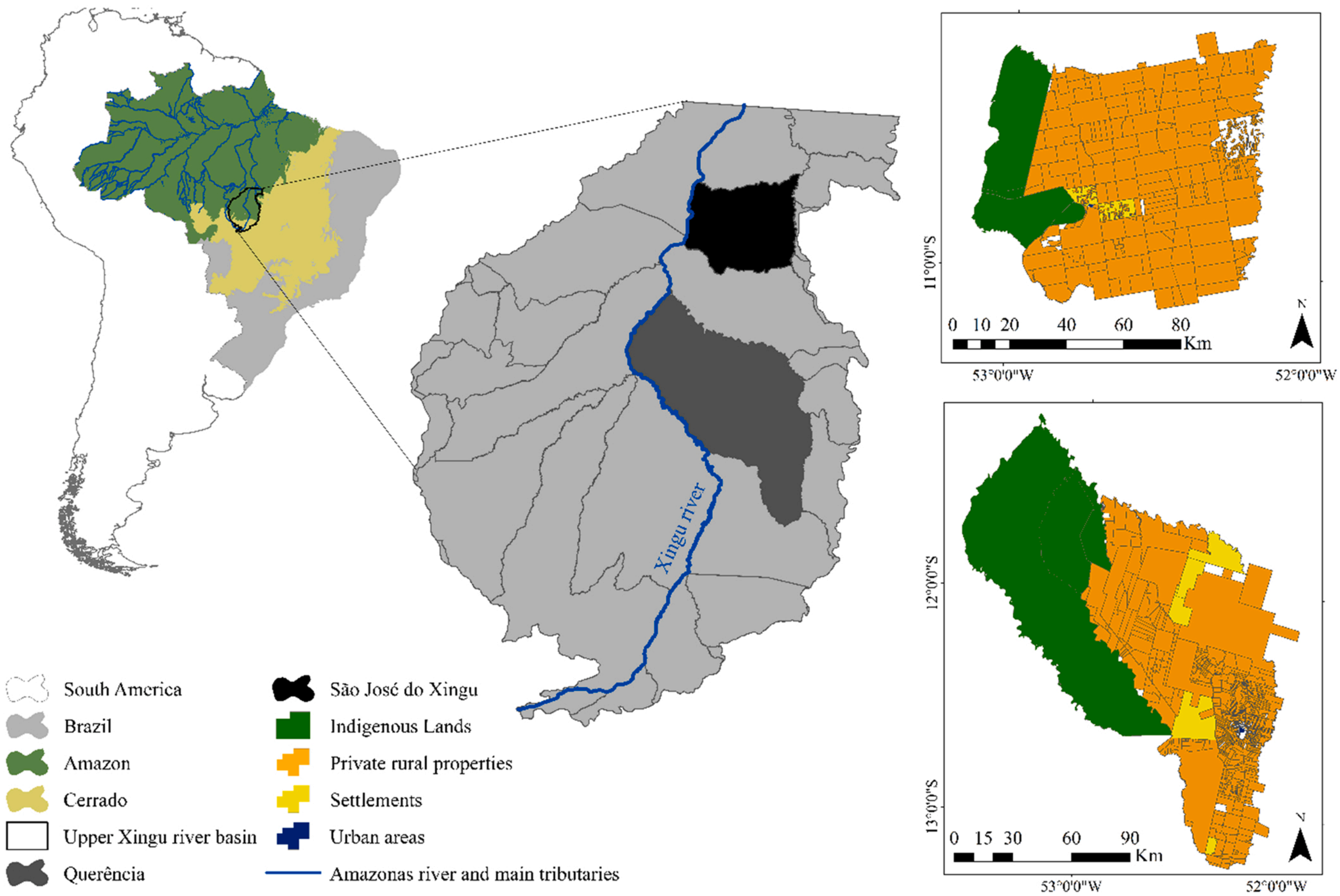

Fig. 1. Location of the municipalities of São José do Xingu and Querência, Mato Grosso, Brazil.

phytophysiognomy (IBGE, 2019b). Indigenous lands represent important components of these landscapes, covering about 41% of Querência and 18% of São José do Xingu. Land concentration is also a striking feature, considering that about 76% of the agricultural land in Querência, and 91% in São José do Xingu, is composed of large-sized farms (> 1200 ha) (Supplementary material 2).

As in several regions in the UXRB, Querência has been presenting a clear trend of decreases in cattle ranching and increases in grain production, especially soybean. In 2018, the cattle herd reached 103,138 heads, almost 50% less than in 2008 (IBGE, 2009a, 2019c), while soybean area increased 108%, reaching 350,000 ha (IBGE, 2009b, 2019d). Currently, Querência is the fourth largest soybean producer in Mato Grosso state, and has the fourth largest gross domestic product (GDP) per capita (IBGE, 2017b, 2019d). In this sense, the municipality is a great example of the crop expansion and intensification process in the basin, also presenting a remarkable history of deforestation, being responsible for the second highest deforestation rate in the state (INPE, 2019).

In São José do Xingu, cattle ranching remains one of the main economic activities, comprising a herd of 216,752 heads in 2018 (IBGE, 2019c). Covering about 80,000 ha in 2018, soybean expansion is recent and still discreet compared to other regions in Mato Grosso (IBGE, 2019d). Deforestation in São José do Xingu is also noteworthy, reaching the equivalent of 58% of the municipal area by 2018 (INPE, 2019).

2.2. Land tenure and agricultural land use

For this study, we developed a modified version of the land tenure map provided by Freitas et al. (2017), which offered a careful treatment of overlapping property problems in the cadastral databases of the National Institute of Colonization and Agrarian Reform (INCRA) and the National System of Rural Environmental Registration (SICAR). We only maintained polygons related to private rural properties or settlement projects. Manual corrections were made by visual interpretation and by consulting the official SICAR database, adding registered missing properties with no overlaps and adjusting polygons with confusing boundaries.

With the exception of settlements related to agrarian reform programs, rural properties were classified according to their size as follows: minifundios (≤ 80 ha), small-sized farms (80–320 ha), medium-sized farms (320–1200 ha) and large-sized farms (> 1200 ha), as defined by Federal Law 8629/1993. These values were originally defined in fiscal modules, a measure which varies amongst municipalities, being fixed at 80 ha for the study areas (INCRA, 2013). To facilitate understanding, values defined in fiscal modules are described only in hectares in this study. Rural properties were also classified according to their main land use, using a 2015 map (Garcia et al., 2019). Therefore, the proportional area occupied by each type of agricultural land use was defined per rural property, which in turn was categorized as (i) crop, (ii) livestock, and (iii) crop-livestock farms (Supplementary material 3).

2.3. Riparian permanent preservation areas

We only considered fluvial channels and headwater PPAs, located in private rural properties, within which vegetation cover is mandatory. Areas corresponding to small dams were disregarded, as well as the portions of consolidated PPAs where there is no vegetation restoration obligation. For delimiting riparian PPAs, water surface areas were extracted from a high spatial resolution land-cover map derived from the digital classification of Rapid Eye satellite images, as described in

Section 2.4. The water surface areas were complemented by hydrographic network maps from the Brazilian Foundation for Sustainable Development (FBDS, 2018) and the Ministry of the Environment and the Brazilian Army (MMA/DSG-EB, 2008). Manual corrections were made based on the SICAR database and the visual analysis of the Rapid Eye images, with the inclusion of water bodies that were not present in these hydrographic network maps, and a review of delimitations. Finally, channel widths were manually defined using five classes, as specified by the NVPL: 0–10, 10–50, 50–200, 200–600 and > 600 m (Brazil, 2012a).

Minimum width requirements for riparian vegetation were defined according to the PPA protection regime (i.e., regular or consolidated). In headwaters, minimum widths were fixed for each regime. Fluvial channel PPAs in the regular regime were delimited according to the channel width classes (Table 1). As there were no channels over 600 m wide in the study areas, this PPA category was not represented.

Consolidated areas were defined based on a consolidated land-use map, provided by the Life Center Institute (ICV), in which PPA minimum width varied according to property size (Table 1). Consolidated PPAs located in properties over 320 ha can have minimum widths ranging from 20 to 100 m (Brazil, 2012a, 2012b). In order to simplify the analysis, their width was fixed at 20 m. Fig. 2 shows PPA boundaries in a section of Querência and exemplifies their variability in the landscape according to the protection regime.

2.4. Land-cover classification

Land-cover maps were derived from the digital classification of Rapid Eye satellite images, with a spatial resolution of 5 m, provided by Planet Labs (https://www.planet.com/). These images were obtained orthorectified and geometrically and radiometrically corrected (Planet, 2019). A total of 64 scenes were used for Querência and 38 for São José do Xingu (Supplementary material 4). The images were acquired in the dry season (May–September) for the years 2012 and 2018, with supplementary images acquired from 2011 and 2019. These periods correspond to the NVPL implementation year and six-year anniversary of this milestone, respectively.

To derive these maps, we used an unsupervised K-means classification method. The scenes with the same acquisition date were mosaicked and classified together, with a limit of 15 iterations and 60 classes. The areas comprising riparian PPAs (Section 2.3) were extracted to assess land-cover per year exclusively within riparian areas. The classes resulting from this classification were then grouped into (i) natural cover and (ii) vegetation deficits (Table 2) based on factors such as shape, texture, spatial arrangement, hue and coloring.

2.5. Accuracy assessment

Reference points for the land-cover maps accuracy assessment were obtained through visual interpretation of high spatial resolution images. Based on the work of Garcia et al. (2019), which collected 2000 ground truth points throughout the UXRB, we proportionally established 100 points in São José do Xingu and 200 in Querência. These points were randomly generated within the riparian PPAs in equal numbers for the two classes (natural cover and vegetation deficits), using the ArcGIS 10.4 Information System. The visual interpretation was carried out by two experienced analysts, based on 2018 Rapid Eye images and an auxiliary image database from the Google Earth platform. Producer's and user's accuracy values, overall accuracy and Kappa index can be found in Supplementary material 5. Kappa indexes for the Querência (0.76) and São José do Xingu (0.86) land-cover maps were satisfactory, according to the reference values suggested by Landis and Koch (1977).

Table 1
Categories and PPA minimum width[a] in the regular and consolidated regimes, applied in the present study based on Federal Law no. 12,651/2012 and its amendments approved by Federal Law 12,727/2012.

Regular regime		Consolidate area	
Fluvial channel width (m)	PPA minimum width (m)	Property size (ha)	PPA minimum width (m)
0–10	30	≤ 80	5
10–50	50	80–160	8
50–200	100	160–320	15
200–600	200	> 320	20
Headwater	50	Headwater	15

[a] PPA minimum width refers to both margins of the fluvial channel or a radius around the headwater.

2.6. Deforestation in riparian areas and implications for the landscape level

Vegetation deficits within riparian PPAs detected through land-cover maps were analyzed in terms of (i) spatial location (municipality), (ii) temporal location (year), (iii) property types, (iv) land use, and (v) protection regimes. Fig. 3 outlines these steps, as well as data sources and categories evaluated in each of the elements mentioned above (i–v).

A set of landscape metrics was used to characterize and analyze riparian PPA structure and dynamics using the high spatial resolution land-cover maps. Selected composition and configuration metrics included: total class area (CA), percentage of landscape of class (PLAND), number (NP) and density (PD) of patches, largest patch index (LPI), total edge (TE), edge density (ED), and Clumpiness Index (CLUMPY) (Supplementary material 6). All metrics were derived at the class level using the 'landscapemetrics' 1.2.2 package (Hesselbarth et al., 2019) in R software (R Core Team, 2020).

3. Results and discussion

3.1. Impacts of land tenure and land use on riparian deforestation

We analyzed 28,362 ha of riparian forests within 627 rural properties distributed across Querência, and 25,973 ha within 310 rural properties across São José do Xingu. In 2012, vegetation deficits were found in 11% and 29% of these areas, respectively (Fig. 4). The distribution of these vegetation deficits among the different types of rural properties and agricultural land use is detailed in Supplementary materials 7 and 8.

Large-sized farms encompassed most of the PPAs and presented the largest vegetation deficits in both periods. In 2012, these properties were responsible for a cleared area equivalent to 8% of total PPA area in Querência, and 27% in São José do Xingu. The second most important contributor was settlements in Querência and medium-sized farms in São José do Xingu, responsible for a vegetation deficit of 1% of the total area of the PPAs in each municipality. Vegetation deficits for all other property types accounted for 1%. Although most of the PPAs in Querência were equally distributed between crop and crop-livestock farms, the latter accounted for most of the clearings, representing 7 of the 11% of vegetation deficit in PPAs in 2012. In São José do Xingu, the largest vegetation deficit was found on livestock farms, which comprise most of the PPAs, responsible for 24 of the 29% of vegetation deficits in 2012.

The relative vegetation deficits (i.e., vegetation deficits per PPA area in each category) were also higher in São José do Xingu. Overall, higher values were found in settlements and minifundios compared to other property types (Fig. 5a). This relationship between settlements and small-scale agricultural productions with high relative vegetation deficits is not new (Nunes et al., 2015; Simões et al., 2017; Farias et al., 2018; Zimbres et al., 2018). Settlement projects often lack proper planning, which can lead to high and ineffective investments, such as extensive deforested areas for the construction of side roads, or to environmental and income problems when the settlement is placed on very sandy or swampy soils, on unfavorable relief areas or over primary forest cover (Soares, 2008). In addition, both settlers and small producers often do not have sufficient logistical, technical and financial

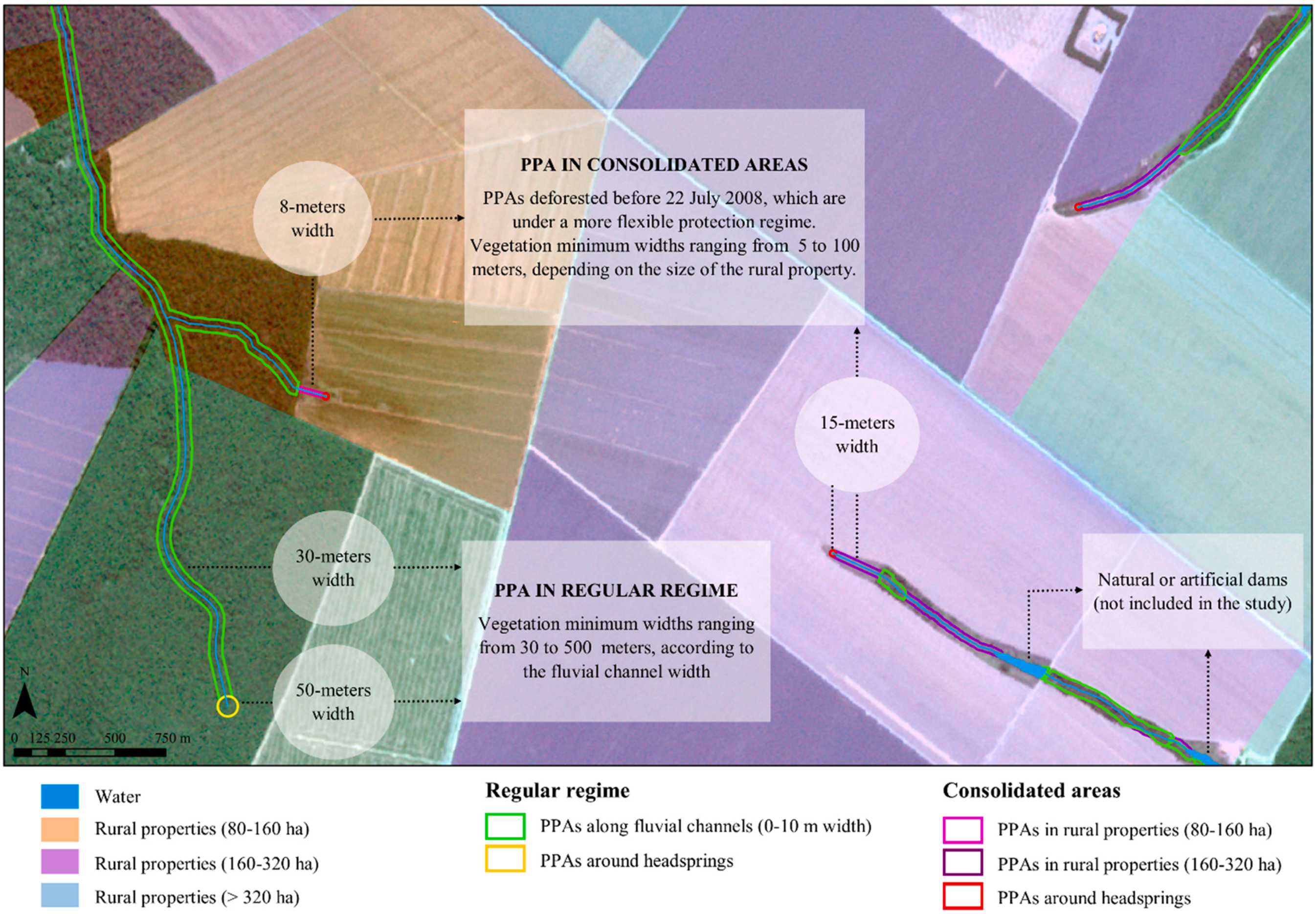

Fig. 2. Riparian PPAs in a section of Querência, exemplifying changes in their minimum width according to the protection regime.

Table 2
Land-cover classes applied in the analysis of the vegetation deficits in riparian PPAs.

Land-cover class	Description
Natural cover	Areas whose coverage indicates compliance with PPA use restrictions required by NVPL, including areas of forest cover, natural non-arboreal vegetation (e.g., floodplain areas), and water surfaces
Vegetation deficits	Areas lacking natural cover, where natural arboreal or non-arboreal cover was cleared and replaced by another cover type, such as croplands, pasturelands, and exposed soil

support (Steward, 2007; Soares, 2008; Cardoso, 2011). This can lead to a number of implications, including low productivity, illegal deforestation, financial problems, land concentration and marginalization (Fatorelli and Mertens, 2010; Alencar et al., 2016). Under these conditions, settlers and small producers commonly end up renting their land to larger ones (Alencar et al., 2016), making it difficult to distinguish those truly responsible for deforestation in these areas.

In terms of land use, the relative vegetation deficits in PPAs varied broadly across municipalities (Fig. 5b). While in Querência the highest and lowest values were presented by crop-livestock and crop farms, respectively, the opposite trend was found in São José do Xingu. These patterns can be a direct result of market pressures on the soybean chain in the Amazon, subject to several requirements in the biome through the Soy Moratorium (SoyM). This was the first voluntary zero deforestation agreement implemented in the tropics, in which the largest representative entities of the sector stopped soybean commercialization from deforested areas after July 2006. Although it was not enough to ensure deforestation-free soy production (Rajão et al.), the SoyM significantly reduced soy expansion over native vegetation cover (Rudorff et al., 2011; Gibbs et al., 2015). The lower relative vegetation deficits found in crop farms in Querência may be due to the municipality's position as the fourth largest exporter in the state, especially in the soybean trade. On the other hand, the SoyM may not have such an important role in São José do Xingu, given its low soy production and export rate (MDIC, 2020).

NVPL was certainly followed by a downward trend in the riparian vegetation deficits. In 2018, six years after the approval of NVPL, both municipalities reduced the vegetation deficits in the PPAs. In Querência, this drop ranged from 18% to 22% among property types. Settlements were the only property category to deviate from this range, reducing their deficits by just 3%. In São José do Xingu, the reduction was slightly lower, ranging from 12% to 17%, except on small-sized farms, which decreased by more than 27% (Supplementary material 7). The main agricultural land use in each municipality showed the highest reductions in deficit extent. However, relative vegetation deficits dropped more significantly in the lower extension categories (Supplementary material 8). Therefore, in Querência, while crop and crop-livestock farms underwent decreased PPA clearings by 20% and 18%, respectively, livestock farms stood out with a reduction of almost 34%. In São José do Xingu, the reduction was higher in crop farms (24%), followed by crop-livestock (20%) and livestock farms (16%).

To understand these patterns, first we must emphasize that although the number of farms (Table 3) and the area with vegetation deficits in riparian PPAs have decreased, NVPL did not stop the advance of new deforestation areas. The reduction of vegetation deficits previously presented reflects the balance between restoration of the vegetation

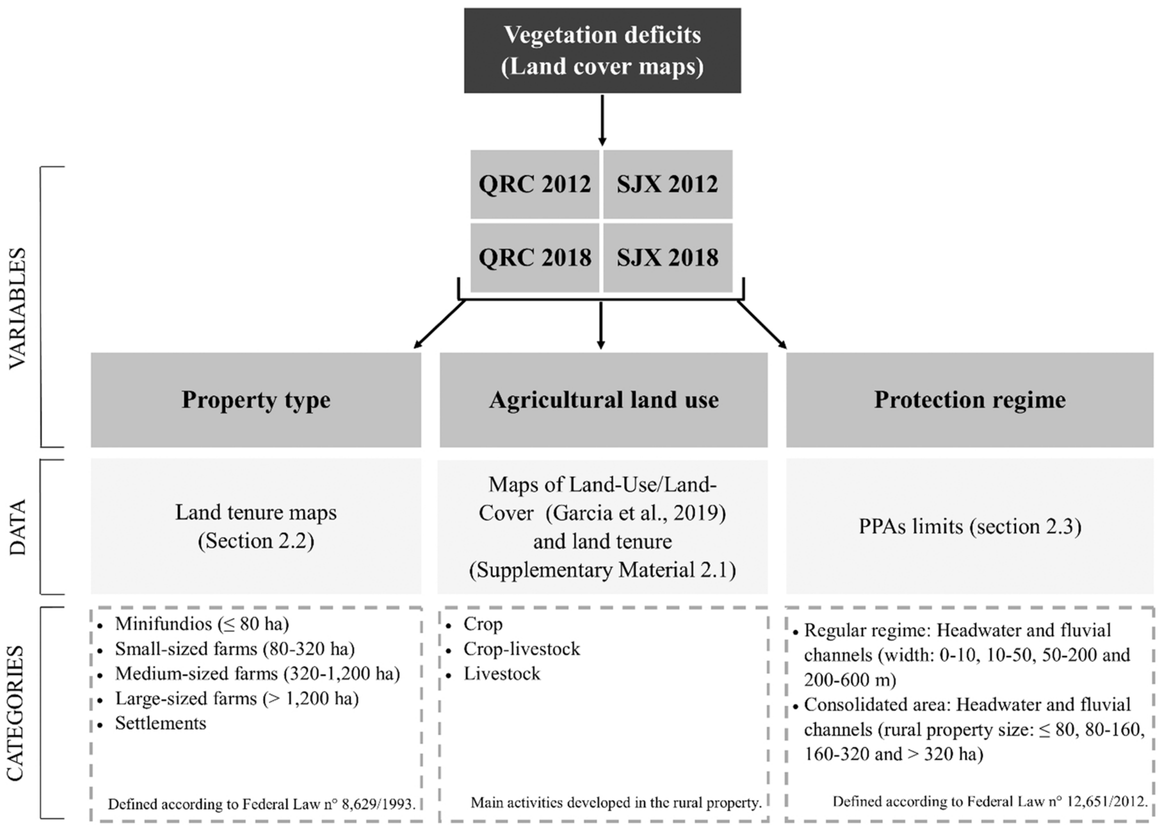

Fig. 3. Methodological steps highlighting (i) variables used to assess deforestation patterns in riparian PPAs across Querência (QRC) and São José do Xingu (SJX), (ii) sources and treatment of the employed data, and (iii) categories evaluated for each selected variable.

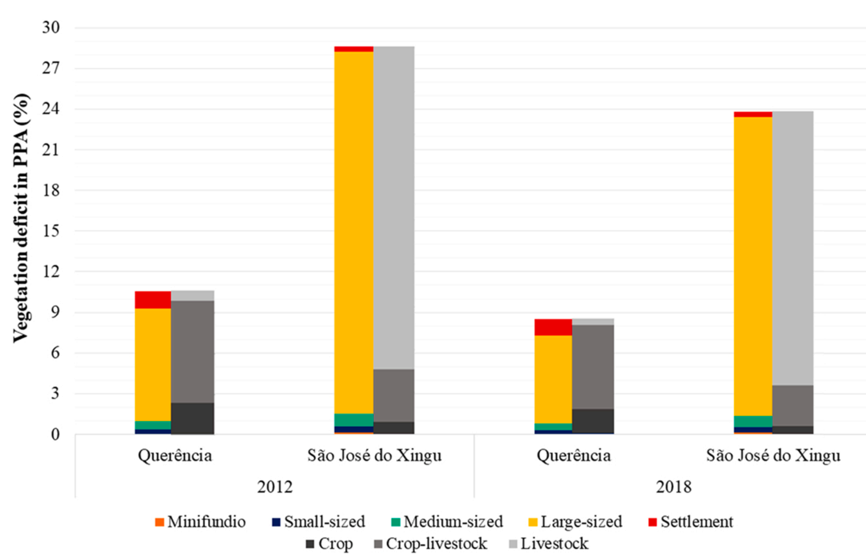

Fig. 4. Vegetation deficits in riparian PPAs in Querência (QRC) and São José do Xingu (SJX) during 2012 and 2018, considering different types of rural property and agricultural land use.

deficits observed in 2012 and new clearings of riparian vegetation (Fig. 6). This restoration process accounted for an area of 825 ha (30% reduction) in Querência, and 1729 ha (23% reduction) in São José do Xingu, while new deforestation in riparian PPAs reached 293 ha (11% increase) and 468 ha (6% increase), respectively.

The extent of new clearings in settlements was close to the extent restored, especially in Querência, leading to the low reduction in vegetation deficits previously observed. In the Action Plan for Prevention and Control of Deforestation in the Legal Amazon (PPCDAm), one of the reference policies for controlling deforestation in the biome, settlements were identified as a critical factor, where the relative vegetation deficit reduction would be below that observed in other property types (MMA, 2013). However, regarding riparian PPAs, our results reinforce that large-sized farms must be the main focus of strengthening monitoring and environmental enforcement actions. The sum of deforestation extents in riparian PPAs in all other property types is minimal compared to that observed on large-sized farms, which were also the major drivers of deforestation advancement over new riparian areas in all types of land

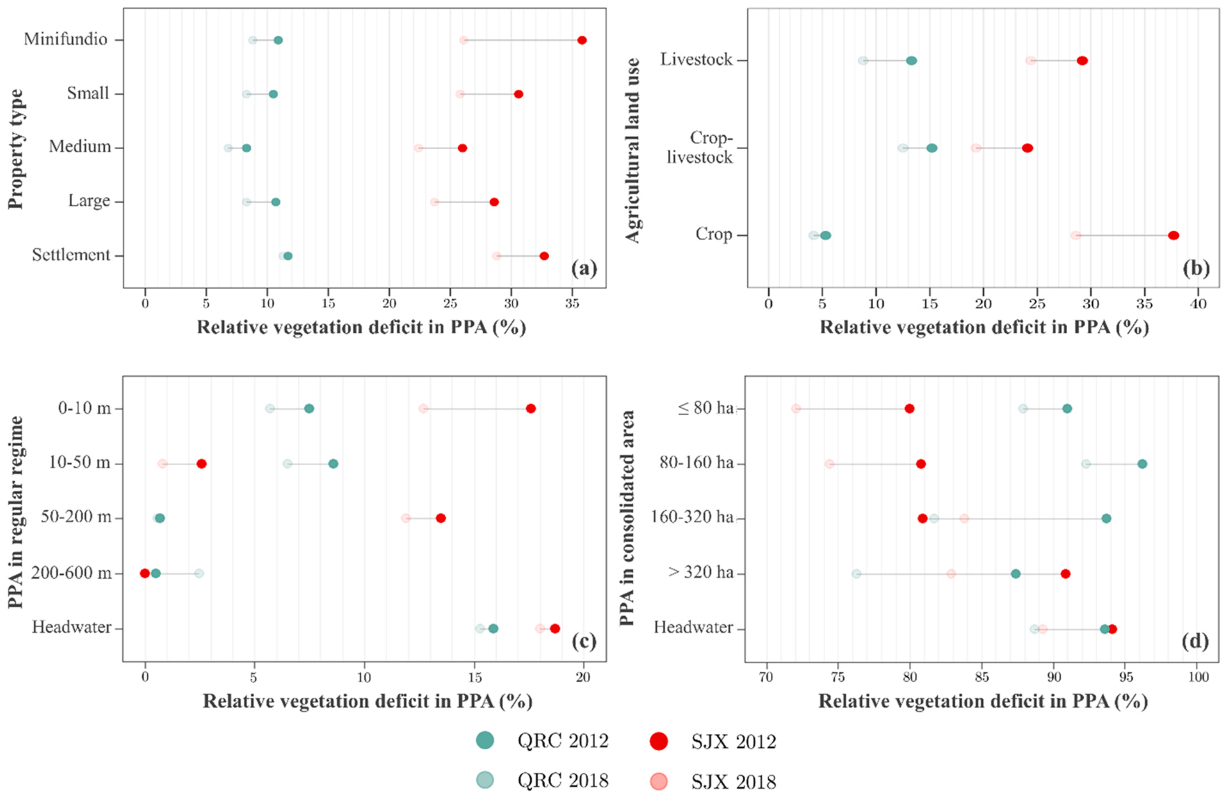

Fig. 5. Relative vegetation deficits in terms of (a) property size, (b) land use, and (c, d) PPA protection regimes.

use, accounting for 71% of new clearings in Querência and 86% in São José do Xingu.

Especially considering Querência, where large-sized and crop farms were the only categories to increase the number of properties with vegetation deficits (Table 3), it is important to discuss some weak points of the SoyM policy in the context of riparian areas. The spatial resolution of the monitoring is not adequate to supervise narrow strips of vegetation, covering only clearings larger than 25 ha (ABIOVE, 2019). Considering that the average deforestation polygon size in our study areas did not exceed 1 ha (Supplementary material 9), it is likely that several areas of riparian deforestation end up going unnoticed in SoyM's monitoring. In addition, a farm out of compliance with the NVPL will not necessarily be subject to sanctions, as this policy regulates only the areas covered by soybeans and not the whole property (ABIOVE, 2014; Gibbs et al., 2015). Illegal deforestation that occurs in areas with other types of land use will not result in the property embargo. SoyM also does not adequately account for the complexity of property ownership in Mato Grosso. It is common for a producer who used multiple properties for production to provide documentation about only a single property, which may also allow the sale of soybeans from embargoed areas (Rausch and Gibbs, 2016; Gollnow et al., 2018). Thus, limitations both in the spatial resolution and in the monitored area may have reduced SoyM's inhibitory effects on riparian deforestation and reflected in the deforestation dynamics observed in the large-sized and crop farms of our study areas.

In Querência, minifundios also showed a considerable increase in new deforested areas, especially in crop-livestock farms. However, the impact of this increase over the total reduction in vegetation deficits was mitigated by restorations in crop farms, which were responsible for most of the 2012 deforestation within the minifundio category (Supplementary material 10). In São José do Xingu, minifundios and small-sized farms had a slightly different dynamic. The former presented newly cleared areas largely surpassed by the restored ones, in addition to the highest vegetation deficit reduction, and small-sized farms paid off all their vegetation deficits in crop-livestock farms and most of it in crop farms. This is a positive trend, especially considering that crop farms had a relative vegetation deficit of almost 50% in riparian PPAs in 2012 (Supplementary material 10). This reduction was only dampened by livestock farms, which were also responsible for a significant amount of the vegetation deficits and presented new clearings in 2018. In fact, new riparian deforestation areas were observed in all types of livestock farms.

The relationship observed between riparian deforestation and livestock farming systems was expected. Riparian vegetation is commonly removed to provide access for cattle to the water courses. Riparian areas associated with pasturelands are also frequently subjected to trampling and consumption of remaining vegetation, especially when not demarcated by fences (Kauffman and Krueger, 1984; Lees and Peres, 2008; Nunes et al., 2015). However, its association with higher riparian deforestation rates does not seem to be a rule. In 2018, livestock farms in Querência showed the lowest extensions of new deforestation and the highest extensions of restoration. On the other hand, deforestation in crop-livestock farms was higher than that observed in crop farms.

There are several factors that may have contributed to this scenario. First, considering the higher levels of technological investment in Querência (Supplementary material 11), it is possible that these livestock-farms have a higher occurrence of management practices that contribute to the recovery of riparian PPAs in relation to those in São José do Xingu, such as the use of fences surrounding riparian vegetation, and an out-river water source that allows cattle not to enter the PPAs to drink water. Second, it is also possible that a larger number of livestock-farms have joined municipal programs for environmental recovery, such as the "Querência + Sustainable Landscapes" program (IPAM-ISA, 2017; Simões et al., 2017). Finally, considering the large conversion of degraded pasture areas for grain production in Querência (Simões et al., 2017), it is possible that part of the crop-livestock farms represents these transition zones, encompassing deforested areas derived from a previous type of land use. However, confirmation of any of these possibilities

Table 3
Number of farms with vegetation deficits in riparian PPAs in Querência (QRC) and São José do Xingu (SJX) in the years of 2012 and 2018, highlighting those that recovered some of the deforested PPA areas observed in 2012 but still presenting new clearings, and those that only presented vegetation deficits in 2018.

Property type		No. of farms with vegetation deficits		No. of farms with some recovery in the 2012 vegetation deficits		No. of farms with vegetation deficits detected only in 2018
		2012 (%)	Change in 2018 (%)	Total (%)	With new clearings (%)	Total (%)
QRC	Minifundio	60	-4	89	54	8
	Small-sized	48	-1	85	59	11
	Medium-sized	69	-5	90	68	4
	Large-sized	80	+ 5	97	96	6
	Settlement	100	0	100	100	–
SJX	Minifundio	97	-3	86	64	0
	Small-sized	93	-2	83	68	3
	Medium-sized	94	-3	100	94	3
	Large-sized	100	0	100	100	–
	Settlement	100	0	100	100	–
Land use type						
QRC	Crop-farm	58	+ 1	87	68	8
	Crop-livestock farm	77	-5	95	77	3
	Livestock-farm	68	-4	91	69	3
SJX	Crop-farm	100	0	100	100	–
	Crop-livestock farm	100	0	100	100	–
	Livestock-farm	98	-2	92	83	0

requires further investigations.

3.2. Protection regime

In Querência, 97% of riparian PPAs remained in the regular regime, which also comprised the largest extent of the vegetation deficits, mainly located along 0–10 m wide fluvial channels (Fig. 7). In São José do Xingu, riparian PPAs around these water bodies also showed high deficits compared to the other PPA categories in the regular regime, but most of the vegetation deficits were found in consolidated areas. This protection regime encompasses 17% of the riparian PPAs in the municipality compared with 3% in Querência. In São José do Xingu, properties larger than 320 ha covered 54% of all vegetation deficit found in 2012. In Querência, this category also presented the second largest vegetation deficit, along with 10–50 m wide fluvial channels in the regular regime (Supplementary materials 12 and 13).

Consolidated PPAs also encompassed the highest relative vegetation deficits in both municipalities (Fig. 5cd). In Querência, these values ranged from 87% in properties larger than 320 ha, to 96% in properties between 80 and 160 ha. In São José do Xingu, relative vegetation deficits varied from 80% in properties with less than 80 ha, to 94% in headwaters. In the regular regime, headwaters also had the highest relative vegetation deficit in Querência (16%) and in São José do Xingu (19%). Considering all types of PPA in the regular regime, the relative vegetation deficits were higher in São José do Xingu (15%) than in Querência (7%).

In 2018, a decrease in vegetation deficits was detected in almost all PPA categories (Fig. 7). The regular regime contained most of the reductions, in which the decline ranged from 22% in Querência to 27% in São José do Xingu (Supplementary materials 12 and 13). This reduction was higher in 0–10 and 10–50 m wide fluvial channels, and lower in the headwaters. The only category in the regular regime to present an increase (0.03%) in vegetation deficits was 200–600 m wide fluvial channels. In the consolidated areas, decreases in riparian vegetation deficits did not exceed 13% in any category. In fact, vegetation cover did not surpass 28% in any category of consolidated PPAs.

This dynamic raises several questions about the weight of gains and losses for the protection of riparian areas within the consolidated regime. The suspension of sanctions and loosening up of legal requirements in effect in this regime were justified as necessary to facilitate the regularization of properties that had failed to comply with the Forest Code of 1965, and to improve social justice of the environmental legislation by reducing conservation requirements in small rural properties (Brancalion et al., 2016). However, the measure was regarded with concern due to the feeling of impunity that could result from this amnesty (Soares-Filho et al., 2014; Roriz and Fearnside, 2015; Brancalion et al., 2016) in addition to the low number of fines executed for crimes of illegal deforestation on private properties (MMA, 2006; IBAMA, 2019). Several studies also point out the inefficiency of such narrow buffer strips in maintaining ecosystem services that justify the very existence of PPAs (Metzger, 2010; Ramos and Anjos, 2014; Guidotti et al., 2020). In fact, consolidated areas brought considerable losses in

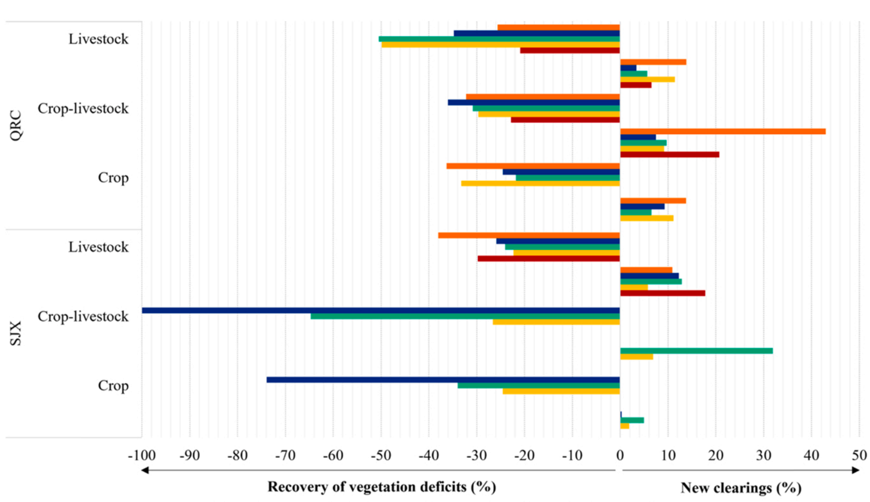

Fig. 6. Recovery of 2012 vegetation deficits (negative values) and new clearings (positive values) in the PPAs of Querência (QRC) and São José do Xingu (SJX) during 2018. Values weighted in relation to the 2012 relative vegetation deficits.

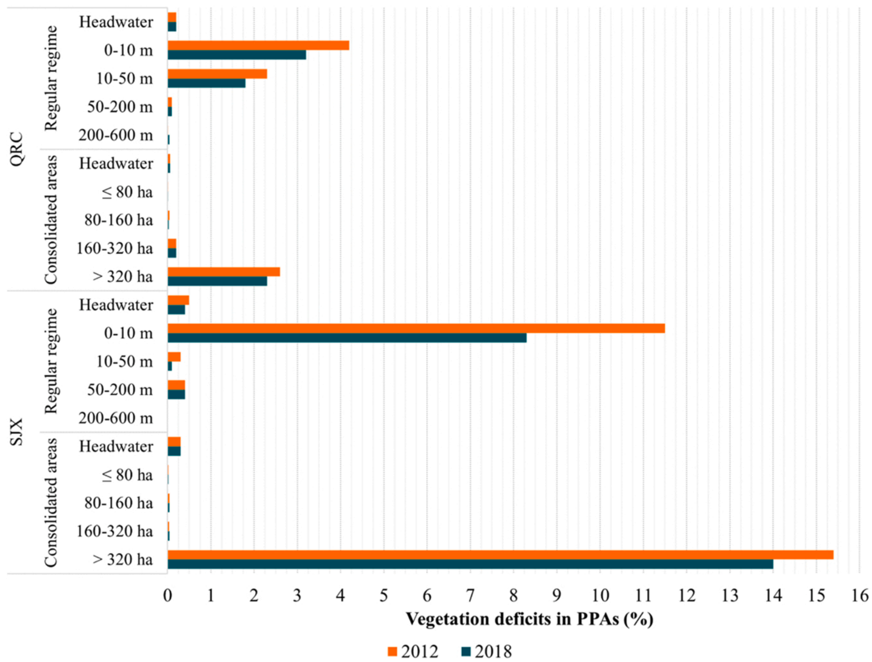

Fig. 7. Vegetation deficits in the riparian PPAs across Querência (QRC) and São José do Xingu (SJX), considering PPA categories in the regular and consolidated regimes.

PPA strips in both study areas. We estimated that about 1021 ha (2%) of the riparian PPAs in Querência and 3309 ha (10%) in São José do Xingu were included in the consolidated regime without mandatory restoration, i.e., exceed the PPA minimum width that must be restored, in which the maintenance of farming, forestry and tourism activities is now allowed.

Although such losses in PPA area significantly reduce the vegetation deficits to be recovered in order to regularize farms in non-compliance with NVPL, improvements in vegetation cover in the consolidated areas were much lower than those found in the PPAs under the regular regime. Considering that the consolidated PPAs encompass the large relative vegetation deficits in our study areas, they were a critical category in which restoration efforts should have been more intense. In addition, consolidated PPAs still present new clearings, even though they occurred mainly in the regular regime PPAs (Fig. 8). In this sense, we raised three major aspects that should be considered regarding the implementation of the consolidated regime in riparian PPAs. First, it favored areas of more pronounced non-compliance with environmental laws and, although it should have facilitated the regularization of the properties by reducing the restoration requirements, the strategy has so far resulted in lower rates of recovery of vegetation deficits. Second, the non-exclusion of the benefits of those who end up repeating the transgressions (i.e., new clearings) is a huge gap left by the NVPL, and can lead to the dangerous logic that the consolidated areas are a right and there is no additional duty to the owners in maintaining the environmental regularity of their property, making it even more unfair to those who fully comply with the environmental laws. Finally, it is possible that the regular regime PPAs become the main focus of new clearings, which have a larger set of use restrictions and, generally, a wider vegetation buffer width.

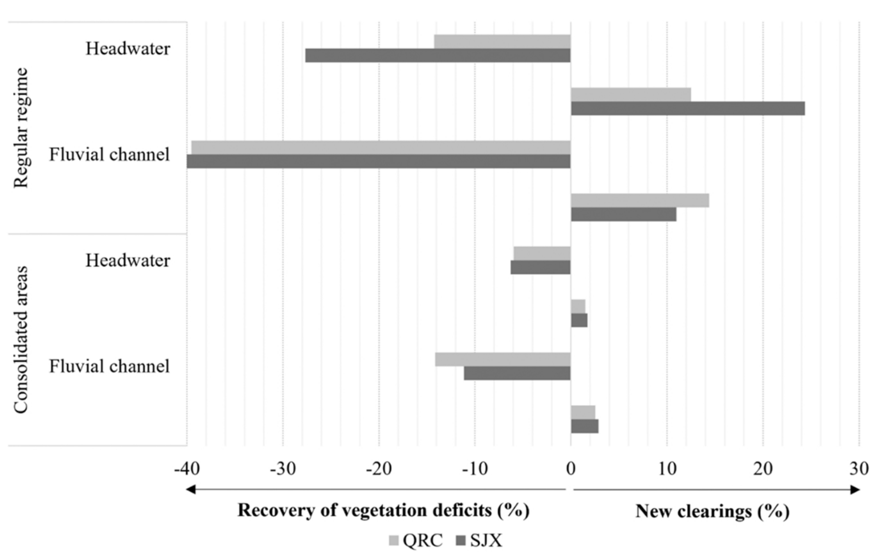

Fig. 8. Recovery of 2012 vegetation deficits (negative values) and new clearings (positive values) in the PPAs of Querência (QRC) and São José do Xingu (SJX) during 2018. Values weighted in relation to the 2012 relative vegetation deficits.

As the headwater region of the Xingu River basin, the high incidence of riparian deforestation in the headwaters of Querência and São José do Xingu also requires attention. In addition to their high relative vegetation deficits and low recovery of vegetation cover, the headwaters had high concentrations of new clearings (Fig. 8). A similar pattern was found in the north of Mato Grosso State, where headwaters were more deforested than any other riparian PPA category (Zimbres et al., 2018). Both headwaters and first-order rivers represent key elements for maintaining water flow and quality, as well as biodiversity, for the entire drainage network (Lowe and Likens, 2005). In the study areas, the intensity of agricultural use in these water bodies (Ballester et al., 2020) was certainly a key factor for the extent of deforestation detected and pointed out their vulnerability to anthropic impacts and the increasing threats to water security.

In addition to the anthropic pressure, first-order rivers and headwaters are often underestimated due to mapping limitations. While the recommended minimum spatial resolution for detecting these water bodies is 1:10,000, the majority of available databases are at best on the scale of 1:25,000 or 1:50,000 for most of the Brazilian territory (Taniwaki et al., 2018). In the present work, about 1265 km of 0–10 m wide fluvial channels identified through visual analysis of Rapid Eye images were not in the official databases or were not well represented. This underestimation limits monitoring and enforcement efforts and minimizes the need to restore riparian PPAs (Taniwaki et al., 2018), which reinforce the importance of investing in a cartographic database at the adequate scale for the implementing and monitoring of any protection policy for riparian areas.

3.3. Landscape metrics and municipal contexts

In order to explore riparian deforestation dynamics at a landscape level, we calculated several landscape metrics in the riparian PPAs of Querência and São José do Xingu and evaluated their changes before (2012) and after (2018) NVPL approval (Table 4). In 2012, in addition to a larger extent of vegetation deficits (CA and PLAND), the PPAs in São José do Xingu presented a much more fragmented structure compared to Querência. In this sense, the municipality presented a great number (NP) and density (PD) of riparian vegetation patches, leading to an increase in forest edge (TE and DE) and less aggregated patches (CLUMPY). As ecological corridors, riparian PPAs facilitate species flow among patches of forest remnants, playing an important role in the conservation of biodiversity in rural matrices. In addition to effects on the dynamics and maintenance of ecosystem services, these impacts on the connectivity of riparian PPAs may end up enhancing the effects of habitat loss and landscape fragmentation (Lees and Peres, 2008).

In addition to their differences in agricultural production and technological investment (Supplementary material 11), the municipalities also have different environmental governance trajectories, which probably played an important role in their difference in the extent of deforestation and structure of riparian buffers. In 2007, Querência was included in the first list of Priority Municipalities in the Amazon (Brazil, 2007; MMA, 2007; INPE, 2019). Known as the blacklisted municipalities, they were responsible for about 45% of deforestation in the biome and became a target of efforts to repress illegal clearings. The blacklisted municipalities went through an intensification of not only law enforcement and monitoring activities, but also economic sanctions and political pressures (Assunção and Rocha, 2019). The Central Bank of Brazil also started to demand supporting documentation of environmental regularization for purposes of agricultural financing in the biome (CBB, 2008), impacting credit concessions.

These measures generated great mobilization in Querência, where farmers were already experiencing trade restrictions from the SoyM, being widely impacted by the new credit and financing restrictions. Command and control activities ended up being complemented by a series of actions arising from the articulation among farmers, public authorities and organized civil society. Due to this mobilization, the deforestation rate dropped by 60% in 2011 compared to the average from 2004 to 2007 (Simões et al., 2017), and Querência became the first municipality in Mato Grosso to be excluded from the list (MMA, 2011). São José do Xingu was not included in any of the lists of Priority Municipalities, even though it had a prominent position in the state's deforestation ranking (INPE, 2019). Considering that the SoyM would have no major impact in a municipality focused on extensive livestock, the pressure on farmers to comply with environmental regulation was quite different between these municipalities.

Therefore, the increases in riparian vegetation in São José do Xingu in 2018 were very promising, presenting a similar rate to that of Querência (Table 4). The change was reflected in the structure of its riparian PPAs in the landscape, reducing their fragmentation (lower values for NP, PD, TE and ED, and increased value for CLUMPY). On the other hand, Querência followed an opposite trend. Despite the increases in riparian vegetation cover (CA and PLAND), the structure of the PPAs in the landscape showed increased fragmentation, presenting growth in the number (NP) and density (PD) of riparian vegetation patches, increased forest edge (TE and DE) and less aggregated patches (CLUMPY). This is a possible reflection of the new clearings in Querência, which were probably concentrated more in some regions. Instead of several small deforestations, not affecting the connectivity among PPAs, these new clearings may have occurred more in the perpendicular sense, segmenting riparian corridors.

It means that local improvements in vegetation cover will not necessarily result in gains in connectivity and protection of ecosystem services. As important tools for the consolidation of connectivity networks in anthropized landscapes (Lees and Peres, 2008; Zimbres et al., 2018), these results highlight that riparian areas should be manage at the landscape level. Moreover, as different responses were found for environmental legislation comply according to regional contexts, the management of recovery of riparian PPAs needs to occur at multiple scales and take into consideration regional heterogeneities to better target efforts and maximize their efficiency (Leal et al., 2016; Luke et al., 2019; Guidotti et al., 2020).

Table 4
Landscape metrics values obtained from land-cover maps of Querência (QRC) and São José do Xingu (SJX). More details on each metric can be found in Supplementary material 6.

	Year	Land-cover class	CA	PLAND	NP	PD	LPI	TE	ED	CLUMPY
QRC	2012	Natural cover	31,001	91.4	11,971	35.308	4.77	839,465	24.76	0.961
		Vegetation deficits	2903	8.6	11,001	32.447	0.16			0.956
	2018	Natural cover	31,597	93.2	13,496	39.806	3.86	849,375	25.05	0.951
		Vegetation deficits	2307	6.8	13,471	39.733	0.12			0.945
SJX	2012	Natural cover	20,899	72.6	27,333	94.961	1.28	2,020,605	70.20	0.953
		Vegetation deficits	7884	27.4	32,142	111.668	0.21			0.950
	2018	Natural cover	22,248	77.3	24,440	84.910	1.56	1,604,585	55.75	0.958
		Vegetation deficits	6535	22.7	25,306	87.919	0.16			0.955

CA – Total class area (ha); PLAND – Percentage of landscape of class (%); NP – Number of patches; PD – Patch density; LPI – Largest patch index (%); TE – Total edge (m); ED – Edge density (m/ha); CLUMPY – Clumpiness index (aggregation index).

3.4. Implications for implementing the NVPL and public policies

Understanding the patterns of riparian deforestation and legal compliance in crop expansion and intensification processes is key to assessing whether technology and productivity increases and, at the same time, keeps pace with advances in environmental protection in agricultural landscape frontier in Amazonia. Land use and land tenure have different impacts on law compliance, as well as the regional law enforcement histories. While low relative vegetation deficit in the PPAs is found associated with crop farms in intensive land use and stricter law enforcement landscapes, for example, an opposite pattern seems to dominate landscapes in an early stage of intensification and greater lack of law enforcement policies.

Although NVPL seemed to favor an upward trend in the legal compliance status of riparian PPAs, new clearing areas are still found, stressing the need for advances in monitoring and law enforcement capacity. Despite of the fact that the Brazilian public electronic Rural Environmental Registry (CAR) made it possible to integrate land tenure and land use in rural properties, its self-declaratory record leads to a lack of a systematic methodology in data collection, validation and comparability. Moreover, investments in resources for this registry consolidation and maintenance of proper functioning of the bodies responsible for the monitoring, inspection and accountability of environmental crimes is essential for the realization of several projects of extreme relevance for the country. The role of a whole-landscape approach to support designing large-scale restoration strategies is also a key issue to conceal production and conservation, restore ecosystem services and contribute to local and regional development.

4. Conclusions

After six years, NVPL were found to have an overall impact in reducing vegetation deficits in riparian PPAs, a trend which was heterogeneously distributed across the landscape and related to the dynamics of land use and protection regimes. Our findings were particularly important to shed light on questions regarding the loosening of the NVPL requirements for the protection of riparian areas, representing the first work to analyze on a temporal scale the intensity and distribution of these impacts, considering different drivers. Although riparian deforestation was reduced since 2012, it was much lower in the consolidated areas than in the PPAs that remained in the regular regime. These areas also presented huge environmental debts, and new clearings were not ceased, even though occurred mainly in the regular regime PPAs. Future efforts must be made through the spatial and temporal expansion of these analyzes to confirm the extent of these trends at the national level.

Properties with the same type of land use still present different patterns of riparian deforestation and restoration rates, which shows that market pressures are not the only driving factors and the degree of technological investment and management techniques should be considered. Stricter command and control measures have shown high potential for controlling riparian deforestation. The intensification of this approach can be important especially in large-sized farms, which accounted for the largest extents of riparian deforestation and remain the main drivers of new clearings. However, the high relative vegetation deficits found in riparian settlements areas and other small agricultural productions underscore the importance of larger support for these social groups, which could benefit greatly from economic incentive measures and technical and logistical support.

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Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at doi:10.17632/g9wns3yyr7.2.

References

Alencar, A., Pereira, C., Castro, I., Cardoso, A., Souza, L., Costa, R., Bentes, A.J., Stella, O., Azevedo, A., Gomes, J., Novaes, R., 2016. Desmatamento nos assentamentos da Amazônia: histórico, tendências e oportunidades. IPAM, Brasília, DF, p. 93.

Amazon Environmental Research Institute; Socio-environmental Institute (IPAM-ISA), 2017. Plano Municipal de Restauro de Áreas Degradadas de Querência. GT Restauração do Fórum Multi-Setorial de Querência. Conselho Municipal de Meio Ambiente. Projeto Querência+: Paisagens Sustentáveis. 23p.

Arvor, D., Daugeard, M., Tritsch, I., Mello-Thery, N.A., Thery, H., Dubreuil, V., 2018. Combining socioeconomic development with environmental governance in the Brazilian Amazon: the Mato Grosso agricultural frontier at a tipping point. Environ. Dev. Sustain 20, 1–22. https://doi.org/10.1007/s10668-016-9889-1.

Assunção, J., Gandour, C., Pessoa, P., Rocha, R., 2017. Property-level assessment of change in forest clearing patterns: the need for tailoring policy in the Amazon. Land Use Policy 66, 18–27. https://doi.org/10.1016/j.landusepol.2017.04.022.

Assunção, J., Rocha, R., 2019. Getting greener by going black: the effect of blacklisting municipalities on Amazon deforestation. Environ. Dev. Econ. 24 (2), 115–137. https://doi.org/10.1017/S1355770X18000499.

Ballester, M.V.R., Garcia, A.S., Rizzo, R., Vilela, V.M.F.N., Nakai, E., Preto, M.F., Casarin, L.P., Tourne, D.C.M., 2020. Soil functions and land management: potential synergies and tradeoffs at the tropical agricultural frontier (Brazil). In: Alves, V.S. (Ed.), Land Use Changes: Management and Applications, 1 ed. Nova Science Publishers, Hauppauge, NY, pp. 59–82.

Bleich, M.E., Mortati, A.F., André, T., Piedade, M.T.F., 2014. Riparian deforestation affects the structural dynamics of headwater streams in Southern Brazilian Amazonia. Trop. Conserv. Sci. 7 (4), 657–676. https://doi.org/10.1177/194008291400700406.

Brancalion, P.H., Garcia, L.C., Loyola, R., Rodrigues, R.R., Pillar, V.D., Lewinsohn, T.M., 2016. A critical analysis of the Native Vegetation Protection Law of Brazil (2012): updates and ongoing initiatives. Nat. Conserv. 14S, e1–e16. https://doi.org/10.1016/j.ncon.2016.03.003.

Brazil, 2007. Decree no. 6321. Dispõe sobre ações relativas à prevenção, monitoramento e controle de desmatamento no Bioma Amazônia, bem como altera e acresce dispositivos ao Decreto no. 3.179, de 21 de setembro de 1999, que dispõe sobre a especificação das sanções aplicáveis às condutas e atividades lesivas ao meio ambiente, e dá outras providências.

Brazil, 2012a, Federal Law no. 12,651. Dispõe sobre a proteção da vegetação nativa; altera as Leis nº 6.938, de 31 de agosto de 1981, 9.393, de 19 de dezembro de 1996, e 11.428, de 22 de dezembro de 2006; revoga as Leis nº 4.771, de 15 de setembro de 1965, e 7.754, de 14 de abril de 1989, e a Medida Provisória nº 2.166–67, de 24 de agosto de 2001; e dá outras providências.

Brazil, 2012b. Federal Law no. 12,727. Altera a Lei no. 12.651, de 25 de maio de 2012, que dispõe sobre a proteção da vegetação nativa; altera as Leis no. 6.938, de 31 de agosto de 1981, 9.393, de 19 de dezembro de 1996, e 11.428, de 22 de dezembro de 2006; e revoga as Leis no. 4.771, de 15 de setembro de 1965, e 7.754, de 14 de abril de 1989, a MP no. 2.166-67, de 24 de agosto de 2001, o item 22 do inciso II do art. 167 da Lei no. 6.015, de 31 de dezembro de 1973, e o § 2º do art. 4º da Lei no. 12.651, de 25 de maio de 2012.

Brazilian Association of Vegetable Oil Industries (ABIOVE), 2014. New Agenda for Soybeans in the Amazon Biome. ABIOVE, São Paulo.

Brazilian Association of Vegetable Oil Industries (ABIOVE), 2019. Moratória da Soja: monitoramento por imagens de satélites dos plantios de soja no bioma Amazônia

(Safra 2017/2018). 36p. ⟨https://abiove.org.br/wp-content/uploads/2019/05/30012019-165924-portugues.pdf⟩.
Brazilian Federal Prosecution Office (PGR), 2013. Direct Actions of Unconstitutionality. ⟨http://www.mpf.mp.br/atuacao-tematica/ccr4/dados-da-atuacao/documentos/adis-propostas⟩.
Brazilian Foundation for Sustainable Development (FBDS), 2018. Mapeamento em Alta Resolução dos Biomas Brasileiros. ⟨http://geo.fbds.org.br/⟩.
Brazilian Institute of Environment and Renewable Natural Resources (IBAMA), 2019. Relatório de gestão do exercício de 2018. MMA, Brasil, p. 138. ⟨https://www.gov.br/ibama/pt-br/centrais-de-conteudo/2019-01-07-relatorio-de-gesta-ibama-2018-pdf⟩.
Brazilian Institute of Geography and Statistics (IBGE), 2009a. Produção da Pecuária Municipal 2008. Rio de Janeiro. ⟨https://cidades.ibge.gov.br⟩.
Brazilian Institute of Geography and Statistics (IBGE), 2009b. Produção Agrícola Municipal 2008. Rio de Janeiro. ⟨https://cidades.ibge.gov.br/Brazil/mt/Querência/pesquisa/14/10193?ano=2008⟩.
Brazilian Institute of Geography and Statistics (IBGE). 2017a. Censo Agropecuário. ⟨https://censos.ibge.gov.br/agro/2017/⟩.
Brazilian Institute of Geography and Statistics (IBGE), 2017b. Produto Interno Bruto dos Municípios. ⟨https://cidades.ibge.gov.br/brasil/mt/querencia/panorama?tipo=ranking&indicador=47001⟩.
Brazilian Institute of Geography and Statistics (IBGE), 2018. IBGE cidades. ⟨https://cidades.ibge.gov.br/⟩.
Brazilian Institute of Geography and Statistics (IBGE), 2019a. População estimada. Diretoria de Pesquisas, Coordenação de População e Indicadores Sociais, Estimativas da população residente com data de referência 1º de julho de 2019. ⟨https://cidades.ibge.gov.br⟩.
Brazilian Institute of Geography and Statistics (IBGE), 2019b. Base de Dados Espacial 1: 250.000, Brazil. ⟨ftp://geoftp.ibge.gov.br/informacoes_ambientais/vegetacao/vetores/escala_250_mil/⟩.
Brazilian Institute of Geography and Statistics (IBGE), 2019c. Produção da Pecuária Municipal 2018. Rio de Janeiro. ⟨https://cidades.ibge.gov.br⟩.
Brazilian Institute of Geography and Statistics (IBGE), 2019d. Produção Agrícola Municipal 2018. Rio de Janeiro. ⟨https://cidades.ibge.gov.br⟩.
Brazilian National Water Agency (ANA), 2006. Nível 2 de sub-bacias do Plano Nacional de Recursos Hídricos. ⟨https://metadados.ana.gov.br/geonetwork⟩.
Burdon, F.J., Mcintosh, A., Harding, J.S., 2013. Habitat loss drives threshold response of benthic invertebrate communities to deposited sediment in agricultural streams. Ecol. Appl. 23 (5), 1036–1047. https://doi.org/10.1890/12-1190.1.
Cardoso, L.V., 2011. Financiamento agroambiental no Brasil: subsídio para o desenvolvimento de políticas de crédito de apoio à regularização ambiental de propriedades rurais. Instituto Socioambiental, São Paulo, p. 82.
Central Bank of Brazil (CBB), 2008. Resolution no. 3545. Altera o MCR 2-1 para estabelecer exigência de documentação comprobatória de regularidade ambiental e outras condicionantes, para fins de financiamento agropecuário no Bioma Amazônia.
Chiavari, J., Lopes, C.L., 2017. Legislação Florestal e de uso da terra: uma comparação internacional. Iniciativa para o uso da terra (INPUT). Climate Policy Initiative (CPI) e Núcleo de Avaliação de Políticas Climáticas da PUC-Rio (NAPC/PUC-Rio). 19p.
Deegan, L.A., Neill, C., Haupert, C.L., Ballester, M.V.R., Krusche, A.V., Victoria, R.L., Thomas, S.M., De Moor, E., 2011. Amazon deforestation alters small stream structure, nitrogen biogeochemistry and connectivity to larger rivers. Biogeochemistry 105 (1–3), 53–74. https://doi.org/10.1007/s10533-010-9540-4.
Elliott, K.J., Vose, J.M., 2016. Effects of riparian zone buffer widths on vegetation diversity in southern Appalachian headwater catchments. For. Ecol. Manag. 376, 9–23. https://doi.org/10.1016/j.foreco.2016.05.046.
Farias, M.H.C.S., Beltrão, N.E.S., Santos, C.A., Cordeiro, Y.E.M., 2018. Impact of rural settlements on the deforestation of the Amazon. Mercator 17, e17009. https://doi.org/10.4215/rm2018.e17009.
Fatorelli, L., Mertens, F., 2010. Integração de políticas e governança ambiental: o caso do licenciamento rural no Brasil. Ambient. Soc. 13 (2), 401–415. https://doi.org/10.1590/S1414-753X2010000200012.
Fearnside, P.M., 2008. Amazon forest maintenance as a source of environmental services. An. Acad. Bras. Ciênc. 80 (1), 101–114. https://doi.org/10.1590/S0001-37652008000100006.
Food and Agriculture Organization of the United Nations (FAO), 2006. Global Forest Resources Assessment 2005: Progress Towards Sustainable Forest Management. Forestry Paper 147, 348p.
Food and Agriculture Organization of the United Nations (FAO), 2019. FAOSTAT: Food and agriculture data. ⟨www.fao.org/statistics/en/⟩.
Freitas, F.L.M., Englund, O., Sparovek, G., Berndes, G., Guidotti, V., Pinto, L.F.G., Mörtberg, U., 2017. Who owns the Brazilian carbon? Glob. Change Biol. 24 (5), 2129–2142. https://doi.org/10.1111/gcb.14011.
Garcia, A.S., Vilela, V.M.F.N., Rizzo, R., West, P., Gerber, J.S., Engstrom, P.M., Ballester, M.V.R., 2019. Assessing land use/cover dynamics and exploring drivers in the Amazon's arc of deforestation through a hierarchical, multi-scale and multi-temporal classification approach. Remote Sens. Appl.: Soc. Environ. 15 (100233), 1–14. https://doi.org/10.1016/j.rsase.2019.05.002.
Gasparri, N.I., de Waroux, Y.P., 2015. The coupling of South American soybean and cattle production frontiers: new challenges for conservation policy and land change science. Conserv. Lett. 8, 290–298. https://doi.org/10.1111/conl.12121.
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 (6220), 377–378. https://doi.org/10.1126/science.aaa0181.
Giglio, L., Csiszar, I., Justice, C.O., 2006. Global distribution and seasonality of active fires as observed with the Terra and Aqua Moderate Resolution Imaging Spectroradiometer (MODIS) sensors. J. Geophys. Res. 3, G02016 https://doi.org/10.1029/2005JG000142.
Godar, J., Tizado, E.J., Pokorny, B., 2012. Who is responsible for deforestation in the Amazon? A spatially explicit analysis along the Transamazon Highway in Brazil. For. Ecol. Manag. 267, 58–73. https://doi.org/10.1016/j.foreco.2011.11.046.
Godar, J., Gardner, T.A., Tizado, E.J., Pacheco, P., 2014. Actor-specific contributions to the deforestation slowdown in the Brazilian Amazon. Proc. Natl. Acad. Sci. USA 111 (43), 15591–15596. https://doi.org/10.1073/pnas.1322825111.
Gollnow, F., Hissa, L.D.B.V., Rufin, P., Lakes, T., 2018. Property-level direct and indirect deforestation for soybean production in the Amazon region of Mato Grosso, Brazil. Land Use Policy 78, 377–385. https://doi.org/10.1016/j.landusepol.2018.07.010.
Guidotti, V., Ferraz, S.F.B., Pinto, L.F.G., Sparovek, G., Taniwaki, R.H., Garcia, L.G., Brancalion, P.H., 2020. Changes in Brazil's Forest Code can erode the potential of riparian buffers to supply watershed services. Land Use Policy 94, 104511. https://doi.org/10.1016/j.landusepol.2020.104511.
Hesselbarth, M.H.K., Sciaini, M., With, K.A., Wiegand, K., Nowosad, J., 2019. landscapemetrics: an open-source R tool to calculate landscape metrics. Ecography 42 (10), 1648–1657. https://doi.org/10.1111/ecog.04617.
Kauffman, J.B., Krueger, W.C., 1984. Livestock impacts on riparian ecosystems and streamside management implications… A review. J. Range Manag. 37 (5), 430–438.
Landis, J.R., Koch, G.G., 1977. A one-way components of variance model for categorical data. Biometrics 33, 671–679.
Leal, C.G., Pompeu, P.S., Gardner, T.A., Leitão, R.P., Hughes, R.M., Kaufmann, P.R., Zuanon, J., Paula, F.R., Ferraz, S.F.B., Thomson, J.R., Nally, R.M., Ferreira, J., Barlow, J., 2016. Multi-scale assessment of human-induced changes to Amazonian instream habitats. Landsc. Ecol. 31, 1725–1745. https://doi.org/10.1007/s10980-016-0358-x.
Lees, A.C., Peres, C.A., 2008. Conservation value of remnant riparian forest corridors of varying quality for Amazonian birds and mammals. Conserv. Biol. 22 (2), 439–449. https://doi.org/10.1111/j.1523-1739.2007.00870.x.
Lowe, W.H., Likens, G.E., 2005. Moving headwater streams to the head of the class. BioScience 55 (3), 196–197. https://doi.org/10.1641/0006-3568(2005)055[0196:MHSTTH]2.0.CO;2.
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.
Luke, S.H., Slade, E.M., Gray, C.L., Annammala, K.V., Drewer, J., Williamson, J., Agama, A.L., Ationg, M., Mitchell, S.L., Vairappan, C.S., Struebig, M.J., 2019. Riparian buffers in tropical agriculture: scientific support, effectiveness and directions for policy. J. Appl. Ecol. 56 (1), 85–92. https://doi.org/10.1111/1365-2664.13280.
Metzger, J.P., 2010. O Código Florestal tem Base Científica? Nat. Conserv. 8 (1), 92–99.
Ministry of Industry, Foreign Trade and Services (MDIC), 2020. Comex Stat: Official system for consulting Brazilian international trade data. ⟨http://comexstat.mdic.gov.br/⟩.
Ministry of the Environment (MMA), 2006. Sistema de licenciamento ambiental em propriedades rurais no estado de Mato Grosso: análise de sua implementação. Instituto Socioambiental (ISA), Instituto Centro de Vida (ICV). MMA, Brasília, p. 177.
Ministry of the Environment (MMA), 2007. Ordinance no. 28. Dispõe sobre os Municípios situados no Bioma Amazônia onde incidirão ações prioritárias de prevenção, monitoramento e controle o desmatamento ilegal.
Ministry of the Environment (MMA), Brazilian Army (MMA/DSG-EB), 2008. Sistema de Informações Cartográficas da SEPLAN-MT. ⟨http://www.seplan.mt.gov.br/-/10951338-bases-cartograficas⟩.
Ministry of the Environment (MMA), 2011. Ordinance no 139. www.jusBrazil.com.br/diarios/26226575/pg-99-secao-1-diario-oficial-da-uniao-dou-de-25-04-2011.
Ministry of the Environment (MMA), 2013. Plano de Ação para prevenção e controle do desmatamento na Amazônia Legal (PPCDAm). MMA, Brasília, p. 174.
Mittermeier, R.A., 2004. Hotspots revisited. Cemex.
Nagy, R.C., Porder, S., Neill, C., Brando, P., Quintino, R.M., Nascimento, S.A., 2015. Structure and composition of altered riparian forests in an agricultural Amazonian landscape. Ecol. Appl. 25 (6), 1725–1738. https://doi.org/10.1890/14-1740.1.
Naiman, R.J., Décamps, H., 1997. The ecology of interfaces: riparian zones. Annu. Rev. Ecol. Syst. 28, 621–658.
National Institute for Space Research (INPE), 2019. Projeto PRODES: Desmatamento nos Municípios da Amazônia Legal. ⟨http://www.dpi.inpe.br/prodesdigital/prodesmunicipal.php⟩.
National Institute of Colonization and Agrarian Reform (INCRA), 2013. Sistema Nacional de Cadastro Rural: Índices básicos de 2013. ⟨http://www.incra.gov.br/sites/default/files/uploads/estrutura-fundiaria/regularizacao-fundiaria/indices-cadastrais/indices_basicos_2013_por_municipio.pdf⟩.
Nepstad, D., McGrath, D., Stickler, C., Alencar, A., Azevedo, A., Swette, B., Bezerra, T., DiGiano, M., Shimada, J.A., da Motta, R.S., Armijo, E., Castello, L., Brando, P., Hansen, M.C., McGrath-Horn, M., Carvalho, O., Hess, L., 2014. Slowing Amazon deforestation through public policy and interventions in beef and soy supply chains. Science 344, 1118–1123.
Nunes, S.S., Barlow, J., Gardner, T.A., Siqueira, J.V., Sales, M.R., Souza, C.M., 2015. A 22 year assessment of deforestation and restoration in riparian forests in the eastern Brazilian Amazon. Environ. Conserv. 42 (3), 193–203. https://doi.org/10.1017/S0376892914000356.
Nunes, S.S., Barlow, J., Gardner, T., Sales, M., Monteiro, D., Souza Jr, C., 2019. Uncertainties in assessing the extent and legal compliance status of riparian forests in the eastern Brazilian Amazon. Land Use Policy 82, 37–47. https://doi.org/10.1016/j.landusepol.2018.11.051.

Planet, 2019. Imagery product specifications. Planet Labs Inc. ⟨https://assets.planet.com/docs/combined-imagery-product-spec-final-may-2019.pdf⟩.

R Core Team, 2020. R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienna, Austria. ⟨www.R-project.org/⟩.

Rajão, R., Soares-Filho, B., Nunes, F., Börner, J., Machado, L., Assis, D., Oliveira, A., Pinto, L., Ribeiro, V., Rausch, L., Gibbs, H., Figueira, D. The rotten apples of Brazil's agribusiness. Science, vol. 369(6501), pp. 246–8.

Ramos, C.C.O., Anjos, L., 2014. The width and biotic integrity of riparian forests affect richness, abundance, and composition of bird communities. Nat. Conserv. 12 (1), 59–64. https://doi.org/10.4322/natcon.2014.011.

Rausch, L.L., Gibbs, H.K., 2016. Property arrangements and soy governance in the Brazilian state of Mato Grosso: implications for deforestation-free production. Land 5 (2), 7. https://doi.org/10.3390/land5020007.

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

Roriz, P.A.C., Fearnside, P.M., 2015. A construção do Código Florestal Brasileiro e as diferentes perspectivas para a proteção das florestas. Novos Cad. – NAEA 18 (2), 51–68. https://doi.org/10.5801/ncn.v18i2.1866.

Rudorff, B.F.T., Adami, M., Aguiar, D.A., Moreira, M.A., Mello, M.P., Fabiani, L., Amaral, D.F., Pires, B.M., 2011. The soy moratorium in the Amazon biome monitored by remote sensing images. Remote Sens. 3 (1), 185–202. https://doi.org/10.3390/rs3010185.

Silva, J.A.A., Nobre, A.D., Manzatto, C.V., Joly, C.A., Rodrigues, R.R., Skorupa, L.A., Nobre, C.A., Ahrens, S., May, P.H., Sá, T.D.A., Cunha, M.C., Rech Filho, E.L., 2012. O Código Florestal e a ciência: contribuições para o diálogo, 2a ed. Sociedade Brasileira para o Progresso da Ciência (SBPC), Academia Brasileira de Ciências (ABC), p. 294.

Simões, C.G., Alves, H.Q., Stabile, M.C.C., Junqueira, R.G.P., 2017. Diagnóstico socioambiental de Querência. Querência+: Paisagens Sustentáveis. Instituto de Pesquisas da Amazônia, Instituto Socioambiental, p. 70. ⟨https://ipam.org.br/wp-content/uploads/2017/07/Diagnostico_final.pdf⟩.

Soares, J.L.N., 2008. A organização territorial de assentamentos rurais para atender a legislação ambiental na Amazônia. Campo-Território 3 (6), 143–155.

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 (6182), 363–364. https://doi.org/10.1126/science.1246663.

Souza, R.A., Miziara, F., Marco Junior, P., 2013. Spatial variation of deforestation rates in the Brazilian Amazon: a complex theater for agrarian technology, agrarian structure and governance by surveillance. Land Use Policy 30, 915–924. https://doi.org/10.1016/j.landusepol.2012.07.001.

Steward, C., 2007. From colonization to "environmental soy": a case study of environmental and socio-economic valuation in the Amazon soy frontier. Agric. Hum. Values 24, 107–122. https://doi.org/10.1007/s10460-006-9030-4.

Sweeney, B.W., Bott, T.L., Jackson, J.K., Kaplan, L.A., Newbold, J.D., Standley, L.J., Hession, W.C., Horwitz, R.J., 2004. Riparian deforestation, stream narrowing, and loss of stream ecosystem services. PNAS 101 (39), 14132–14137. https://doi.org/10.1073/pnas.0405895101.

Taniwaki, R.H., Forte, Y.A., Silva, G.O., Brancalion, P.H., Cogueto, C.V., Filoso, S., Ferraz, S.F., 2018. The native vegetation protection law of Brazil and the challenge for first-order stream conservation. Perspect. Ecol. Conserv. 16 (1), 49–53. https://doi.org/10.1016/j.pecon.2017.08.007.

Velasquez, C., Alves, H.Q., Bernasconi, P. (Eds.), 2010. Fique por dentro: a Bacia do Rio Xingu em Mato Grosso. Instituto Socioambiental, Instituto Centro de Vida, São Paulo, p. 40.

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.