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Land use footprints and policies in Brazil

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ABSTRACT

Ecological footprints have been assessed widely from a resource production-consumption perspective but much less from a land use *per capita* availability-demand standpoint. The later view is key to land use policy because it sheds light on the need of changing or adapting uses to get a balance between those of ecological (e.g., forests, riparian wetlands) and those of socioeconomic (e.g., cropland) value. Thus, the purpose of this study was to introduce the *LUEF* – Land Use Ecological Footprint, defined as the area of a specific use or occupation in a region over the population of that region in a pre-defined timeframe. The index was then applied to the Brazilian territory at the macro region resolution, and to the 2015 – 2019 period. The results identified deforestation, urban densification, cropland/pasture expansion and riparian forest restoration as main drivers of *LUEF* change in the studied period, as well as supporters of concomitant gross domestic product. The results also revealed negative consequences of *LUEF* changes for water security and organic carbon stocks in the top layer of soils (decline). Some regional metrics were proposed to revert these consequences, namely the control of urban *LUEF* above 100 m^2/hab to keep water security at the National average level, and of cropland *LUEF* below 900 m^2/hab to preserve organic carbon stock in the macro regions' topsoils close to the National average (46.9 Mg/ha), reducing greenhouse gas emissions in the sequel. The leveling of those *LUEF*s at the aforementioned values requires intensification of ongoing policy initiatives relevant to mitigate the land use ecological footprints. The study identified various examples, which included the Brazilian Forest Code, the National policies on urban and family agriculture, Payment for Ecosystem Services programs, among others. Overall, the study recognized Brazil as being in the right track to pursue sustainable land use.

1. Introduction

Understanding the impacts of land use change, such as deforestation with subsequent agricultural, industrial or urban land use, is of increasing pertinence, given their importance as drivers of natural ecosystem changes (Quezada et al., 2022). In the current global scenario, appropriate land use requires a balance between the maintenance of ecosystem services and socioeconomic development. Without these trade-offs, the conversion of natural cover into anthropic land will likely cause environmental damage, with widespread disruption of ecosystem functions (Valera et al., 2017, 2016). In Brazil, the competition between conservation of native vegetation and agro-industrial activities and urban sprawl has caused ample deforestation (de Oliveira et al., 2023; Parras et al., 2020). Frequently, the competition has evolved into serious environmental conflicts, because the expansion of productive areas over natural vegetation areas has reduced the system's capacity to conserve

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vital services, particularly those related to water resources in watersheds (Pissarra et al., 2019; Valera et al., 2019). On the other hand, population growth, urban sprawl and rapid social and economic developments have stressed even more the environment endangering its capacity to provide ecosystem services (Li et al., 2021). It is therefore urgent to reinforce public policies that ensure provision of services from healthy ecosystems, as they are foundations of sustainable development, either of economy, society or culture, as well as delivers of invaluable benefits for human survival (Dong et al., 2021).

The basis of policy is science, and science stands out on methods that are capable of delivering metrics to feed the policies. As regards ecosystem services provision, the ecological footprint approach (Bazan, 1997; Wackernagel, 1998) is considered complete, comprehensive and effective to assess the regional natural capital use (Ahmad et al., 2020). The method cleverly links the human social system to the natural ecosystem through a unified reference: the area of biologically productive land (Li et al., 2022). Biological productive areas comprise agricultural soils used for food or fiber production, fishing grounds, and forest soils used for timber production, but exclude deserts, glaciers and the open ocean. They are quantified through global hectares (gha), which represent the biologic productive area available to sustain the needs of humans and activities within a political administrative region, i.e., the region's biocapacity or carrying capacity. The ecological footprint, on the other hand, represents the amount of that area required to sustain those humans and activities in the same region, besides absorbing and rendering harmless the corresponding waste (Green et al., 2019).

The concepts of biocapacity and ecological footprint are meant to analyze past or present consumption patterns (Dai et al., 2023; Marquardt et al., 2021), as well as to forecast future consumption trends. The studies that used the ecological footprint method are numerous and have covered all sorts of resources, in separate, combined in nexus approaches (e.g., water-land-energy-food), or addressing the natural capital as whole (Dembińska et al., 2022; Guo et al., 2022; Yao et al., 2022). For specific resources, one can refer a recent investigation in the Northern Xinjiang region in China covering the 2000 – 2020 period, which reported, among other findings, an increase in the energy footprints of industrial cities that also raised the pressure on water resources supply (Yue et al., 2023). Two other studies evaluated the agriculture and livestock water footprints in China and Thailand (Jaibumrung et al., 2023; Kou et al., 2023). Researchers have also dedicated efforts to identify and quantify the role of ecological footprint driving forces (Kazemzadeh et al., 2023; Xin et al., 2023). In this context, some works carried out in China and various OECD (Organization for Economic Co-operation and Development) and sub-Saharan Africa countries, quantified the effects of urbanization and industrialization (negative), as well as of renewable energy use and technological innovation (positive), on the ecological footprint (Appiah et al., 2023; Kassouri, 2021; Kirikkaleli et al., 2023; Li et al., 2023; Zhang et al., 2022). The relationship of ecological footprint with foreign trade in the top ten fastest developing countries in the global economy have also been demonstrated recently (Cutcu et al., 2023). There are also child or related approaches to this supply (biocapacity) *versus* demand (ecological footprint) dichotomy, such as the touristic footprint, which is the ecological footprint of visitors to a given touristic destination (Lin et al., 2018; Mancini et al., 2022; Phumalee et al., 2018); the carbon footprint that links the ecological footprint to greenhouse gases warming potential (de Arce and Mahía, 2023; P. Deng et al., 2023; L. Liu et al., 2023; Z. Liu et al., 2023; Ma et al., 2023; Marchi et al., 2023; Mitoma, 2023; Righi et al., 2023); or the emergy ecological footprint that relates the ecological footprint to an amount of energy consumed in the transformations that generate a product or a service (Du et al., 2022; Fartout Enayat et al., 2023; Liu et al., 2022; Pan and Guo, 2023; Xie et al., 2022; Zadehdabagh et al., 2022). Taken together, the ecological footprint approaches (*sensu lato*) set a natural capital conservation paradigm through sustainable consumption or use.

Despite the abundant literature on the ecological footprint, there are gaps remaining to close. One of those gaps relates with the assessment of a land use ecological footprint watched from a landscape planning perspective. That kind of assessment would seek to understand how the population is distributed as function of land uses or covers in a specific region, considering the ecologic function of some covers (e.g., forests, wetlands) in comparison with the socio-economic function of others (e.g., cropland, pastures, build up environment), and how the demographic trends shape that distribution changing the balance between those functions. Ultimately, the analysis would provide subsides to land use policies and landscape plans about regions that need land use conversions to reverse low ecologic/socio-economic ratios, as well as regions that can improve socio-economic development given their high ratios. With the motivation to overcome this challenge, the general purpose of this study was to introduce the *LUEF* – Land Use Ecological Footprint index and use it to assess a land use ecological footprint from the aforementioned landscape planning standpoint. The specific purposes were: (1) Define the *LUEF* and provide a rational for using this index; (2) Monitor the *LUEF* over a certain period (2015 – 2019) to detect spatio-temporal trends and try to understand them; (3) Relate the components of *LUEF*, such as the *LUEF* of agriculture or the *LUEF* of build environment, with potential drivers or consequences; (4) Discuss the results from a policy perspective. The area used for testing was the entire territory of Brazil given the importance of this continental country in the global economy (e.g., commodity production and trade) and ecological conservation (e.g., the Amazon) agendas. The novelty and key contribution of this research was the assessment and presentation of ecological footprints from a brand-new perspective, which shifted the footprint analysis from the conventional production – consumption view to the land use (ecological *versus* socio-economic function) view. To our best knowledge, this has barely been done before, at least while using a metric to trace back and forth the land use ecological footprint. In that regard, the *LUEF* indicator (the metric) introduced in this paper is also a key contribution to the study. It is worth recalling that the *LUEF* is well scoped (land use area divided by the population living in a pre-defined region, usually an administrative region) and based on readily available data (just a land use and cover map, a map of administrative divisions to set up the spatial resolution, and data on population compiled from a demographic census at the same spatial resolution), which makes it transposable to any scenario worldwide with no difficulty.

2. Materials and methods

2.1. Study area

The study area is the entire territory of Brazil (Fig. 1), which covers 8514,876 km^2 distributed among 26 States and the Federal District (Brasília). The states are subdivided into municipalities, which in turn are merged into micro-regions, meso-regions and finally five large macro-regions: North, Northeast, Center-West (or Midwest), South and Southeast. Besides the administrative division, the Brazilian territory is usually divided into five morphoclimatic domains, known as Biomes and defined on the basis of environmental, climatic, fauna and flora characteristics: Amazon, Cerrado, Caatinga, Pantanal, Atlantic Forest and Pampa.

2.2. Datasets

2.2.1. Land use and cover data retrieval

The current study is based on the Copernicus product CGLS-LC100 Collection 3 (https://www.copernicus.eu/en), which is currently prepared with data from the PROBA-V sensor and has been widely used in many recent land use and cover change studies (Kasmaeeyazdi et al., 2021; Masiliūnas et al., 2021; Rodríguez-Benito et al., 2020; Samuele et al., 2022; Taramelli et al., 2019; Xiao et al., 2022; Xu et al., 2022). In addition, the product is being used in projects sponsored by

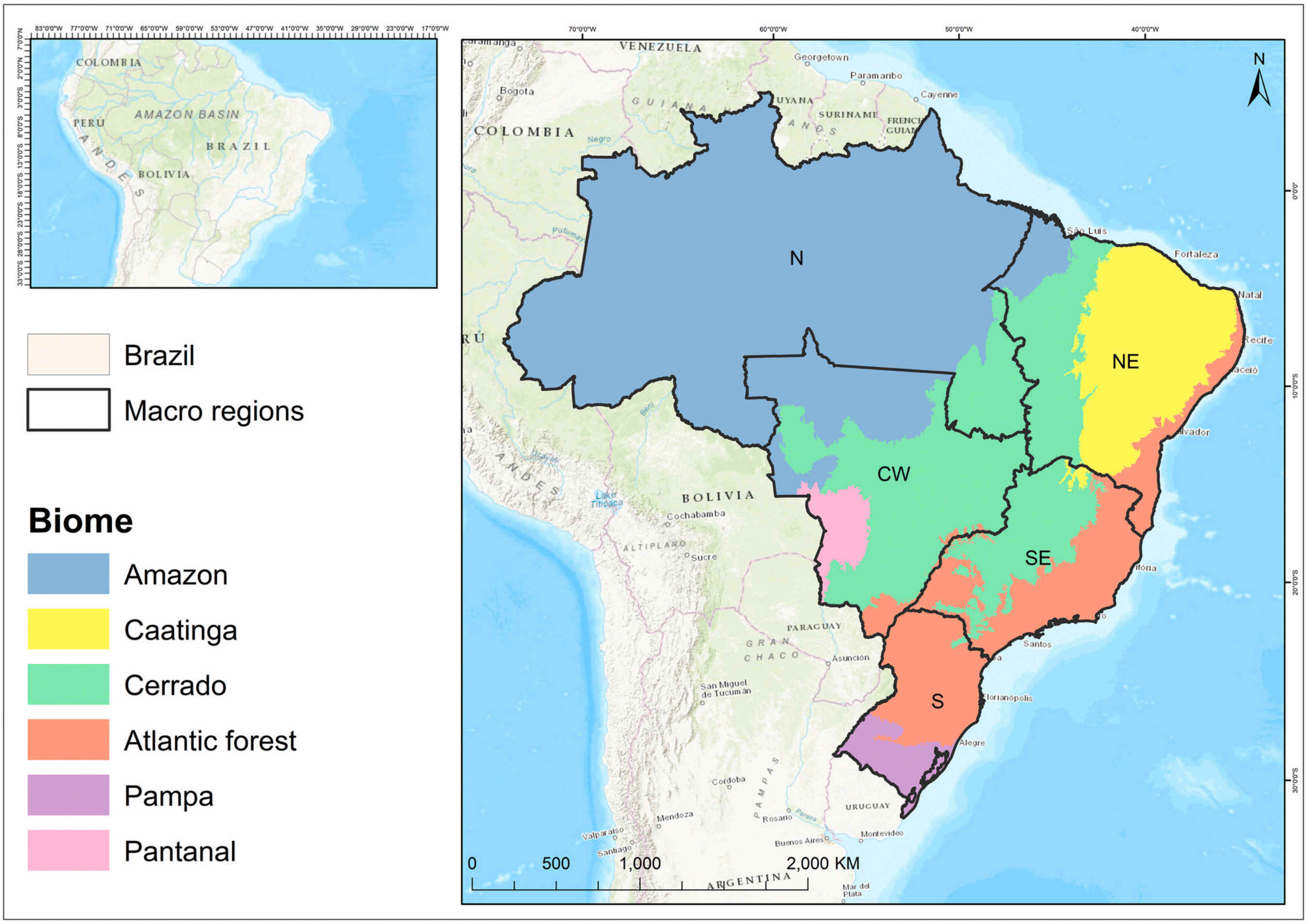

Fig. 1. Macro regions of Brazil (N – North; NE – Northeast; CW – Center-West (or Midwest); S – South; SE - Southeast) and spatial distribution of Biomes (Pampa, Pantanal, Cerrado, Caatinga, Amazon and Atlantic Forest; represented with colors).

non-governmental organizations, such as the World Conservation Monitoring Centre that has created a global forest management layer and has been funded by the United Nations Environmental Programme; or the WaPOR – Water Productivity Open Access Portal that has generated annual agriculture maps for Africa starting in 2010, extending the class of crops into irrigated and rainfed agriculture, and has been funded by the Food and Agriculture Organization. The Copernicus resource comprises 23 land use and cover classes aligned with UN-FAO's Land Cover Classification System, being assessed at 100 m spatial resolution. The 23 classes are described at three levels of detail and, in addition, the product includes fraction maps that provide proportions of coverage for 10 main classes; a forest type layer with subtypes (e.g., deciduous needle leaf, evergreen broad leaf); among other features. The overall accuracy at classification level 1 (the more general descriptions) reaches 80.2%, while for level 2 it drops to 75.4%. Moreover, the product has captured reasonably well the land use and cover changes occurring in recent years globally, with an overall accuracy of 99.6%.

Currently, the Copernicus platform stores land use and cover data from the 2015 – 2019 period (https://land.copernicus.eu/global/products/lc; assess in 20 November 2023). We downloaded that time series to use in the present study (see Figure S1 in the Supplementary Materials) and considered seven land use and cover classes in the footprint assessment. The Brazilian territory is covered with 6 raster images, with a dimension of 20° × 20° each. The land use and cover classes are described according to the Buchhorn and co-workers' study (https://zenodo.org/record/3939050), as follows: "*forest*: in the current study the level 1 classification was adopted, which assembles together deciduous, evergreen and mixed forests. The reason for using the more general classes was because although Brazil presents a high continental variability of forest typologies it was assumed that these formations play a similar role in the provision of ecosystem services; *herbaceous vegetation (pastures)*: plants without persistent stem or shoots above ground and lacking definite firm structure. Tree and shrub cover is less than 10%; *cropland*: lands covered with temporary crops followed by harvest and a bare soil period (e.g., single and multiple cropping systems); *Shrubs*: woody perennial plants with persistent and woody stems and without any defined main stem being less than 5 m tall. The shrub foliage can be either evergreen or deciduous; *wetland vegetation*: lands with a permanent mixture of water and herbaceous or woody vegetation. The vegetation can be present in either salt, brackish, or fresh water; *urban and build up areas*: land covered by buildings and other man-made structures; *other uses*: this layer accommodated all the other uses, because their cartographic expression is very small at the selected scale (the entire Brazil), such as water bodies or bare land. The spatial distribution of land uses and covers corresponding to the aforementioned selection in the year of 2019, is portrayed in Fig. 2.

2.2.2. Land use and cover data preparation

In order to prepare the land use and cover data for application in the ecological footprint assessment, the six raster files covering the Brazilian territory were processed separately in the ArcGIS software of ESRI (https://www.esri.com/), as follows: (*i*) the raster image was re-projected onto the SIRGAS 2000 coordinate system, a geodetic reference for the Americas, and then the pixels contained in the Brazilian territory were selected using the *Extract by Mask* tool; (*ii*) the *Merge* function was used to assemble the re-projected and clipped images onto a single land use

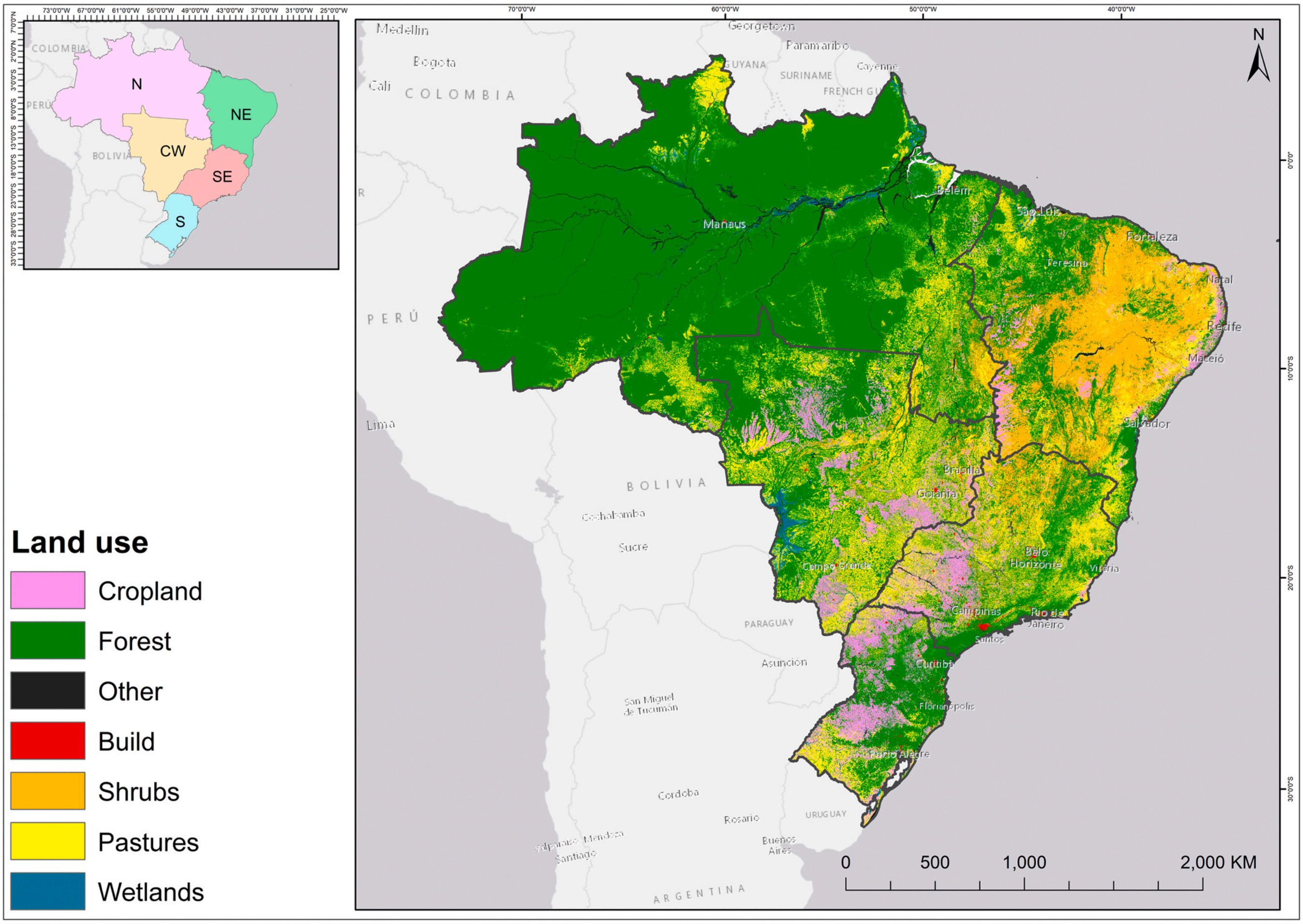

Fig. 2. Land use and cover map of Brazil in 2019, with the identification of classes used in the land use ecological footprint assessment, overlayed by the macro regions. The maps representing the entire studied period (2015 – 2019) are provided as Fig. S2 in the Supplementary Materials. Source: https://www.copernicus.eu/en.

and cover map, and then the map was reclassified according to the 7-class system described in Section 2.2.1; (*iii*) a zonal analysis was carried out inside the 5 macro regions of Brazil (Fig. 1), using the *Zonal Histogram* function that counted the pixels representing the various uses in the various regions. The tool calculates the area of land use and cover p (LU_p) within a region using Eq. (1):

$$Area\quad (LU_p) = \left(\sum_{i=1}^{n} pixel_{i,p}\right) \times 0.01 \tag{1}$$

where n is the number of pixels of that use counted in the target region and the constant term (0.01) is the pixel area in square kilometers; (*iv*) step (*iii*) was repeated for every macro region and all the previous steps were replicated for every year in the 2015 – 2019 period. The dataset of land use and cover areas per macro region and year is presented in Table 1.

2.2.3. *Demographic distribution data retrieval and preparation*

The demographic data was downloaded from the Brazilian Institute for Geography and Statistics (in Portuguese: Instituto Brasileiro de Geografia e Estatística; https://www.ibge.gov.br). The time series matched the one used for land use and cover (2015 – 2019). The data was retrieved at the municipality scale and then aggregated at macro region scale (Table 1).

2.2.4. *Spatial and temporal resolution issues*

The data on land use and cover have an explicit spatial resolution, because they correspond to a count of pixels with a predefined dimension located inside a predefined region. The Copernicus raster files used in this study were delivered at 100 × 100 m resolution. The area covered by a specific land use or cover within a macro region is the number of pixels counted inside that region, multiplied by 10.000 m^2. On the other hand, population data have an implicit spatial resolution, because they correspond to a scalar assigned to a predefined space. In the current study, they are the people living in a macro region regardless of where they exactly live inside that region and whether the population is well distributed or concentrated in specific places. The pixel count of specific land uses or covers, as well as the population counts, neglect spatial variability inside the selected region. This makes the analysis scale dependent. And this is particularly true for *LUEF* because this metric depends on both the pixel and population counts. Thus, the results presented in the current regional assessment are valid for the macro region scale, but cannot be directly transposed to finer resolution scales (e.g., the municipality scale).

Currently, the Copernicus coverage of land use spans solely the 2015 – 2019 period (https://land.copernicus.eu/global/products/lc; assess in 20 November 2023), which is relatively outdated. This is, however, no reason to question the study's pertinence or opportunity, because the main goal was mostly to detect and interpret recent trends in the *LUEF* over the entire territory of Brazil rather than evaluate this ecological footprint indicator on a specific date.

Table 1
Spatio-temporal distribution of population and land uses in Brazil, in the 2015–2019 period, with discrimination of macro region (see also Figure S3 in the Supplementary Materials). The population numbers represent millions of habitants whereas the land use or cover numbers represent percentage of coverage. The macro regions refer to those indicated in Fig. 1.

Year	2015	2016	2017	2018	2019
Macro region	North (3859,076 km^2)				
Population	17.5	17.7	17.9	18.2	18.4
Forest	85.1	85.2	85.2	85.2	85.1
Shrubs	1.9	1.9	1.9	1.9	1.9
Wetlands	1.1	1.3	1.4	1.5	1.5
Pastures	4.5	4.5	4.4	4.3	4.3
Cropland	0.4	0.4	0.4	0.4	0.4
Built environment	0.1	0.1	0.1	0.1	0.1
Other	7.0	6.7	6.7	6.6	6.7
Macro region	Northeast (1556,677 km^2)				
Population	56.6	56.9	57.3	56.8	57.1
Forest	31.3	31.8	31.8	31.8	32.1
Shrubs	39.8	40.4	40.3	40.0	38.6
Wetlands	0.4	0.6	0.7	0.8	0.9
Pastures	19.8	20.1	20.0	20.0	20.5
Cropland	6.1	6.2	6.3	6.4	6.9
Built environment	0.4	0.4	0.4	0.4	0.4
Other	2.2	0.6	0.5	0.5	0.6
Macro region	Midwest (1608,812 km^2)				
Population	15.4	15.7	15.9	16.1	16.3
Forest	47.5	47.6	47.4	47.4	47.3
Shrubs	5.5	5.5	5.4	5.4	5.4
Wetlands	1.9	2.1	2.3	2.5	2.6
Pastures	32.1	32.1	31.9	31.6	31.5
Cropland	11.9	12.0	12.1	12.3	12.3
Built environment	0.2	0.2	0.2	0.2	0.2
Other	0.9	0.6	0.6	0.6	0.6
Macro region	South (564,375 km^2)				
Population	29.2	29.4	29.6	29.8	30.0
Forest	72.0	72.6	72.4	71.2	72.0
Shrubs	1.7	1.7	1.7	1.6	1.6
Wetlands	4.2	4.7	5.0	5.3	5.7
Pastures	6.2	6.2	6.1	6.0	6.1
Cropland	7.7	7.7	7.7	7.6	7.7
Built environment	3.1	3.1	3.1	3.1	3.1
Other	5.2	4.0	3.9	5.2	3.8
Macro region	Southeast (925,936 km^2)				
Population	85.7	86.4	86.9	87.7	88.4
Forest	37.5	37.4	37.4	37.4	37.5
Shrubs	12.0	12.0	12.0	11.9	11.8
Wetlands	0.5	0.5	0.6	0.7	0.8
Pastures	34.1	34.0	33.9	33.7	33.5
Cropland	13.4	13.5	13.6	13.8	13.9
Built environment	1.4	1.4	1.4	1.4	1.4
Other	1.2	1.2	1.1	1.1	1.1

2.3. Land use ecological footprint approach

This study proposes the Land Use Ecological Footprint (*LUEF*) index to estimate the ecological footprint of a land use within a political administrative region. The index relates the footprint with the distribution of people per use or cover in the region and is measured as ratio of land use or cover area (LU_p) over the region's population (P_r), in a predefined timeframe:

$$LUEF_p = \frac{LU_p}{P_r}, \text{for every region } r \text{ and land use or cover } p \quad (2)$$

In practice, the *LUEF* tracks the potential human demand for ecosystem services retrievable from a specific land use or cover within the target region, based on the overlay of land uses or covers and population distributions. It is worth mentioning that the *LUs* to consider will be of two types: those with ecological function and those with socio-economic function. For example, in the macro regions of Brazil where the *LUEF* approach will be tested (Fig. 2), the *LUs* of forest, shrubs and wetlands are likely of ecological function, whereas the *LUs* of cropland, pastures and build environment represent the socio-economic function. In both cases, a low *LUEF* will mean a stronger pressure of humans over the respective *LU* and a high *LUEF* a weaker pressure.

When applied to a time series of land uses or covers and corresponding population distributions, the analysis of *LUEF versus* year diagrams allows identifying trends, which can be of no change, decrease or increase. The interpretation in each case will depend on the combined analysis of *LUEF*, *LU* and *P* variations as function of time, because different trends of *LU* and *P* can lead to similar trends of *LUEF* (the ratio analysis problem). Under the growth trend of population observed in Brazil in the past years, however, the interpretation becomes more straightforward. In that scenario, the no change case implies that population growth has caused a proportional *LU* increase, meaning that *LU* increase was mostly population-controlled. On the other hand, the decrease case indicates a pressure increase of population over the *LU*, which has not changed or changed more slowly than the population in the same period. And finally, the increase case points to faster changes of *LU* relative to *P*, meaning that the first are likely controlled by factors other than population growth. The environmental implications (ecological footprint) of all these patterns will depend on whether the *LU* changes affected uses with ecological or socio-economic function, as will be better understood in the forthcoming discussion.

2.4. Statistical analysis

Following the calculation of *LUEF*s for all the *LU*s and macro regions in Brazil, the *LUEF*s from each region were plotted against the measurement year and a linear or nonlinear function was fitted to the scatter points to look for trends. Linear or nonlinear regression analyses were also used to discuss causality among the *LUEF*s and potentially related independent variables (e.g., gross domestic product, carbon stocks in top layers of soil, etc.). A hierarchical cluster analysis based on Euclidean distances and the Ward's (1963) agglomerative method, was used to look for a global assessment of footprint interactions, namely between those of ecological function and those of socio-economic function. The cluster analysis was also used to see how the macro regions globally behaved as regards the ecological footprint.

3. Results

The calculated *LUEF*s are illustrated in Fig. 3 (see also Table S1 in the Supplementary Materials). The forest cover presents the highest *LUEF*s in Brazil and in all of its macro regions, with the exception of Northeast region where the shrubs prevail. Thus, the ecological function dominates in the country. There is, however, a sharp decreasing trend from the North region where the average forest *LUEF* is 15.1 ha/hab, to the Midwest (4.1 ha/hab), and finally to the Northeast, South and Southeast regions (0.35 – 0.72 ha/hab). As mentioned above, the ecological function of shrubs dominates in the Northeast (0.91 ha/hab), sustained by the Caatinga biome (Fig. 1), but is also well represented in the Midwest (0.47 ha/hab) and the Southeast (0.11 ha/hab), supported by the Cerrado biome. Finally, the ecological function of wetlands is marked in the Midwest (0.20 ha/hab) by the Pantanal, in the North (0.24 ha/hab) by the Amazon, and in all the macro regions by the riparian forests. As regards the *LU*s of socio-economic function, the results show higher *LUEF*s allocated to pastures, when compared to the *LUEF*s of croplands, meaning that the croplands at disposal of humans for food production are less available than the pasturelands accessible to raise livestock. The difference between these two *LUEF*s replicates across the country (with the exception of South region), namely in the North (0.78 > 0.07 ha/hab), Northeast (0.46 > 0.15 ha/hab), Midwest (2.76 > 1.05 ha/hab) and Southeast (0.32 > 0.13 ha/hab). In the South, the *LUEF*s of pasture and cropland are similar (around 0.05 ha/hab). This is also the region with least pasture and cropland availability *per capita*, considering the very low *LUEF*s. The *LUEF* related with the urban and industrial uses are similar across the macro regions, being in the 0.01 – 0.02 ha/hab range.

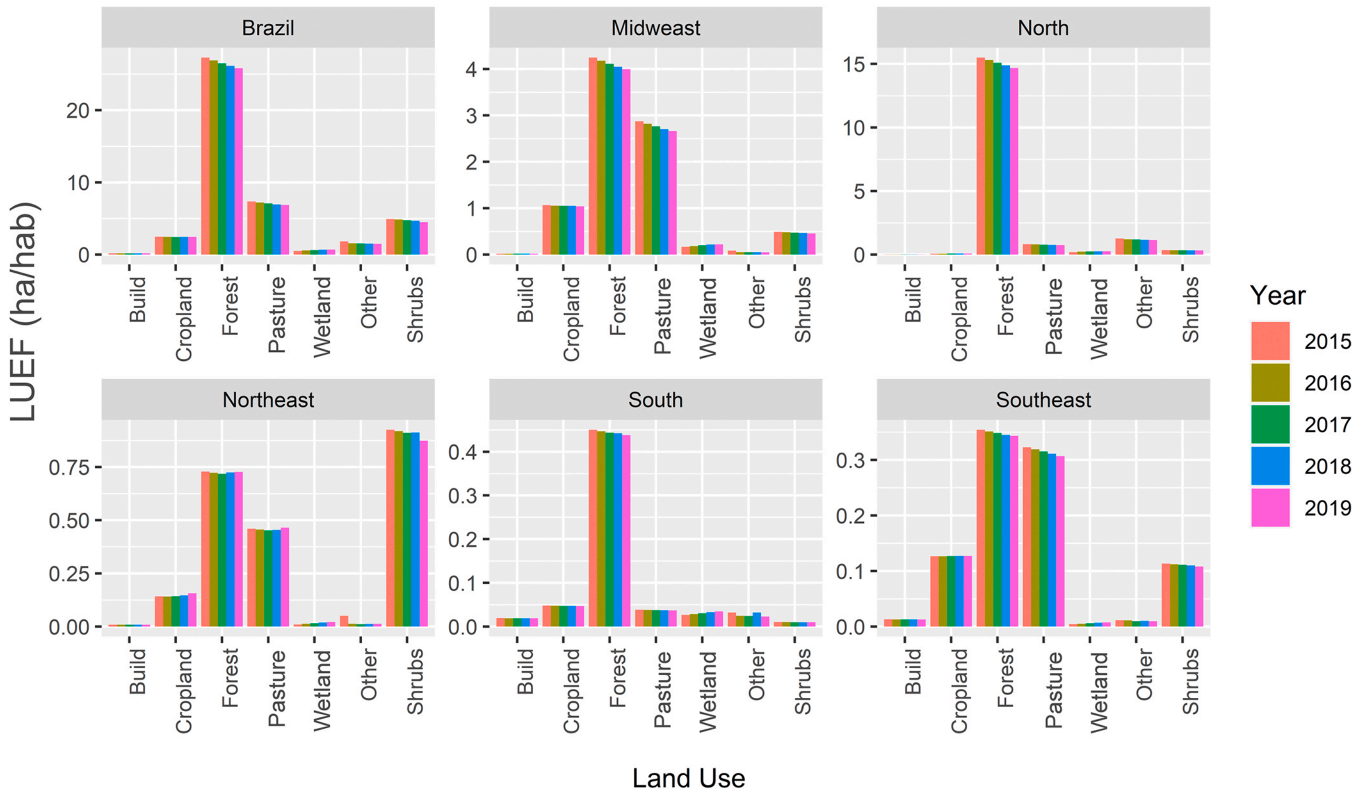

Fig. 3. Land Use Ecological Footprint (*LUEF*) of main land uses and covers in Brazil, with discrimination of macro region and year (2015 – 2019 period). The *LUEF* numbers used to draw the figure are listed in the Supplementary Materials (Table S1).

The regression analyses performed between the various *LUEF*s and the measurement year are illustrated in Fig. 4 (see the statistical parameters in Table S2 of Supplementary Materials). In the studied period (2015 – 2019), the population in the macro regions increased continuously at rates between 180,000 (in the South) and 660,000 (in the Southeast) habitants per year (Table 1), with the exception of Northeast region that experienced a fluctuation pattern (increase between 2015 and 2017, decrease in 2018 and increase again in 2019).

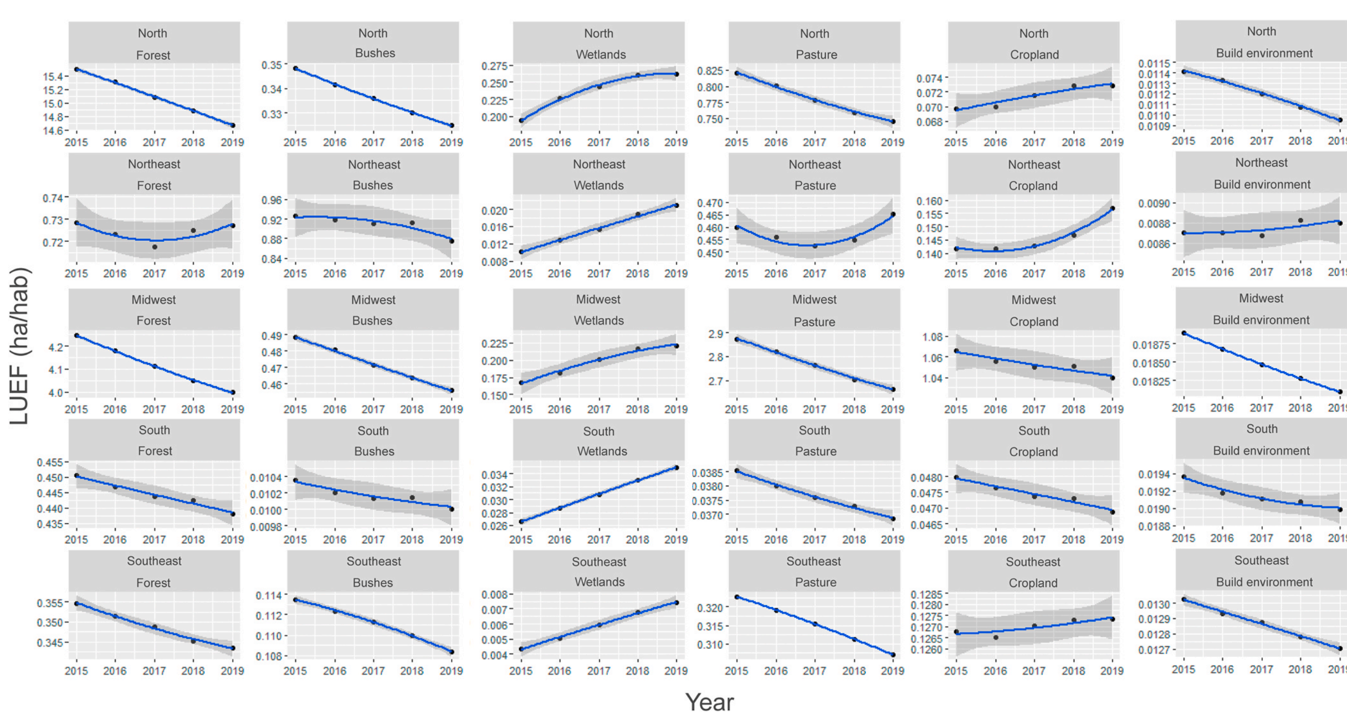

Fig. 4. Regression analysis performed between the *LUEF*s of forest, shrubs, wetlands, pastureland, cropland and build environment, and time considering the 2015 – 2019 period. The regression parameters (linear coefficient, angular coefficient) and performance indicators (R^2, *p*-value) are listed in Table S2 (Supplementary Materials).

The evolution of population had reflexes on the regression results. As per the numbers in Table S2, the regressions were mostly significant where the population grew continuously and essentially non-significant in the Northeast region. Thus, the subsequent descriptions will be restricted to the North, Midwest, South and Southeast regions. In these areas, the *LUEF*s of forest, shrubs, pasture and build environment generally decreased in the 2015 – 2019 period. These fallings indicate potential threats to the ecological function of forests and shrubs, as well as pressure increases on livestock production and urbanization / industrialization, because the corresponding available lands became scarcer *per capita*. The wetland's *LUEF* increased across the four regions while the cropland's *LUEF* increased in the North, Northeast and, to a smaller extent, the Southeast macro regions. These risings imply the conversion of other uses to produce more wetland and cropland. In keeping with the results of hierarchical clustering, the *LUEF* pattern of forests has been markedly different from the other *LUEF*s (Fig. 5a). Besides, the North and Midwest regions aggregate on one side of Fig. 5b, meaning that their *LUEF* patterns are similar, the same being observed for the Northeast, South and Southeast regions.

4. Discussion

4.1. General appreciation of LUEF estimates

The *LUEF* of forests and shrubs decreased linearly in the 2015 – 2019 period and that was probably a result of continued deforestation. Fig. 6a illustrates deforestation in 2019 at the state level, combined with the percentage of deforestation growth in the various biomes in the 2019 – 2022 period (see biome distribution in Fig. 1). The map was prepared with data from the latest report of MapBiomas alert released in 2023 (http://alerta.mapbiomas.org/), and makes evident how active deforestation was in periods close to the 2015 – 2019 period. The deforested area was larger in the North (736,591 ha) and Midwest (262,528 ha) regions, while the 2019 – 2022 growth has affected mostly the Caatinga (910.2%) and Pampa (377.9%) biomes. The larger forest *LUEF*s of North and Midwest regions are likely the cause of their aggregation in Fig. 5b, while the larger forest *LUEF*s probably explain why they were isolated from the other *LUEF*s in Fig. 5a.

The evolution of forest *LUEF* portrayed in Fig. 4 (decline) probably describes deforestation as consequence of economic development (e.g., cropland expansion) and/or urbanization. A similar pattern was recognized in various recent studies (Barbosa et al., 2023; da Silva et al., 2023; Mullan et al., 2021; Santos et al., 2022). And the latest MapBiomas alert report (http://alerta.mapbiomas.org/), which summarized the prominent vectors of deforestation, also highlighted the impact of crop and livestock production on deforestation assigning a 98.6% share to these economic activities. In the present study, the *LUEF* of cropland did observe an increase in the North, Northeast and Southeast macro regions (Fig. 4) corroborating *LU* conversions to cropland in those regions. The *LUEF* of cropland did not increase in the Midwest and South macro regions, but was reported to occur in the Midwest's Mato Grosso state in a study conducted by (Vieira et al., 2022). In that study, the authors referred a threefold increase of cropland in the 2000 – 2018 period, at the expense of a Cerrado biome vegetation decline, explaining the expansion with the growth of soy and corn prices and consequent production demand. Thus, our results for the Midwest were striking. Eventually, population has increased more rapidly in these regions than the corresponding cropland expansion, explaining the decrease of cropland *LUEF* in the 2015 – 2019 period. A similar explanation could be presented for the general decrease of pasture *LUEF*.

The urban *LUEF* in 2015 varied between 87 m^2/hab in the Northeast and 194 m^2/hab in the South. In the next five years, it increased by 1% in the Northeast and decreased between 2% and 4% in the other macro regions (Fig. 4 and Table S2 of Supplementary Materials). The *LUEF* values are consistent with those reported in (Gao and O'Neill, 2020), who estimated urban land for Brazil in 2000 to vary between 72 and 117 m^2/hab. On the other hand, the reduction of urban land *per capita* combined with the growth of population, in the North, Northeast, South and Southeast regions, denotes urban densification in these regions (Li et al., 2022). Thus, in the 2015 – 2019 period, Brazil has generally

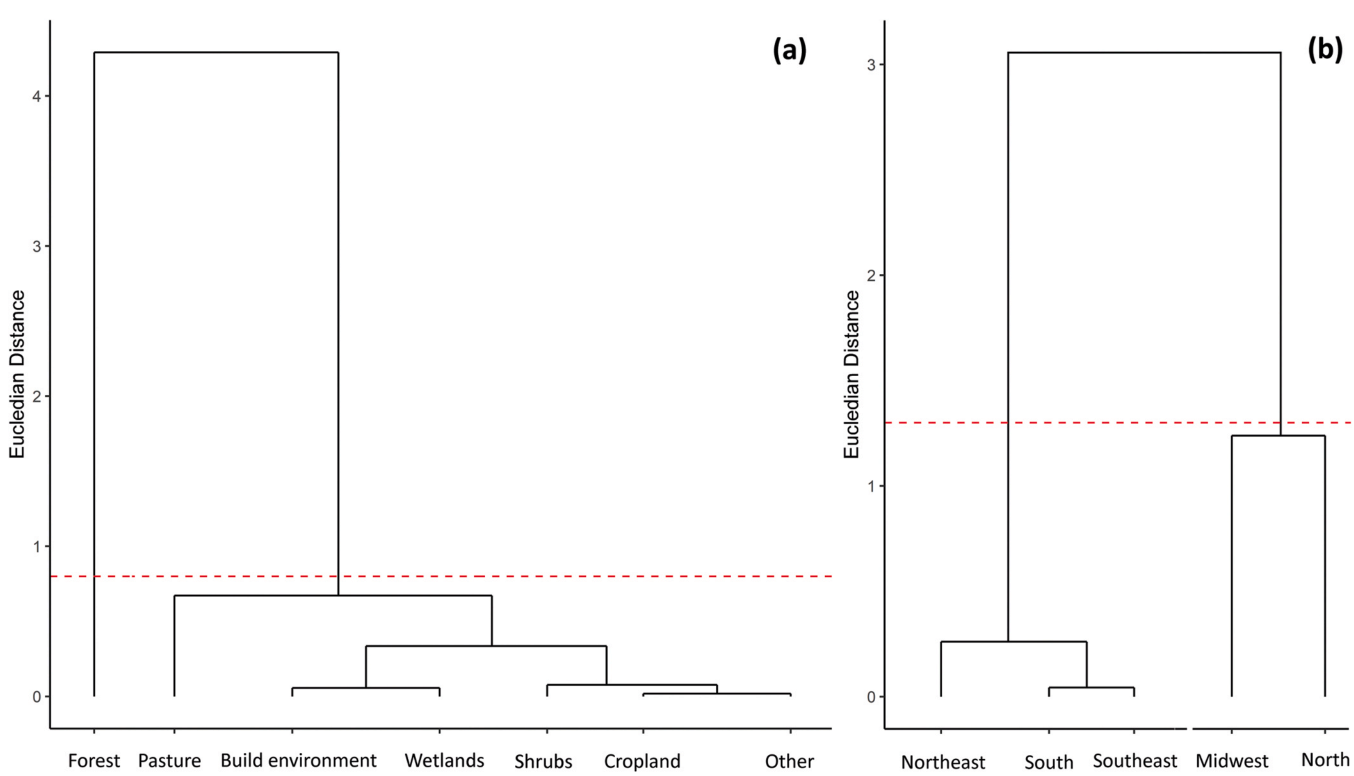

Fig. 5. Hierarchical clustering of *LUEF*s aggregated per (a) land uses and (b) macro regions.

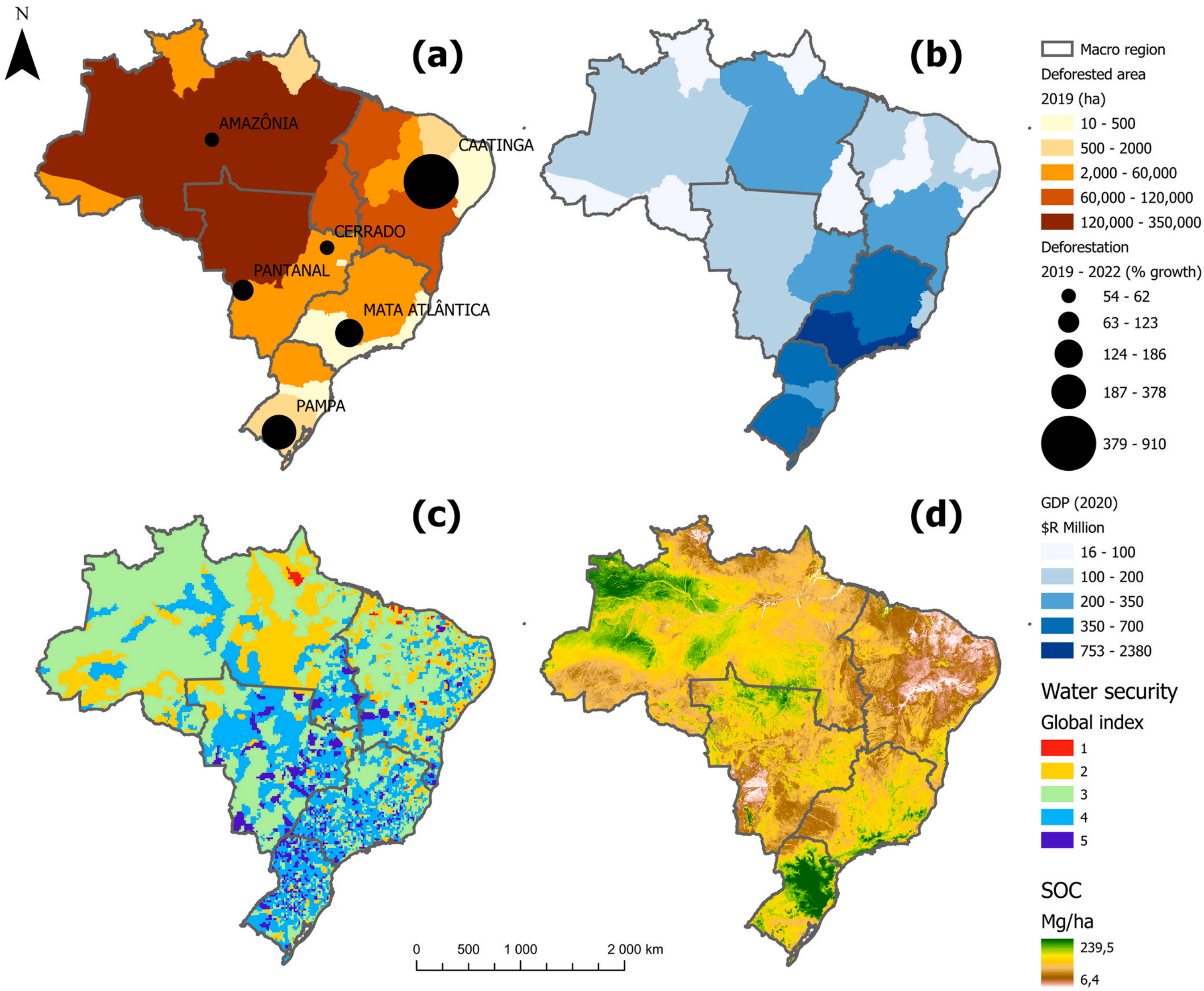

Fig. 6. Spatial distribution of (a) deforested area in 2019 at state level, coupled with deforestation growth in the 2015 – 2019 period at biome level; (b) gross domestic product (GDP) in 2020 at state level; (c) water security in 2021 at municipality level; (d) organic carbon stock (0 – 30 cm depth) assessed in 2021 at 1 km spatial resolution. Additional information provided in Section 4.2.

moved more people into more dense urban centers, with the exception of Northeast region that might have experienced some urban expansion because a stable population (around 57 million) was settled on a region with slightly growing *LUEF* (from 87 to 88 m^2/hab).

The increase of wetland *LUEF* across all the regions (Fig. 4) was probably a reflex of riparian forest restoration. A number of examples from Brazil can be retrieved from the scientific literature that describe active or passive restoration programs of riparian corridors in various landscapes, namely sugar cane fields (Bieluczyk et al., 2023), mining areas (Barbosa et al., 2023), water reservoirs (Cortez-Silva et al., 2020), forest plantations (Fockink et al., 2022; Zangalli et al., 2023), urban areas (Ribeiro et al., 2022). Compliance with the Forest Code (Brazilian Federal Law no. 12651/2012) was likely the legal trigger of these riparian forest restoration initiatives, and the *LUEF* results (steady wetland increase) suggest an efficient implementation of this legislation at the macro region scale in the studied period.

4.2. Causality among LUEF change drivers and ecosystem service proxies

The previous section exposed relationships between the various *LUEF*s and potential drivers of *LUEF* change overtime. With the purpose of deepening that analysis, the drivers are now compared with potentially affected ecosystem service proxies. Deforestation was selected as key driver of *LUEF* change, because it is unquestionable in Brazil (Fig. 6a). The gross domestic product (GDP) complemented deforestation as main driver, because the GDP aggregates the financial result of all economic activities, with more than probable impacts on the *LUEFs*. As regards proxies of ecosystem services, water security was selected to represent a provision service while the soil carbon storage was adopted to represent a support / regulation service. Cultural or recreational proxies were not considered in the causality analysis. The selection of drivers and proxies had also in mind the availability of data at resolutions better than the macro region (e.g., at the municipality level), so the aggregated values at the macro region could smooth uncertainty and hence be seen as reliable. The spatial distributions of all selected drivers and proxies are illustrated in Fig. 6, whereas the macro region averages are summarized in Table 2.

As will be seen in Sections 4.2.1 to 4.2.3, the causality is expressed as linear regressions (Figs. 7 to 10). In spite of being possible to admit other more complex (e.g., non-linear) models, the current study did not consider them for two reasons: (1) with a single exception, the coefficients of determination resulted from the linear regressions were

Table 2
Information on deforestation (in 2019), gross domestic product (GDP in 2020), water security (in 2021) and soil organic carbon storage (reported in 2021), aggregated at the macro region scale.

Driver or proxy	Unit	North	Northeast	Midwest	South	Southeast	Data source reference
GDP in 2020	$R million	478	1079	791	1308	3953	https://ibge.gov.br/
Deforestation in 2019	ha	736,591	191,609	262,528	3803	27,038	http://alerta.mapbiomas.org/
Water Security in 2021	dimensionless	2.9	3.1	3.6	3.8	3.6	https://dadosabertos.ana.gov.br/
Organic carbon stock	Mg/ha	44.3	30.4	41.0	56.1	43.8	http://geoinfo.cnps.embrapa.br/

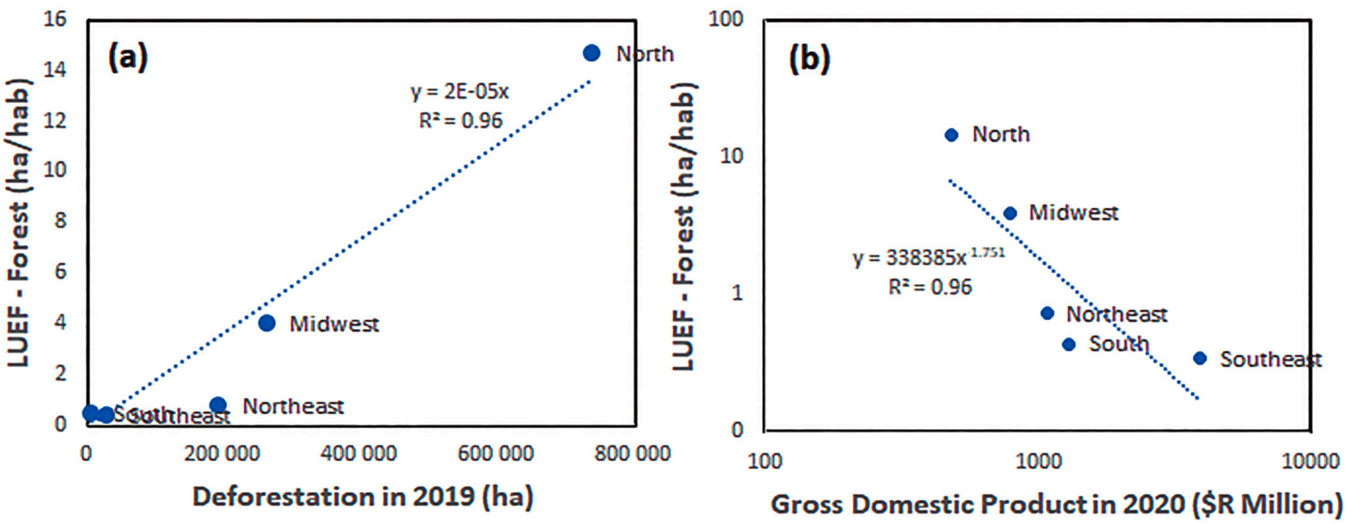

Fig. 7. Causality among the *LUEF* of forest and (a) the total deforested area in 2019, or (b) the gross domestic product in 2020.

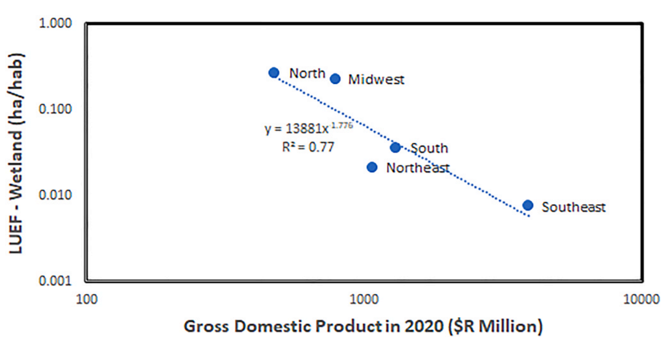

Fig. 8. Causality among the domestic product of 2020 in the macro regions of Brazil and the average *LUEF* index of wetlands in the 2015 – 2019 period.

high (≥ 0.7) to very high (≥ 0.9); (2) the number of points plotted in the scatter diagrams that illustrated the causal effects (Figs. 7 to 10) were small (just five, representing the five macro regions). This refrained us from envisaging cause-and-effect relationships other than the simplest one – the linear regression. This causality issue is inherently a scale problem. More complex cause-and effect relationships could be detected at more detailed resolutions (e.g., the municipality resolution), where the aforementioned scatter plots would be populated with a larger number of points potentially affirming the linear or pointing to other relationships. That analysis was, however, beyond the scope of our more regional assessment, and for that reason was not investigated in this study.

4.2.1. Deforestation and reforestation

Deforestation in Brazil in the 2015 – 2019 period occurred preferably where the availability of forest land was higher, meaning where the *LUEF*s were larger (Fig. 7a). This would be the expected result, but an exception has occurred in the Northeast region where deforestation was larger than anticipated from the *LUEF*. Indeed, a close look into Fig. 7a exposes the deviation of Northeast's blue point from the trend line, namely towards higher-than-expected deforestation (to the right). The 2023 report of MapBiomas alert (http://alerta.mapbiomas.org/) also warned for the largest percent growth of deforestation in the Caatinga biome in the 2019 – 2022 period (910%; cf. Fig. 6a), which is prevalent in the Northeast. The linkage of forest *LUEF* decline to gross domestic product (GDP) increase is evident in Fig. 7b, which reveals a power function relating the growth of GDP to the drop of forest land availability *per capita*. A previous study found a similar relationship (Santiago and Couto, 2020). In that study, the authors asked themselves whether the increase in income caused by deforestation tends to decrease because a region has developed or because there are no longer any forest resources. They came to the conclusion that the drastic decline in native forest resources appears to determine their decoupling from economic development. A study conducted in southern Amazonia between 2010 and 2019 linked deforestation with economic growth in the livestock sector (Santos et al., 2021), because stock farming is practiced extensively and hence depends on land take possibility. The work of Trigueiro and co-authors (Trigueiro et al., 2020) used geographically weighted regression to assess the relationship between increasing deforestation and predictor variables, highlighting the influence of various economic development parameters, such as greater access to rural credit concession in the Northeast or the distance to roads in the North and Northeast regions. The link of increasing deforestation to proximity of nearest roads, cities, and ports was also seen in a study focused on the Atlantic Forests of Argentina, Brazil and Paraguay (Mohebalian et al., 2022); while the link of increasing deforestation to regular road network development, coupled with its integration with the clandestine roadway complex, was assessed with machine learning methods in the oriental Amazon (das Neves et al., 2021). Collectively, these evidences of a coupling between increasing deforestation and economic development in Brazil corroborate our regression between decreasing *LUEF* and increasing GDP.

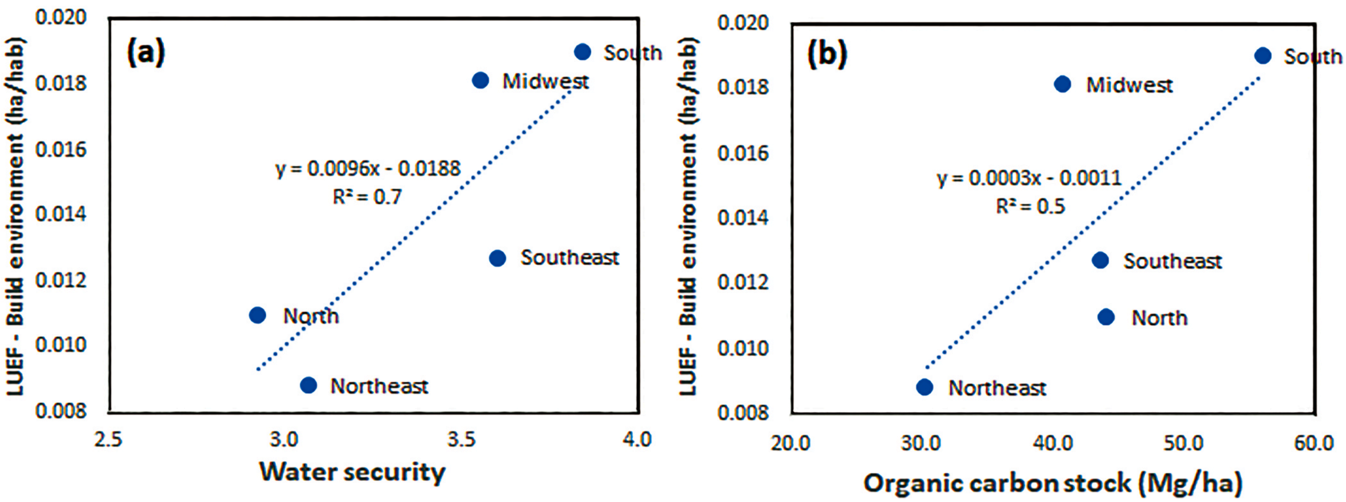

Fig. 9. Causality among the *LUEF* of build environment and (a) a provision service proxy (water security), and (b) a regulation service proxy (organic carbon stock).

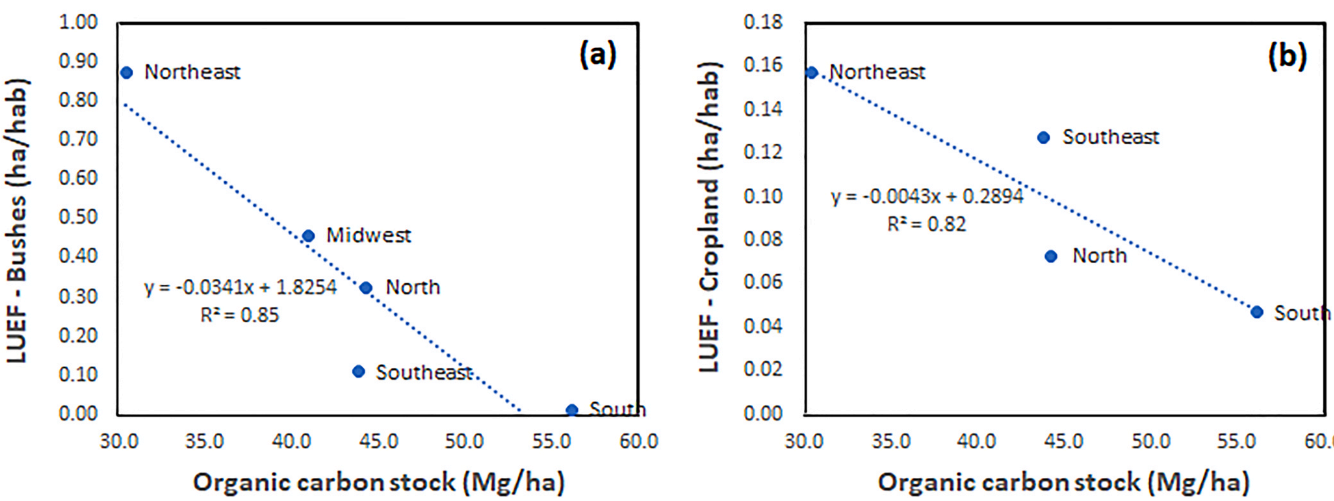

Fig. 10. Causality among organic carbon stock and (a) the *LUEF* of shrubs; (b) the *LUEF* of cropland.

Restoration of riparian forests was the explanation given above for the increase of wetland *LUEF* overtime (Fig. 4). That was a promising result, but Fig. 8 shows a negative regression between the gross domestic product of macro regions in 2020 and the *LUEF* of wetlands. Thus, in spite of all the efforts to restore riparian forests, economic growth was still a threat to these ecological and water quality buffers in the studied period.

4.2.2. *Urban densification*

Urban densification was detected in our results and is often viewed as route to sustainable urban growth, because it tends to decelerate land take (Herdt and Jonkman, 2023; Mohajeri et al., 2023) and minimize urban development costs (Digafe et al., 2023). However, there are also negative impacts to acknowledge, namely as regards the urban's thermal, illumination and ventilation environments. For example, in central Wuhan, China, urban densification has led to warming and increase in the heat stress, besides other effects (Deng et al., 2023). Another example, now from Trondheim, Norway, revealed that an urban densification project proposed for the development of a university campus, would reduce solar accessibility in the outdoor area below the recommended minimum. Finally, in the Gangnam district in Seoul, South Korea, the development of urban superblocks reduced ventilation in the inner blocks (Maing, 2022). Other impacts of urban densification include higher rents subsequent to revamping in favor of density (Gerber and Debrunner, 2022), stress on the performance of buried water infrastructure (Kaur et al., 2022, 2020), among others. In Brazil, the current study detected significant causality among the *LUEF* of build environment and two ecosystem service proxies, namely water security (provision service proxy) and soil organic carbon stock (regulation service proxy). Fig. 9a,b expose a decline of water security and carbon stocks for increasing urban densification (smaller *LUEF*s). The impacts of urbanization on water security and carbon stocks have been recognized before. As regards water security, the observed decline likely results from an amplified supply – demand gap, more difficult asses to securely managed water and sanitation services, enhanced deterioration of water quality, among other issues that develop along with population concentration, as was recognized for various urban centers undergoing densification (Hoekstra et al., 2018; Norman et al., 2013). The lack of improved water supply infrastructure is frequently ignored as cause of water stress in densified urban centers (McDonald et al., 2014; Padowski et al., 2016), and can also be a source of water security decline in Brazilian densified cities. In the case of carbon stocks, the impacts are likely related with soil sealing. In that context, a study conducted in three French cities (Cambou et al., 2023) revealed that soil organic carbon stocks, in particulate organic matter fractions, were 25–48 times higher in park than in sealed soils, and similar results were obtained in urban areas of Milan, Italy (Canedoli et al., 2020). Soil organic carbon stocks were also compared across urbanization gradients in a dryland region, with the following results: ~ 55 Mg/ha under impervious surface, ~ 60 Mg/ha in bare land, ~ 80 Mg/ha in urban green space and ~ 100 Mg/ha in cropland (Yan et al., 2015). Some studies refer the potential of green areas established in the urban environment to mitigate the effects of soil sealing (Vasenev et al., 2018), but that feature was not captured in our case. An interesting result from Fig. 9a is that the build environment's *LUEF* needs to be > 100 m^2/hab to ensure a moderate water security (level 3; Fig. 5a) at the macro region scale. Besides, if that *LUEF* component would grow above 130 m^2/hab, the carbon stock would approach the average of Brazil's soils (46.9 Mg/ha, at the 0 – 30 cm depth; Vasques et al., 2021). These reference *LUEF*s are merely indicative but nevertheless are useful indicators (metrics) for landscape planning and policy.

4.2.3. Shrubs, croplands and organic carbon stocks

The *LUEF*s of shrubs and cropland both presented a strong negative causality with organic carbon stocks assessed at depths < 30 cm ($R^2 > 0.8$) in the 2015 – 2019 period, as illustrated in Fig. 10a,b. The results for shrubs indicate that the regions where there is larger occupation by shrubs *per capita* are those where the storage is smaller, and those where the occupation is sparser represent better pools for organic carbon storage. This is striking, because shrubs are natural pools of organic carbon and so a larger occupation *per capita* should lead to a larger storage. The explanation for the results probably relies on the type of shrub vegetation predominating in the various macro regions, which is mediating the observed trend. For example, the south region has a very small shrub *LUEF* but is largely occupied with Atlantic Forest vegetation (Fig. 1). A recent study compiled information on organic carbon stocks in soils from Brazilian biomes (de Souza Medeiros et al., 2022), which revealed the largest stocks in soils from the Atlantic Forest biome (96.4 Mg/kg, at depths < 30 cm; Table 3). On the other hand, the Northeast region has the largest shrub *LUEF* but is mostly occupied with Caatinga vegetation where the soils stock nearly half (46.4 Mg/ha) the organic carbon stored in the Atlantic Forest soils. The stock differences across biomes may depend on multiple factors, but the content of clay in the soils seem to play a vital role, as highlighted in the aforementioned work (see Table 3, which shows clay contents 4.7 × larger in the Atlantic Forest soils relative to the Caatinga soils) but also in other studies (Grüneberg et al., 2013; Jantalia et al., 2007; Osman et al., 2023; Peng et al., 2023). The trend observed in Fig. 10b says that the larger is the area available for (or occupied with) agriculture, the smaller are the organic carbon stocks, and that the South and Northeast, again, represent the end member regions. Eventually, the resemblance between Fig. 10a and Fig. 10b means that agriculture has expanded significantly at the expense of shrubland conversion, in the 2015 – 2019 period. On the other hand, the decline of organic carbon stocks for increasing cropland *LUEF*s suggests a stock loss in the conversion of a previous use (e.g., native vegetation) into cropland. The results of (de Souza Medeiros et al., 2022) are clarifying in that regard. As per the values in Table 3, a trendline $\Delta_{SOC} = 2.4 - 0.62 \times t$, with $R^2 = 0.85$, could be fitted to the difference between the carbon stocks after and before the native vegetation to cropland conversions (Δ_{SOC}; deduced from columns 5 and 6 in Table 3) and the time that has passed since then (*t*; column 7). The trendline is clear ($R^2 > 0.8$) in showing that organic carbon stocks in the 0 – 30 cm layer may drop by some 0.62 Mg/ha every year after the conversion. Other studies also sustained carbon losses resulting from native vegetation to cropland conversions (Locatelli et al., 2022). Following the rationale used to define some thresholds for the build environment *LUEF* (Section 4.2.2), the trendline of Fig. 10b indicates the need of keeping cropland *LUEF* < 900 m^2/hab to achieve organic carbon stocks in the 0 – 30 cm layer of soils around the National average (46.4 Mg/ha).

Table 3
Organic carbon stocks in soils (SOC) from Brazilian biomes, assessed at depths < 30 cm at *n* locations. Additionally, it is informed the carbon stock observed after conversion of natural vegetation into cropland, considering the number of years after the conversion, and the clay content of soils. The data was compiled from (de Souza Medeiros et al., 2022).

Biome	*n*	Clay content (g/kg)	Average depth (cm)	SOC stock (Mg/ha): native vegetation	SOC stock (Mg/ha): cropland	Period following land use conversion (yr)
Amazon	2	730.0	30.0	49.0	51.5	1.5
Atlantic Forest	2	634.0	22.5	96.4	66.3	45.0
Caatinga	31	134.6	14.7	46.4	40.1	9.5
Cerrado	23	548.1	23.0	55.1	45.4	17.9
Pampa	1	220.0	20.0	39.3	27.8	34.0

4.3. Policy considerations

Deforestation was (and still is) a motor of economic growth (Fig. 7b), but was (and still is) also accompanied by numerous environmental consequences such as amplified greenhouse gas emissions, water quality deterioration, biodiversity decline (Faria et al., 2023; Galán-Acedo et al., 2021; Galinato and Galinato, 2016; Kong et al., 2022), among other damages. As regards the greenhouse gas emissions, for example, the ongoing monitoring conducted by the SEEG platform (https://plataforma.seeg.eco.br/) shows how the emissions of CO_2 in Brazil since 1998 can be mostly explained by deforestation and described through a linear relationship $CO_2(t) = 0.335 \times \text{Deforestation}$ ($R^2 = 0.99$), where the CO_2 emissions are given in Gton/yr and the deforested area in Mha/yr. The monitored data also reveal contrasting scenarios before and after 2004, with emissions being 1.5 Gton/yr for a deforestation of 4.6 Mha/yr in the 1998 – 2004 period, and 0.9 Gton/yr for a deforestation of 2.6 Mha/yr in the 2005 – 2021 period. The observed reduction was probably the result of policy (Brazilian Forest code; Federal Law no. 12651/2012) and, more importantly, of action against deforestation derived therefrom, coupled with reforestation. Considering the successful outcome observed in the last couple of decades, it would be important to intensify the measures in the coming years to bring forests and the economy to a balance.

The latest report on the MapBiomas alert released in 2023 (http://alerta.mapbiomas.org/) showed how the action of public and private entities contributed to reduce deforestation across Brazil in the most recent years (2019 – 2022 period). The achievements were made possible through an interdisciplinary commitment involving many public institutions, namely the Brazilian Institute for the Environment and Renewable Resources (in Portuguese: IBAMA – Instituto Brasileiro do Meio Ambiente e dos Recursos Naturais Renováveis), the Chico Mendes Institute for the Conservation of Biodiversity (ICMBio – Instituto Chico Mendes de Conservação da Biodiversidade), the National Council of Legal Amazon (CNAL – Conselho Nacional da Amazônia Legal), the Brazilian Forest Service (SFB – Serviço Florestal Brasileiro), the National Foundation of Indigenous People (FUNAI – Fundação Nacional dos Povos Indígenas), the National Institute for the Colonization and Agrarian Reform (INCRA – Instituto Nacional de Colonização e Reforma Agrária), the state organizations of environment (OEMA – Órgãos Estaduais de Meio Ambiente), the state Public Ministries, and the Federal Police. Among the private actor, the commitment of finance institutions was vital because it ensured compliance of deforestation-related projects with the sector's rules and regulations, thus avoiding benefiting the commercial use of illegally deforested areas. The public action comprised infringement and/or embargo notices over 46.3% of all deforested area in the studied period (6.6 Mha), mostly from the federal institutions IBAMA and ICMBio (10.2%) and the state institutions Public Ministry and OEMAs (8.7% and 17.1%, respectively). In the Atlantic Forest biome, a specific deforestation combat program ("Mata Atlântica em Pé") detected nearly 15,000 ha of illegal deforestation between 2019 and 2022. The action against these infractions resulted in penalties that approached $R 161,800,000. The persistence on this combat could lead deforestation to a zero-level at the national scale, as was recently forecasted for the Amazon by (Silva et al., 2023). The years to come will show how Brazil will direct policies and action in that regard.

A parcel of deforestation that has occurred in the 2015 – 2019 period and has been assessed in the current study, was related with the suppression of riparian forests, namely for the expansion of pasture or cropland. The results of our study showed a recovery of riparian wetlands in that period (Fig. 4), which is certainly a positive outcome of deforestation combat. However, as farmland restoration jeopardizes agricultural production and the livelihoods of smallholder families, the Forest Code enforcement at this scale has been weak and a national debate over the code led to its 2012 revision, which severely limited the restoration required by the law. Alternatively, payments for ecosystem services (PES) are coming in the form of subsidies for agroecological activities (Richards et al., 2020; Trevisan et al., 2016) and, irrespective of some setbacks of PES programs, they are being implemented all over Brazil (Mota et al., 2023).

The payments for ecosystem services can help farmers to comply with the Forest Code rules but will barely be able to respond to the increasing food demand triggered by the growth of population. If that demographic trend continues, then additional measures are sought to implement, or intensify, namely productivity increases to provide more food in the same space. The results of our study showed an increase in the cropland *LUEF* in some macro regions (Fig. 4) and also revealed concomitant losses of organic carbon in the top layer of soils (Fig. 10b). Productivity increase and the implementation of adequate management practices are therefore the right path to stabilize the cropland *LUEF* and restore organic carbon stocks in the future improving soil fertility. The agriculture census (https://sidra.ibge.gov.br/) provides historical data on crop productivity. As regards cereals in the 2006 – 2023 period, the panorama in the macro regions were of marginal increases of planted area (Fig. 11a), with exception of Midwest region, but of expressive productivity increases (Fig. 11b). This is good news because it places Brazil in the right track. The reasons for the observed productivity increases rely on numerous positive factors (https://www.embrapa.br/), such as: (1) public policies, including (a) the Soy and beef moratoriums that hampered acquisition of these products from degraded areas; (b) the already mentioned Forest Code; (c) the ABC plan that, among other things, aim to encourage the adoption of sustainable production systems that increase the income of producers, especially with the expansion of the following technologies: recovery of degraded pastures; crop-livestock-forest integration and agroforestry systems; direct planting system; biological nitrogen fixation; and planted forests; (d) The National Program for Strengthening Family Farming (PRONAF), which aims to strengthen family farming through subsidized financing of agricultural and non-agricultural services. This program ensures the diversification of agricultural activities on family farms, enables entrepreneurship through the processing and agro-industrialization of food produced by family farming, as well as meets market requirements and the adoption of conservation practices for environmentally, economically and socially sustainable production; (2) Other measures, comprising rationalize the use of fertilizers, control processes, intensify the use of technologies with consequent sparing effect of scarce resources ("land sparing" effect), optimize investment ("land fixed cost" effect) and manage in a modern way resources and people for income generation, that is, to be an activity not opposed to sustainability. Together, intensification of these public policies and the action from the private sector, are expected to ensure food security in the country (compliance with Brazilian law no. 11346/2016) and export it worldwide.

Large-scale policy and action on controlling cropland *LUEF*, as discussed in the previous paragraph, can be scaled down to the urban areas. The results of our study pointed to urban densification in the Brazilian macro regions in the 2015 – 2019 period (Fig. 4), with negative consequences for water security (Fig. 9a) and for the stocks of organic carbon in soils (Fig. 9b). An effective policy to revert these negative outcomes relies on the expansion and intensification of urban agriculture supported by the National Program for Urban and Peri-urban Agriculture (https://www.gov.br/mds/pt-br/acoes-e-programas/inclusao-produtiva-urbana/agricultura-urbana). There are a number of ongoing initiatives in that regard distributed across the Brazilian territory (https://100politicas.escolhas.org/#main). Essentially based on educational gardens, the initiatives comprise capacitation courses for farmers; municipal policies that promote the opening of marketing channels, distribution and certification of products; promotion of production through the provision of land, infrastructure, fertilizers, and tax credits or incentives; participatory management and regulation of the activity. Concrete water security measures embedded in urban agriculture programs can involve implementation of rainwater harvesting for irrigation, which can cover nearly 20% of irrigation needs (Mahmoud et al., 2014), a shift to drop irrigation to increase water yields, or the use of hydroponics as alternative agriculture technique, which can reduce water requirements by nearly 30% (Rufí-Salís et al., 2020). As regards carbon stocks, the extra green cover brough by urban agriculture areas inevitably accounts for an increased biomass and hence the carbon stock of a region (Lwasa et al., 2014). From an urban planning perspective, however, if urban agriculture is implemented along with other interventions, such as green parks and urban forests, the impacts on increased carbon stocks could be more visible (Sagar et al., 2022).

In all the facets of land use footprints, Brazil has already implemented policies that can mitigate the concomitant negative impacts. Intensification of all those policies and their coordination across this continental country are the necessary steps to neutralize or, preferably, eradicate the footprints, thus granting human welfare and healthy

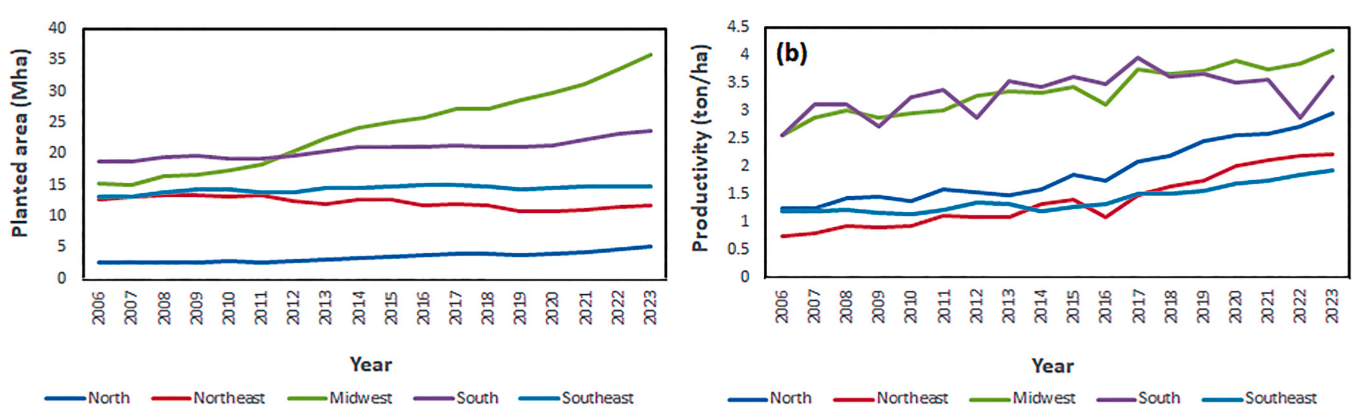

Fig. 11. Evolution of planted area and productivity of cereals in the macro regions of Brazil between 2006 and 2023. The source data were retrieved from https://sidra.ibge.gov.br/.

ecosystems to the future generations. It is worth recalling, however, that policies and plans are just frameworks aiming to contribute to a more sustainable and resilient urban environment. They are paramount to trigger and develop routes towards sustainability, but the concrete measures embedded in the urban agriculture and urban planning programs, coupled with their effective implementation on site, are the real deal, the key to the programs' success.

5. Concluding remark

An approach to assess the ecological footprint of land uses was presented in this study, termed *LUEF*. Standing on the ratio between the area of a certain land use or occupation, measured inside the boundaries of a political administrative unit, and the population of that region, the *LUEF* index was assessed in the five macro regions of Brazil: North, Northeast, Midwest, South, and Southeast. The *LUEF* index comprises various components, one for each major land use or occupation identified in the studied area. In Brazil, the components were: forest, shrubs, wetlands, cropland, pastures, and build environment. The time series of *LUEF* components spanning the 2015 – 2019 period, highlighted deforestation, urban densification, cropland / pasture increase, and riparian wetland restoration as main paths of *LUEF* change, and exposed environmental consequences derived therefrom, namely water security and soil organic carbon stock declines. In spite of these preoccupying results, monitoring data on various land use footprint indicators, such as cereal productivity, placed Brazil in a sustainability path manifest in continuous and expressive productivity increases since 2006. Sustainable land use is still a far-reaching goal, but the intensification of key policies was regarded in this study as fuel to keep it on the agenda, such as the soy and beef moratoriums, the Brazilian Forest Code, the ABC plan, Payments for Ecosystem Service programs, the National Program for Strengthening Family Farming, the National Program for Urban and Peri-urban Agriculture, among others. Given the continental scale of Brazil, with huge economic asymmetries and biome diversity, more important than the policies is their interdisciplinary implementation to avoid displacement of a problem from one state, region or biome to the neighbor. In the era of web information and communication, the competent authorities are sought to use these channels to monitor policy implementation across the country and homogenize them to ensure regional cohesion and overall success.

CRediT authorship contribution statement

Carlos Alberto Valera: Validation, Resources, Formal analysis. **Renata Cristina Araújo Costa:** Visualization, Methodology. **Fernando Pacheco:** Writing – review & editing, Validation, Supervision, Formal analysis. **Luís Filipe Sanches Fernandes:** Writing – review & editing, Validation, Formal analysis. **Teresa Cristina Tarlé Pissarra:** Validation, Supervision, Resources, Project administration, Methodology, Formal analysis, Conceptualization. **Rafael Parras:** Writing – original draft, Methodology, Investigation, Data curation, Conceptualization. **Gislaine Costa de Mendonça:** Visualization, Methodology. **Juan Ricardo Rocha:** Software, Investigation. **Luis Miguel Costa:** Methodology, Data curation.

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.

Data availability

Data will be made available on request.

Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at doi:10.1016/j.landusepol.2024.107121.

References

Ahmad, M., Jiang, P., Majeed, A., Umar, M., Khan, Z., Muhammad, S., 2020. The dynamic impact of natural resources, technological innovations and economic growth on ecological footprint: An advanced panel data estimation. Resour. Policy 69, 101817. https://doi.org/10.1016/j.resourpol.2020.101817.

Appiah, M., Li, M., Naeem, M.A., Karim, S., 2023. Greening the globe: Uncovering the impact of environmental policy, renewable energy, and innovation on ecological footprint. Technol. Forecast. Soc. Change 192, 122561. https://doi.org/10.1016/j.techfore.2023.122561.

Barbosa, R., de, S., de Matos Rodrigues, J.I., de Oliveira, V.P., Martins, W.B.R., Coelho, C.A.C., do Carmo, W.F.S., dos Santos, N.J.R., de Sousa, W.V., de São Paulo Aguiar, E.A.L., 2023. Restoration of riparian ecosystems posterior to tin mining in the Central Amazon: Restoration indicators and selection of suitable species for planting. Ecol. Eng. 193, 107007 https://doi.org/10.1016/j.ecoleng.2023.107007.

Barbosa, V.A., Nabout, J.C., Cunha, H.F. da, 2023. Spatial and temporal deforestation in the Brazilian Savanna: The discrepancy between observed and licensed deforestation in the state of Goiás. Land Use Policy 131, 106730. https://doi.org/10.1016/j.landusepol.2023.106730.

Bazan, G., 1997. Our Ecological Footprint: reducing human impact on the earth. Electron. Green J. 1 https://doi.org/10.5070/G31710273.

Bieluczyk, W., Merloti, L.F., Cherubin, M.R., Mendes, L.W., Bendassolli, J.A., Rodrigues, R.R., Camargo, P.B. de, van der Putten, W.H., Tsai, S.M., 2023. Forest restoration rehabilitates soil multifunctionality in riparian zones of sugarcane production landscapes. Sci. Total Environ. 888, 164175 https://doi.org/10.1016/j.scitotenv.2023.164175.

Cambou, A., Chevallier, T., Barthès, B.G., Derrien, D., Cannavo, P., Bouchard, A., Allory, V., Schwartz, C., Vidal-Beaudet, L., 2023. The impact of urbanization on soil organic carbon stocks and particle size and density fractions. J. Soils Sediment. 23, 792–803. https://doi.org/10.1007/s11368-022-03352-3.

Canedoli, C., Ferrè, C., El Khair, D.A., Padoa-Schioppa, E., Comolli, R., 2020. Soil organic carbon stock in different urban land uses: high stock evidence in urban parks. Urban Ecosyst. 23, 159–171. https://doi.org/10.1007/s11252-019-00901-6.

Cortez-Silva, E.E., Santos, G. de S., Leite, M.G.P., Eskinazi-Sant'Anna, E.M., 2020. Response of cladoceran assemblages to restoration of riparian vegetation: A case study in a tropical reservoir of Brazil. Limnologica 85, 125822. https://doi.org/10.1016/j.limno.2020.125822.

Cutcu, I., Beyaz, A., Gerlikhan, S.G., Kilic, Y., 2023. Is ecological footprint related to foreign trade? Evidence from the top ten fastest developing countries in the global economy. J. Clean. Prod. 413, 137517 https://doi.org/10.1016/j.jclepro.2023.137517.

da Silva, C.F.A., de Andrade, M.O., dos Santos, A.M., de Melo, S.N., 2023. Road network and deforestation of indigenous lands in the Brazilian Amazon. Transp. Res. Part D. Transp. Environ. 119, 103735 https://doi.org/10.1016/j.trd.2023.103735.

Dai, J., Ouyang, Y., Hou, J., Cai, L., 2023. Long-time series assessment of the sustainable development of Xiamen City in China based on ecological footprint calculations. Ecol. Indic. 148, 110130 https://doi.org/10.1016/j.ecolind.2023.110130.

das Neves, P.B.T., Blanco, C.J.C., Montenegro Duarte, A.A.A., das Neves, F.B.S., das Neves, I.B.S., de Paula dos Santos, M.H., 2021. Amazon rainforest deforestation influenced by clandestine and regular roadway network. Land Use Policy 108, 105510. https://doi.org/10.1016/j.landusepol.2021.105510.

de Arce, R., Mahía, R., 2023. Unbiasing the estimate of the role of income in carbon footprint of households: Analysis of the Spanish case as a pilot study. Heliyon 9, e16394. https://doi.org/10.1016/j.heliyon.2023.e16394.

de Oliveira, A.R., Pacheco, F.A.L., Fernandes, L.F.S., da Costa, A.M., 2023. Trends of potentially illegal deforestation over protected areas of Rio das Velhas basin, Brazil. L. Degrad. Dev. https://doi.org/10.1002/ldr.4799.

de Souza Medeiros, A., dos Santos, T.C., Maia, S.M.F., 2022. Effect of long-term and soil depth on soil organic carbon stocks after conversion from native vegetation to conventional tillage systems in Brazil. Soil Tillage Res. 219, 105336 https://doi.org/10.1016/j.still.2022.105336.

Dembińska, I., Kauf, S., Tłuczak, A., Szopik-Depczyńska, K., Marzantowicz, Ł., Ioppolo, G., 2022. The impact of space development structure on the level of ecological footprint - Shift share analysis for European Union countries. Sci. Total Environ. 851, 157936 https://doi.org/10.1016/j.scitotenv.2022.157936.

Deng, P., Yu, X., Zhou, Y., 2023. Cradle-to-grave lifecycle carbon footprint analysis and frontier decarbonization pathways of district buildings in subtropical Guangzhou, China. J. Clean. Prod., 137921 https://doi.org/10.1016/j.jclepro.2023.137921.

Deng, X., Cao, Q., Wang, L., Wang, W., Wang, S., Wang, S., Wang, L., 2023. Characterizing urban densification and quantifying its effects on urban thermal environments and human thermal comfort. Landsc. Urban Plan. 237, 104803 https://doi.org/10.1016/j.landurbplan.2023.104803.

Digafe, B.M., Adam, A.G., Shibeshi, G.B., 2023. Ethiopian urban land allocation policy and its contribution to urban densification. Heliyon 9, e17557. https://doi.org/10.1016/j.heliyon.2023.e17557.

Dong, H., Feng, Z., Yang, Y., Li, P., You, Z., 2021. Sustainability assessment of critical natural capital: a case study of water resources in Qinghai Province, China. J. Clean. Prod. 286, 125532 https://doi.org/10.1016/j.jclepro.2020.125532.

Du, Y.-W., Wang, Y.-C., Li, W.-S., 2022. Emergy ecological footprint method considering uncertainty and its application in evaluating marine ranching resources and environmental carrying capacity. J. Clean. Prod. 336, 130363 https://doi.org/10.1016/j.jclepro.2022.130363.

Faria, D., Morante-Filho, J.C., Baumgarten, J., Bovendorp, R.S., Cazetta, E., Gaiotto, F.A., Mariano-Neto, E., Mielke, M.S., Pessoa, M.S., Rocha-Santos, L., Santos, A.S., Soares, L.A.S.S., Talora, D.C., Vieira, E.M., Benchimol, M., 2023. The breakdown of ecosystem functionality driven by deforestation in a global biodiversity hotspot. Biol. Conserv. 283, 110126 https://doi.org/10.1016/j.biocon.2023.110126.

Fartout Enayat, F., Ghanbari, S.A., Asgharipour, M.R., Seyedabadi, E., 2023. Emergy ecological footprint analysis of Yaghooti grape production in the Sistan region of Iran. Ecol. Modell. 481, 110332 https://doi.org/10.1016/j.ecolmodel.2023.110332.

Fockink, G.D., Zangalli, C., de Oliveira, E., Luz, M., de, S., de Goes, M.P., da Silva, A.C., Floriani, M.M.P., Nicoletti, M.F., Kanieski, M.R., 2022. Ecological indicators of passive restoration in South Brazil's Atlantic Forest areas with former Pinus taeda L. plantations. Ecol. Eng. 179, 106604 https://doi.org/10.1016/j.ecoleng.2022.106604.

Galán-Acedo, C., Spaan, D., Bicca-Marques, J.C., de Azevedo, R.B., Villalobos, F., Rosete-Vergés, F., 2021. Regional deforestation drives the impact of forest cover and matrix quality on primate species richness. Biol. Conserv. 263, 109338 https://doi.org/10.1016/j.biocon.2021.109338.

Galinato, G.I., Galinato, S.P., 2016. The effects of government spending on deforestation due to agricultural land expansion and CO2 related emissions. Ecol. Econ. 122, 43–53. https://doi.org/10.1016/j.ecolecon.2015.10.025.

Gao, J., O'Neill, B.C., 2020. Mapping global urban land for the 21st century with data-driven simulations and Shared Socioeconomic Pathways. Nat. Commun. 11, 2302. https://doi.org/10.1038/s41467-020-15788-7.

Gerber, J.-D., Debrunner, G., 2022. Planning with power. Implementing urban densification policies in Zurich, Switzerland. Land Use Policy 123, 106400. https://doi.org/10.1016/j.landusepol.2022.106400.

Green, J.K., Seneviratne, S.I., Berg, A.M., Findell, K.L., Hagemann, S., Lawrence, D.M., Gentine, P., 2019. Large influence of soil moisture on long-term terrestrial carbon uptake. Nature 565, 476–479. https://doi.org/10.1038/s41586-018-0848-x.

Grüneberg, E., Schöning, I., Hessenmöller, D., Schulze, E.-D., Weisser, W.W., 2013. Organic layer and clay content control soil organic carbon stocks in density fractions of differently managed German beech forests. . Ecol. Manag. 303, 1–10. https://doi.org/10.1016/j.foreco.2013.03.014.

Guo, J., Wei, Z., Xie, X., Ren, J., Zhou, H., 2022. Dynamic change and driving force of natural capital in Qinghai Province based on the three-dimensional ecological footprint, China. Ecol. Indic. 145, 109673 https://doi.org/10.1016/j.ecolind.2022.109673.

Herdt, T., Jonkman, A.R., 2023. The acceptance of density: Conflicts of public and private interests in public debate on urban densification. Cities 140, 104451. https://doi.org/10.1016/j.cities.2023.104451.

Hoekstra, A.Y., Buurman, J., van Ginkel, K.C.H., 2018. Urban water security: A review. Environ. Res. Lett. 13, 053002 https://doi.org/10.1088/1748-9326/aaba52.

Jaibumrung, K., Nilsalab, P., Gheewala, S.H., Musikavong, C., 2023. Ecological footprint, water scarcity footprint, and benefit to cost ratio analysis towards sustainable rice production in Thailand. Sustain. Prod. Consum. 39, 79–92. https://doi.org/10.1016/j.spc.2023.04.019.

Jantalia, C.P., Resck, D.V.S., Alves, B.J.R., Zotarelli, L., Urquiaga, S., Boddey, R.M., 2007. Tillage effect on C stocks of a clayey Oxisol under a soybean-based crop rotation in the Brazilian Cerrado region. Soil Tillage Res. 95, 97–109. https://doi.org/10.1016/j.still.2006.11.005.

Kasmaeeyazdi, S., Abdolmaleki, M., Ibrahim, E., Jiang, J., Marzan, I., Rodríguez, I.B., 2021. Copernicus data to boost raw material source management: Illustrations from the RawMatCop programme. Resour. Policy 74, 102384. https://doi.org/10.1016/j.resourpol.2021.102384.

Kassouri, Y., 2021. Monitoring the spatial spillover effects of urbanization on water, built-up land and ecological footprints in sub-Saharan Africa. J. Environ. Manag. 300, 113690 https://doi.org/10.1016/j.jenvman.2021.113690.

Kaur, M., Hewage, K., Sadiq, R., 2022. Investigating temporal dynamics of urban densification on the buried water infrastructure performance. Cities 129, 103836. https://doi.org/10.1016/j.cities.2022.103836.

Kaur, M., Hewage, K., Sadiq, R., 2020. Investigating the impacts of urban densification on buried water infrastructure through DPSIR framework. J. Clean. Prod. 259, 120897 https://doi.org/10.1016/j.jclepro.2020.120897.

Kazemzadeh, E., Fuinhas, J.A., Salehnia, N., Koengkan, M., Silva, N., 2023. Assessing influential factors for ecological footprints: A complex solution approach. J. Clean. Prod. 414, 137574 https://doi.org/10.1016/j.jclepro.2023.137574.

Kirikkaleli, D., Sofuoğlu, E., Ojekemi, O., 2023. Does patents on environmental technologies matter for the ecological footprint in the USA? Evidence from the novel Fourier ARDL approach. Geosci. Front. 14, 101564 https://doi.org/10.1016/j.gsf.2023.101564.

Kong, X., Ghaffar, S., Determann, M., Friese, K., Jomaa, S., Mi, C., Shatwell, T., Rinke, K., Rode, M., 2022. Reservoir water quality deterioration due to deforestation emphasizes the indirect effects of global change. Water Res. 221, 118721 https://doi.org/10.1016/j.watres.2022.118721.

Kou, J., Li, C., Ma, W., 2023. The coupling of agricultural water footprint and socioeconomic development in ecological functional zones: A case study of Gansu Province, China. Anthropocene 43, 100391. https://doi.org/10.1016/j.ancene.2023.100391.

Li, M., Verburg, P.H., van Vliet, J., 2022. Global trends and local variations in land take per person. Landsc. Urban Plan. 218, 104308 https://doi.org/10.1016/j.landurbplan.2021.104308.

Li, P., Zhang, R., Wei, H., Xu, L., 2022. Assessment of physical quantity and value of natural capital in China since the 21st century based on a modified ecological footprint model. Sci. Total Environ. 806, 150676 https://doi.org/10.1016/j.scitotenv.2021.150676.

Li, P., Zhang, R., Xu, L., 2021. Three-dimensional ecological footprint based on ecosystem service value and their drivers: A case study of Urumqi. Ecol. Indic. 131, 108117 https://doi.org/10.1016/j.ecolind.2021.108117.

Li, Z., Murshed, M., Yan, P., 2023. Driving force analysis and prediction of ecological footprint in urban agglomeration based on extended STIRPAT model and shared socioeconomic pathways (SSPs). J. Clean. Prod. 383, 135424 https://doi.org/10.1016/j.jclepro.2022.135424.

Lin, W., Li, Y., Li, X., Xu, D., 2018. The dynamic analysis and evaluation on tourist ecological footprint of city: Take Shanghai as an instance. Sustain. Cities Soc. 37, 541–549. https://doi.org/10.1016/j.scs.2017.12.003.

Liu, L., Xu, L., Wang, S., Tian, X., 2023. The transition patterns of rural household carbon footprint in China. Resour. Environ. Sustain. 13, 100122 https://doi.org/10.1016/j.resenv.2023.100122.

Liu, Y., Qu, Y., Cang, Y., Ding, X., 2022. Ecological security assessment for megacities in the Yangtze River basin: Applying improved emergy-ecological footprint and DEA-SBM model. Ecol. Indic. 134, 108481 https://doi.org/10.1016/j.ecolind.2021.108481.

Liu, Z., Tian, J., Wang, K., Lan, J., 2023. The impact of farmland circulation on the carbon footprint of agricultural cultivation in China. Econ. Anal. Policy 78, 792–801. https://doi.org/10.1016/j.eap.2023.04.010.

Locatelli, J.L., Santos, R.S., Cherubin, M.R., Cerri, C.E.P., 2022. Changes in soil organic matter fractions induced by cropland and pasture expansion in Brazil's new agricultural frontier. Geoderma Reg. 28, e00474 https://doi.org/10.1016/j.geodrs.2021.e00474.

Lwasa, S., Mugagga, F., Wahab, B., Simon, D., Connors, J., Griffith, C., 2014. Urban and peri-urban agriculture and forestry: Transcending poverty alleviation to climate change mitigation and adaptation. Urban Clim. 7, 92–106. https://doi.org/10.1016/j.uclim.2013.10.007.

Ma, B., Bashir, M.F., Peng, X., Strielkowski, W., Kirikkaleli, D., 2023. Analyzing research trends of universities' carbon footprint: An integrated review. Gondwana Res. 121, 259–275. https://doi.org/10.1016/j.gr.2023.05.008.

Mahmoud, W.H., Elagib, N.A., Gaese, H., Heinrich, J., 2014. Rainfall conditions and rainwater harvesting potential in the urban area of Khartoum. Resour. Conserv. Recycl. 91, 89–99. https://doi.org/10.1016/j.resconrec.2014.07.014.

Maing, M., 2022. Superblock transformation in Seoul Megacity: Effects of block densification on urban ventilation patterns. Landsc. Urban Plan. 222, 104401 https://doi.org/10.1016/j.landurbplan.2022.104401.

Mancini, M.S., Barioni, D., Danelutti, C., Barnias, A., Bračanov, V., Capanna Piscè, G., Chappaz, G., Đuković, B., Guarneri, D., Lang, M., Martín, I., Matamoros Reverté, S., Morell, I., Peçulaj, A., Prvan, M., Randone, M., Sampson, J., Santarossa, L., Santini, F., Selmani, J., Ser, C., Sinibaldi, I., Topi, M., Treglia, V., Zirletta, S., Galli, A., 2022. Ecological Footprint and tourism: Development and sustainability monitoring of ecotourism packages in Mediterranean Protected Areas. J. Outdoor Recreat. Tour. 38, 100513 https://doi.org/10.1016/j.jort.2022.100513.

Marchi, M., Marchettini, N., Neri, E., Esposito, G., Niccolucci, V., Pulselli, F.M., Lajo, M., Rissone, E., Pulselli, R.M., 2023. Carbon Footprint offset of a managed Bamboo plantation in temperate regions. Sustain. Prod. Consum. https://doi.org/10.1016/j.spc.2023.05.025.

Marquardt, S.G., Doelman, J.C., Daioglou, V., Tabeau, A., Schipper, A.M., Sim, S., Kulak, M., Steinmann, Z.J.N., Stehfest, E., Wilting, H.C., Huijbregts, M.A.J., 2021. Identifying regional drivers of future land-based biodiversity footprints. Glob. Environ. Chang. 69, 102304 https://doi.org/10.1016/j.gloenvcha.2021.102304.

Masiliūnas, D., Tsendbazar, N.-E., Herold, M., Lesiv, M., Buchhorn, M., Verbesselt, J., 2021. Global land characterisation using land cover fractions at 100 m resolution. Remote Sens. Environ. 259, 112409 https://doi.org/10.1016/j.rse.2021.112409.

McDonald, R.I., Weber, K., Padowski, J., Flörke, M., Schneider, C., Green, P.A., Gleeson, T., Eckman, S., Lehner, B., Balk, D., Boucher, T., Grill, G., Montgomery, M., 2014. Water on an urban planet: Urbanization and the reach of urban water infrastructure. Glob. Environ. Chang. 27, 96–105. https://doi.org/10.1016/j.gloenvcha.2014.04.022.

Mitoma, H., 2023. Carbon footprint analysis considering production activities of informal sector: The case of manufacturing industries of India. Energy Econ., 106809 https://doi.org/10.1016/j.eneco.2023.106809.

Mohajeri, N., Walch, A., Smith, A., Gudmundsson, A., Assouline, D., Russell, T., Hall, J., 2023. A machine learning methodology to quantify the potential of urban densification in the Oxford-Cambridge Arc, United Kingdom. Sustain. Cities Soc. 92, 104451 https://doi.org/10.1016/j.scs.2023.104451.

Mohebalian, P.M., Lopez, L.N., Tischner, A.B., Aguilar, F.X., 2022. Deforestation in South America's tri-national Paraná Atlantic Forest: Trends and associational factors. . Policy Econ. 137, 102697 https://doi.org/10.1016/j.forpol.2022.102697.

Mota, P.K., da Costa, A.M., Prado, R.B., Fernandes, L.F.S., Pissarra, T.C.T., Pacheco, F.A.L., 2023. Payment for Environmental Services: A critical review of schemes, concepts, and practice in Brazil. Sci. Total Environ. 899, 165639 https://doi.org/10.1016/j.scitotenv.2023.165639.

Mullan, K., Caviglia-Harris, J.L., Sills, E.O., 2021. Sustainability of agricultural production following deforestation in the tropics: Evidence on the value of newly-deforested, long-deforested and forested land in the Brazilian Amazon. Land Use Policy 108, 105660. https://doi.org/10.1016/j.landusepol.2021.105660.

Norman, E.S., Dunn, G., Bakker, K., Allen, D.M., Cavalcanti de Albuquerque, R., 2013. Water Security Assessment: Integrating Governance and Freshwater Indicators. Water Resour. Manag. 27, 535–551. https://doi.org/10.1007/s11269-012-0200-4.

Osman, H.E.M., Elaidarous, A.A., El-Morsy, M.H., Eid, E.M., Keshta, A.E., 2023. Soils with more clay and dense vegetation were rich in soil carbon along Wadi Al-Sharaea, Makkah, Saudi Arabia. Heliyon 9, e12988. https://doi.org/10.1016/j.heliyon.2023.e12988.

Padowski, J.C., Carrera, L., Jawitz, J.W., 2016. Overcoming Urban Water Insecurity with Infrastructure and Institutions. Water Resour. Manag. 30, 4913–4926. https://doi.org/10.1007/s11269-016-1461-0.

Pan, X., Guo, S., 2023. Dynamic decomposition and regional differences of urban emergy ecological footprint in the Yangtze River Delta. J. Environ. Manag. 326, 116698 https://doi.org/10.1016/j.jenvman.2022.116698.

Parras, R., de Mendonça, G.C., Costa, R.C.A., Pissarra, T.C.T., Valera, C.A., Fernandes, L.F.S., Pacheco, F.A.L., 2020. The configuration of forest cover in Ribeirão Preto: A diagnosis of Brazil's forest code implementation. Sustain. https://doi.org/10.3390/su12145686.

Peng, Y., Rieke, E.L., Chahal, I., Norris, C.E., Janovicek, K., Mitchell, J.P., Roozeboom, K.L., Hayden, Z.D., Strock, J.S., Machado, S., Sykes, V.R., Deen, B., Tavarez, O.B., Gamble, A.V., Scow, K.M., Brainard, D.C., Millar, N., Johnson, G.A., Schindelbeck, R.R., Kurtz, K.S.M., van Es, H., Kumar, S., Van Eerd, L.L., 2023. Maximizing soil organic carbon stocks under cover cropping: insights from long-term agricultural experiments in North America. Agric. Ecosyst. Environ. 356, 108599 https://doi.org/10.1016/j.agee.2023.108599.

Phumalee, U., Phongkhieo, N.T., Emphandhu, D., Bejranonda, S., 2018. Touristic ecological footprint in Mu Ko Surin National Park. Kasetsart J. Soc. Sci. 39, 1–8. https://doi.org/10.1016/j.kjss.2018.01.003.

Pissarra, T.C.T., Valera, C.A., Costa, R.C.A., Siqueira, H.E., Martins Filho, M.V., Valle Júnior, R.F.do, Sanches Fernandes, L.F., Pacheco, F.A.L., 2019. A Regression Model of Stream Water Quality Based on Interactions between Landscape Composition and Riparian Buffer Width in Small Catchments. Water 11, 1757. https://doi.org/10.3390/w11091757.

Quezada, J.C., Guillaume, T., Poeplau, C., Ghazoul, J., Buttler, A., 2022. Deforestation-free land-use change and organic matter-centered management improve the C footprint of oil palm expansion. Glob. Chang. Biol. 28, 2476–2490. https://doi.org/10.1111/gcb.16069.

Ribeiro, M.P., de Mello, K., Valente, R.A., 2022. How can forest fragments support protected areas connectivity in an urban landscape in Brazil? Urban For. Urban Green. 74, 127683 https://doi.org/10.1016/j.ufug.2022.127683.

Richards, R.C., Petrie, R., Christ, B., Ditt, E., Kennedy, C.J., 2020. Farmer preferences for reforestation contracts in Brazil's Atlantic Forest. For. Policy Econ. 118, 102235 https://doi.org/10.1016/j.forpol.2020.102235.

Righi, S., Viganò, E., Panzone, L., 2023. Consumer concerns over food insecurity drive reduction in the carbon footprint of food consumption. Sustain. Prod. Consum. 39, 451–465. https://doi.org/10.1016/j.spc.2023.05.027.

Rodríguez-Benito, C.V., Navarro, G., Caballero, I., 2020. Using Copernicus Sentinel-2 and Sentinel-3 data to monitor harmful algal blooms in Southern Chile during the COVID-19 lockdown. Mar. Pollut. Bull. 161, 111722 https://doi.org/10.1016/j.marpolbul.2020.111722.

Rufí-Salís, M., Brunnhofer, N., Petit-Boix, A., Gabarrell, X., Guisasola, A., Villalba, G., 2020. Can wastewater feed cities? Determining the feasibility and environmental burdens of struvite recovery and reuse for urban regions. Sci. Total Environ. 737, 139783 https://doi.org/10.1016/j.scitotenv.2020.139783.

Sagar, U.S., Singh, Y., Mahalingam, A., Malladi, T., 2022. Future impacts of Urban and Peri-urban agriculture on carbon stock and land surface temperatures in India. Urban Clim. 45, 101267 https://doi.org/10.1016/j.uclim.2022.101267.

Samuele, D.P., Federica, G., Filippo, S., Enrico, B.-M., 2022. A simplified method for water depth mapping over crops during flood based on Copernicus and DTM open data. Agric. Water Manag. 269, 107642 https://doi.org/10.1016/j.agwat.2022.107642.

Santiago, A.R., Couto, H.T.Z. do, 2020. Socioeconomic development versus deforestation: considerations on the sustainability of economic and social growth in most Brazilian municipalities. Environ. Dev. 35, 100520 https://doi.org/10.1016/j.envdev.2020.100520.

Santos, A.M. dos, Silva, C.F.A. da, Almeida Junior, P.M. de, Rudke, A.P., Melo, S.N. de, 2021. Deforestation drivers in the Brazilian Amazon: assessing new spatial predictors. J. Environ. Manag. 294, 113020 https://doi.org/10.1016/j.jenvman.2021.113020.

Santos, Y.L.F., Yanai, A.M., Ramos, C.J.P., Graça, P.M.L.A., Veiga, J.A.P., Correia, F.W.S., Fearnside, P.M., 2022. Amazon deforestation and urban expansion: Simulating future growth in the Manaus Metropolitan Region, Brazil. J. Environ. Manag. 304, 114279 https://doi.org/10.1016/j.jenvman.2021.114279.

Silva, J.G. da, Almeida, R.B. de, Carvalho, L.V., 2023. An economic analysis of a zero-deforestation policy in the Brazilian Amazon. Ecol. Econ. 203, 107613 https://doi.org/10.1016/j.ecolecon.2022.107613.

Taramelli, Lissoni, Piedelobo, Schiavon, Valentini, Xuan, González-Aguilera, 2019. Monitoring Green Infrastructure for Natural Water Retention Using Copernicus Global Land Products. Remote Sens. 11, 1583. https://doi.org/10.3390/rs11131583.

Trevisan, A.C.D., Schmitt-Filho, A.L., Farley, J., Fantini, A.C., Longo, C., 2016. Farmer perceptions, policy and reforestation in Santa Catarina, Brazil. Ecol. Econ. 130, 53–63. https://doi.org/10.1016/j.ecolecon.2016.06.024.

Trigueiro, W.R., Nabout, J.C., Tessarolo, G., 2020. Uncovering the spatial variability of recent deforestation drivers in the Brazilian Cerrado. J. Environ. Manag. 275, 111243 https://doi.org/10.1016/j.jenvman.2020.111243.

Valera, C., Pissarra, T., Filho, M., Valle Júnior, R., Oliveira, C., Moura, J., Sanches Fernandes, L., Pacheco, F., 2019. The Buffer Capacity of Riparian Vegetation to Control Water Quality in Anthropogenic Catchments from a Legally Protected Area: A Critical View over the Brazilian New Forest Code. Water 11, 549. https://doi.org/10.3390/w11030549.

Valera, C.A., Pissarra, T.C.T., Martins Filho, M.V., Valle Junior, R.F., Sanches Fernandes, L.F., Pacheco, F.A.L., 2017. A legal framework with scientific basis for applying the 'polluter pays principle' to soil conservation in rural watersheds in Brazil. Land Use Policy 66, 61–71. https://doi.org/10.1016/j.landusepol.2017.04.036.

Valera, C.A., Valle Junior, R.F., Varandas, S.G.P., Sanches Fernandes, L.F., Pacheco, F.A.L., 2016. The role of environmental land use conflicts in soil fertility: A study on the Uberaba River basin, Brazil. Sci. Total Environ. 562, 463–473. https://doi.org/10.1016/j.scitotenv.2016.04.046.

Vasenev, V.I., Stoorvogel, J.J., Leemans, R., Valentini, R., Hajiaghayeva, R.A., 2018. Projection of urban expansion and related changes in soil carbon stocks in the Moscow Region. J. Clean. Prod. 170, 902–914. https://doi.org/10.1016/j.jclepro.2017.09.161.

Vasques, G.M., Coelho, M.R., Dart, R.O., Baca, J.F.M., De, M., 2021. Soil Organic Carbon Stock at 0–30 cm Map for Brazil with 1 km Spatial Resolution. Technical Report, Embrapa Solos, Rio de Janeiro, Brazil, 3pg.

Vieira, D.C., Sanches, I.D., Montibeller, B., Prudente, V.H.R., Hansen, M.C., Baggett, A., Adami, M., 2022. Cropland expansion, intensification, and reduction in Mato Grosso state, Brazil, between the crop years 2000/01 to 2017/18. Remote Sens. Appl. Soc. Environ. 28, 100841 https://doi.org/10.1016/j.rsase.2022.100841.

Wackernagel, M., 1998. The ecological footprint of Santiago de Chile. Local Environ. 3, 7–25. https://doi.org/10.1080/13549839808725541.

Xiao, S., Fügener, T., Wende, W., Yan, W., Chen, H., Syrbe, R., Xue, B., 2022. The dynamics of vegetation and implications for ecosystem services in the context of urbanisation: An example from Huangyan-Taizhou, China. Ecol. Eng. 179, 106614 https://doi.org/10.1016/j.ecoleng.2022.106614.

Xie, B., Zhang, X., Lu, J., Liu, F., Fan, Y., 2022. Research on ecological evaluation of Shanghai port logistics based on emergy ecological footprint models. Ecol. Indic. 139, 108916 https://doi.org/10.1016/j.ecolind.2022.108916.

Xin, M., Guo, H., Li, S., Chen, L., 2023. Can China achieve ecological sustainability? An LMDI analysis of ecological footprint and economic development decoupling. Ecol. Indic. 151, 110313 https://doi.org/10.1016/j.ecolind.2023.110313.

Xu, L., Herold, M., Tsendbazar, N.-E., Masiliūnas, D., Li, L., Lesiv, M., Fritz, S., Verbesselt, J., 2022. Time series analysis for global land cover change monitoring: A comparison across sensors. Remote Sens. Environ. 271, 112905 https://doi.org/10.1016/j.rse.2022.112905.

Yan, Y., Kuang, W., Zhang, C., Chen, C., 2015. Impacts of impervious surface expansion on soil organic carbon – a spatially explicit study. Sci. Rep. 5, 17905 https://doi.org/10.1038/srep17905.

Yao, X., Chen, W., Song, C., Gao, S., 2022. Sustainability and efficiency of water-land-energy-food nexus based on emergy-ecological footprint and data envelopment analysis: Case of an important agriculture and ecological region in Northeast China. J. Clean. Prod. 379, 134854 https://doi.org/10.1016/j.jclepro.2022.134854.

Yue, Z., Liu, H., Xu, Z., Wang, Y., 2023. Evaluation of sustainability in northern Xinjiang based on ecological footprint-planetary boundary system framework. Ecol. Indic. 150, 110270 https://doi.org/10.1016/j.ecolind.2023.110270.

Zadehdabagh, N., Monavari, S.M., Kargari, N., Taghavi, L., Pirasteh, S., 2022. Sustainability of agroecosystems by indices: A comparative study between indicators of ecological footprint sustainability and emergy analysis; a case study in Dez catchment, Iran. Ecol. Modell. 474, 110165 https://doi.org/10.1016/j.ecolmodel.2022.110165.

Zangalli, C., Fockink, G.D., de Almeida, B.R.S., de Oliveira, E., de Goes, M.P., Nicoletti, M.F., Floriani, M.M.P., Kanieski, M.R., 2023. Ecological indicators of forest regeneration in areas of Araucaria Forest in South Brazil. Ecol. Eng. 194, 107035 https://doi.org/10.1016/j.ecoleng.2023.107035.

Zhang, M., Ao, Y., Liu, M., Zhao, Y., Lin, K., Cheng, T., 2022. Ecological security assessment based on ecosystem service value and ecological footprint in the Pearl River Delta urban agglomeration, China. Ecol. Indic. 144, 109528 https://doi.org/10.1016/j.ecolind.2022.109528.