Research Article
doi: 10.1590/01047760202329013193
vol(29), 2023

CERNE

Although it lacks connectivity, isolated urban forest fragments can deliver similar amounts of ecosystem services as in protected areas

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FOREST ECOLOGY

ABSTRACT

Backgrounds: Urban forests can provide citizens with various types and quantities of ecosystem services. However, the contribution of isolated forest fragments to this process and its value are not well understood. Therefore, our main goal was to analyze the patterns of ecosystem service delivery by three forest fragments along an urban-to-rural gradient. Carbon storage was estimated using three different models (general and local), with input variables such as diameter at breast height (DBH), total height, and wood density. Rainfall interception was assessed using water collectors placed inside the forest fragment, at the border, and outside, throughout the four seasons. Data were collected from three forest fragments of different types (urban, periurban, and rural), each covering an area of 1.0 hectare.

Results: Isolated forest fragments did not significantly differ ($p>0.01$) in terms of carbon stocks compared to a protected area. The urban fragment had a carbon stock of 33.70 $mg.ha^{-1}$, while the rural fragment had a carbon stock of 37.19 $mg.ha^{-1}$. Regarding rainfall interception, there were no significant differences ($p>0.01$) among the forest fragments. The highest average rainfall interception percentage (58.65%) was observed during the summer. The capacity for rainfall interception changed from the border (lower) to the center of the fragment (higher) in a similar manner for both isolated and connected forest fragments.

Conclusion: Isolated Forest fragments can provide ecosystem services (carbon storage and rainfall interception) in the same way as connected forest fragments in the periurban and rural places, like in protected areas. It highlights the importance of promoting the protection of forests fragments in cities.

Keywords: carbon storage; green areas; rainfall interception; urban ecology

HIGHLIGHTS:

We compared the quality of forest fragments in an urban-to-rural gradient.
Urban forest fragment delivers similar amount of ecosystem services as a protected one.
Rainfall interception values were not different, although higher for the urban fragment
Carbon storage was lower in the urban fragment, but not different from the others.

AGUIAR, J. T.; BRUN, F. G. K.; PEDRO HIGUCHI, P.; BOBROWSKI, R. Although it lacks connectivity, isolated urban forest fragments can deliver similar amounts of ecosystem services as in protected areas. CERNE, v.29, e-103193, doi: 10.1590/01047760202329013193

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Received: November, 11/2022 Accepted: June, 14/2023

CERNE
ISSN 0104-7760
UNIVERSIDADE FEDERAL DE LAVRAS

INTRODUCTION

The urbanization of rural or natural areas has led to significant changes in ecosystems around the world, which has resulted in the loss of natural habitats, forest fragmentation, forest and biodiversity degradation and homogenization (Newbold et al. 2015; Hodges; Mckinney, 2018). The effects of urbanization on native flora and fauna are complex because it can simultaneously lead to high local extinction rates, but also create refuges for rare or threatened species (Madre et al. 2014; Ives et al. 2016).

The impact of fragmentation on forest species comes from the forest area reduction and the fragment isolation from the other remaining fragments (Fahrig, 2020). Both problems can affect forest dynamics. Generally, the closer the fragments are to each other, the easier the movement of species between them, which can reduce the risk of local extinction and, consequently, the loss of ecosystem services (Estavillo; Pardini; Da Rocha, 2013; Hillebrand et al., 2017). On the other hand, the conservation of urban biodiversity is essential for the provisioning of ecosystem services to the urban population, including food supply, microclimate regulation, biomass production, water supply and purification, pollination, and habitat for forest species (Luederitz et al. 2015; Decocq et al. 2016; Liang et al. 2016; Mori et al. 2017).

In this sense, different studies have shown the contribution of forest fragments to the rainfall interception (Freitas et al., 2016; Liu et al., 2018; Groppo et al., 2019) and carbon storage (Cunha et al., 2009; Ferraz et al., 2014; Azevedo et al., 2018). In addition, other studies reported the contribution of different types of urban green infrastructures (street trees, parks, private gardens) to the promotion of ecosystem services (Derkzen et al., 2015; Strohbach; Haase, 2012; Sutton; Anderson, 2016). What is remarkable from these studies is that they highlight the importance of vegetation for the provision of ecosystem services. However, nothing has yet been reported on the contribution and limitations of the isolated urban forest fragments to the provision of ecosystem services.

In fact, isolated forest fragments in urban areas are sensitive to microclimatic and anthropogenic effects, supporting only a small proportion of the original forest biodiversity composition, so their value for conservation is often disregarded as it alters the dispersal processes of plants, mainly for the ones that rely on zoocorical dispersal (Pfeifer et al., 2017; Gelmi-Candusso; Hämäläinen, 2019). However, the importance of the heterogeneity of this habitat for the conservation of biodiversity is increasingly recognized (Arroyo-Rodriguez et al., 2020; Watling; Fang, 2020).

Considering the general assumption that urban forest fragments lack quality and species diversity, we hypothesized that this kind of urban green infrastructure cannot provide ecosystem services in the same amount as forests in protected areas or more connected ones in the landscape. Although all the problems and importance of forests to the urban population the main goal of this research was to verify the distinctions in the supply of ecosystem services of different forest fragments along an urban-to-rural gradient, due to different characteristics of connectivity and isolation.

MATERIAL AND METHODS

Study area

This exploratory and experimental-based study was conducted in three different areas along the urban-to-rural gradient. Two of the study areas were situated in the urban and periurban regions of Irati, Paraná State, Brazil, while the third area was located in the countryside of Fernandes Pinheiro, within the Irati National Forest (Figure 1). These study sites are approximately 150 km away from the capital city, Curitiba, in the south-central region of Paraná State. They are situated within the Mixed Ombrophilous Forest (MOF) ecosystem. The local climate is classified as Cfb according to the Köppen-Geiger classification, characterized by mild summers and winters with frequent and severe frosts, but no distinct dry season. The average annual temperature is 17.5°C, with a monthly average rainfall of 193.97 mm. The annual rainfall reaches 1,582 mm, and the average monthly relative humidity is 80.3% (Sawczuk et al., 2014; Roveda et al., 2018).

The urban forest fragment, named São Francisco Forest, is situated in the downtown area of Irati, near one of the main avenues. It occupies a slightly larger area of 1.0 ha and is surrounded by sidewalks and streets. Beyond the forest, there are various residential buildings with diverse patterns and associated gardens (Figures 1 and 2). The periurban fragment is located within the campus of Midwestern State University (UNICENTRO) in Irati. It represents an urban-rural transition area, covering approximately 32.0 ha of forest. For the study, we selected a forest sample of 1.0 ha located at the forest edge adjacent to the campus buildings to replicate the uncovered-to-covered land pattern observed in the urban fragment (Figures 1 and 2). The rural fragment encompasses a total area of 3,495 ha and is situated within the Irati National Forest (FLONA). FLONA is a Conservation Unit classified as Sustainable Use and is in an advanced stage of secondary succession (Sawczuk et al., 2014). In 2002, 25 permanent sample plots were established in one section of FLONA, each measuring 1.0 hectare. For this study, data from block 17 were utilized as it is located at the forest edge adjacent to an agricultural area, resembling the uncovered-to-covered land pattern found in the urban fragment (Figures 1 and 2).

Estimation of the above-ground biomass and carbon storage

To ensure comparability between the forest fragments, the study areas were established with a total area of 1.0 ha, considering the limiting size of the urban fragment for field data acquisition. In each forest fragment, data were collected from fifty plots, each measuring 200 m^2 (10 m x 20 m). All trees with a diameter at breast height (DBH) of ≥ 10 cm (Sawczuk et al., 2014) were measured for their perimeter at breast height (PBH) using a measuring tape, and their total height using a digital clinometer Vertex. When it was not possible to measure the total height directly due to crown density, it was estimated based on a 3.0-meter-high vertical reference. Small branches and leaves were collected from all species for proper identification, and the species were recorded in the herbarium of the Forest Engineering Department (HUCO).

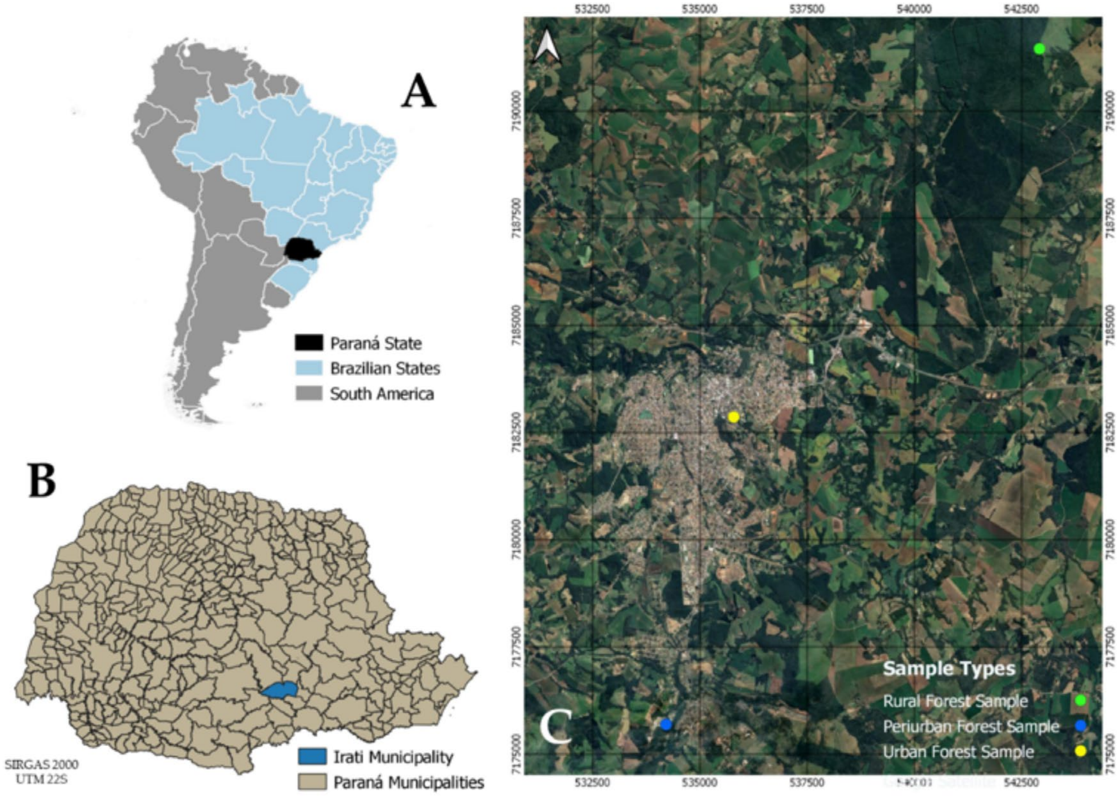

Figure 1: Localization of the forest fragments studied: Paraná State in Brazil and South America (A), the Irati Municipality in Paraná State (B) and the forest samples in the city of Irati (C).

Figure 2: Surrounding landscape in the border of forests studied (A1, A2, A3), the general canopy structure (B1, B2, B3), and the understory regeneration (C1, C2, C3), for the rural (A1 to C1), the periurban (B1 to B3), and the urban forest (C1 to C3).

The estimation of above-ground biomass and carbon storage was performed using indirect methods and models (Chart 1) proposed by Chave et al. (2014), Martins (2011), and Ratuchne (2010). To ensure repeatability within each fragment and enable comparative analysis with the tested models, data from 10 plots of 200 m^2 were combined, resulting in 5 plots of 2,000 m^2 per study area. The first model used was the pantropical allometric model proposed by Chave et al. (2014), which estimates above-ground biomass (Table 1, Equation 1). This model was developed based on a global tree database, incorporating information from 58 different locations that cover a wide range of climatic conditions and tropical vegetation types. The specific mass values used in this model were obtained from the literature (Carvalho, 1994; Lorenzi, 2000), and for species without available information, average values were calculated using available values for the same genus. From the above-ground biomass values obtained from this model, carbon storage per tree was estimated by multiplying the values by 0.5. This procedure was adopted considering that, on average, carbon content represents 50% of tree biomass in the Mixed Ombrophilous Forest (Watzlawick et al., 2014). Additional models used for comparisons were proposed by Ratuchne (2010) and Martins (2011) based on data acquired from forest fragments in the Mixed Ombrophilous Forest ecosystem, specifically the Montana type. Both authors proposed models for estimating above-ground biomass (Table 1, Equation 2 and 4) and carbon storage (Table 1, Equations 3 and 5).

To analyze data to found out if there would be difference in the above-ground biomass and carbon storage values among the forest fragments and the models applied, we adopted a Randomized Block Design, considering each forest fragment as a block and each model applied as the experimental treatment. The ANOVA was performed by verifying the residues homoscedasticity by the Bartlett test (p-value>0.05). If there were significant differences, the means were compared by Tukey's test at 5% error probability. All statistical analyzes were performed in the statistical programming language R (R Development Core Team, 2019), through the R Studio interface version 4.0.0.

Estimation of the rainfall interception and its distinctions

To understand if the three different forest fragments would provide different responses to the rainfall interception, we estimated the rainfall interception capacity from data collected in homemade rainfall collectors (Figure 3A and 3B). Those devices were created from PET bottles (same model, same size), with standardized dimensions of 22.0 cm length and 10.1 cm upper diameter opening. The collectors were fixed on bamboos at 1.0 m above soil surface (Figure 3A and 3B) in three different positions according to the forest fragment, with three repetitions each (Figure 3C), as follows: outside the forest, on the edge (~5.0m from the border limit) and in the middle of the fragment. The collectors installed outside the fragments were used as a control point, to get information on the real precipitation occurred in the place, without interference from the canopy cover of trees. On the other hand, collectors installed on the edge of the fragments were used to verify the influence of the edge effect on the rainfall interception due to the wind movement over the crowns. The collectors installed in the middle of the fragments served as a source of information to understand how much rainfall the urban, the periurban and the rural forest fragments would intercept.

Table 1: Equations from the models used to estimate the above-ground biomass (P) and the carbon storage (C), being Equation 1 from Chave et al. (2014), Equations 2 and 3 from Ratuchne (2010), and Equations 4 and 5 from Martins (2011).

Equations
$\mathbf{P} = 0.0673 * (\rho * DBH^2 * h)^{0.976}$ (1)
$\mathbf{P} = 284.499-(58.61*DBH)+(4.213*DBH^2)-(0.107*DBH^3)+(0.001*DBH^4)-(5.68^6)*dap^5$ (2)
$\mathbf{C} = 1.343 + 0.088*DBH^2 + 0.005*(DBH^2*h)$ (3)
$\mathbf{P} = 0.04821*DBH^{1.34374} * h^{1.26829}$ (4)
$\mathbf{C} = 0.01996*DBH^{1.35703} * h^{1.28396}$ (5)

Where: P – above-ground biomass (kg); C - stored carbon (kg/tree); ρ - wood specific mass (g.cm-1); DBH - diameter at breast height (1.3 m); Ht - tree total height (m).

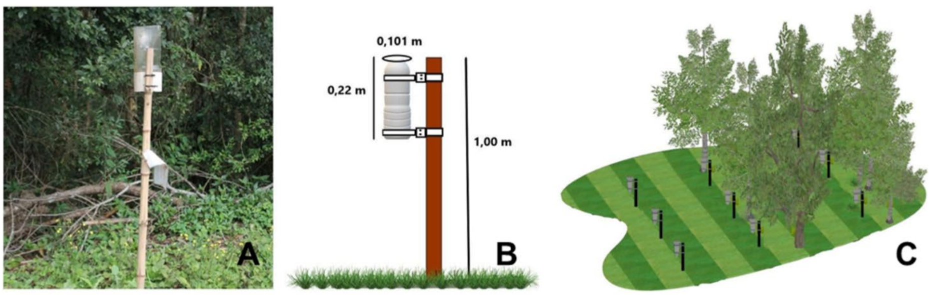

Figure 3: Image of the rain collector made and installed (A), illustrative scheme of the dimensions of the collector (B), and an illustrative scheme of the collector distribution in the outside, on the edge and in the center of the study areas (C).

Data acquisition was carried out on each rainy day for an entire month, in the period corresponding to the middle of the different seasons from 2019 to 2020, that is, between April and May (autumn), July and August (winter), October and November (spring) and January and February (summer). This seasonal evaluation is justified mainly taking into account that in Paraná State there is a wide spatial variability of precipitation evidenced by Baú et al. (2006), who evaluated monthly rainfall series in the western mesoregion of Paraná State. The amount of water collected during each day at each rainfall collector was transferred to a Glass Becker to check the amount of rain (mL). Values of each collector, in mL, were transformed in precipitation values, in mm, through the equation [6] proposed by Oliveira (2014). Where: collector opening area - πr^2, r - collector opening radius (meters)

$$\text{Preciptation (mm)} = \frac{\text{rainfall amount (liters)}}{\text{collector opening area } (m^2)} \quad (6)$$

The rainfall interception was calculated by the difference from the mean values between the precipitation outside and inside each forest fragment to explore the contribution of them to this ecosystem service. Differences among forest fragments and seasons were determined by analyzing the amount of precipitation in the center of each fragment (from rainfall collectors inside each forest fragment) through a Randomized Block Design. Seasons of the year were used as treatments and the three forest fragments as blocks to control possible local environmental variations. The Anova was performed by verifying the homoscedasticity of the residues, using the Bartlett test (p-value>0.05). If there were significant differences, the means were compared by Tukey's test at 5% error probability. All statistical analyzes were performed in the statistical programming language R (R Development Core Team, 2019), through the R Studio interface version 4.0.0.

RESULTS

No significant difference was found ($p>0.05$) among the types of forest fragments evaluated (treatments tested) for the above-ground biomass estimation nor to the carbon storage (Table 2). Otherwise, the models used to estimate those variables promoted significant differences on estimations made ($p<0.05$). The number of trees sampled, the average DBH and basal area for each forest fragment were: 510 trees, 24.65 cm and 6.46 $m^2.ha^{-1}$ (urban fragment), 521 trees, 24.75 cm and 6.37 $m^2.ha^{-1}$ (periurban fragment), and 737 trees, 22.19 cm and 7.53 $m^2.ha^{-1}$ (rural fragment).

The comparison of the pantropical model proposed by Chave et al. (2014) with the Ratuchne (2010) model for above-ground biomass estimation showed similar values (Table 3). However, when comparing the mean values of carbon storage estimations, there was a significant difference ($p<0.01$). The carbon storage values estimated by the Chave et al. (2014) model were 30.66% higher than those estimated by the Ratuchne (2010) model and 84.02% higher than the values estimated by the Martins (2011) model. It should be noted that the Martins (2011) model tended to underestimate both above-ground biomass and carbon storage estimations.

Table 2: Summary of the analysis of variance for carbon storage and above-ground biomass estimations in different forest fragments and according to different models tested, with the source of variation (SV), degree of freedom (DF) and mean square (MS).

SV	Carbon Storage			Above-ground Biomass		
	DF	MS	*p*-value	DF	MS	*p*-value
Fragments	2	6492.1	0.3893	2	1.75^{-2}	0.3029
Models	2	208.7	2.26^{-16}	2	1.93^{-4}	2.26^{-16}
Residual	40	216.1		40	1.57^{-4}	
	Bartlett test: χ^2=5.8208, DF=2, p-value=0.0545			Bartlett test: χ^2=5.7623, DF=2, p-value=0.0565		

Table 3: Differences among the mean values of above-ground biomass and carbon storage at 5% error probability for the models' estimations and the type of forest fragment analyzed.

Carbon Storage ($mg.ha^{-1}$)			
Model	**Average**	**Fragment**	**Average**
Chave et al. (2014)	91.1166 a	Rural	74.3973 a
Ratuchne (2010)	69.7340 b	Periurban	67.5308 a
Martins (2011)	49.5141 c	Urban	68.4366 a
Above-ground biomass ($mg.ha^{-1}$)			
Model	**Average**	**Fragment**	**Average**
Chave et al. (2014)	182.2382 a	Rural	148.9581 a
Ratuchne (2010)	200.5713 a	Periurban	123.9432 a
Martins (2011)	56.61020 b	Urban	166.5135 a

Means followed by the same letter, in the columns, do not differ from each other by Tukey's test (p>0.05)

The urban fragment presented the highest mean value for the above-ground biomass, 11.78% greater than the rural fragment, but no difference among forest fragments were found (p>0.05). On the other hand, the rural fragment presented mean value 9.23% greater than the periurban and 8.01% than the urban fragment, for the carbon storage estimated.

Concerning the rainfall interception, we did not find a significant difference (p>0.01) for the mean values in each season, and not among the mean values obtained at each forest fragment (Table 4 and Figure 4). Although, the urban fragment promoted more rainfall interception than the other fragments compared. The precipitation tends to be more variable in the Autumn and less variable in the Winter, but with a greater number of events in the Summer and less in the Winter (Figure 4 A). At the same time, a small quantity of rainfall interception during the Winter and greater in the other seasons (Figure 4 C).

The highest percentage of rainfall interception was observed in the Summer season (58.65%), followed by the Winter (52.31%), the Autumn (42.85%), the Spring (37.69%) seasons. Concerning the forest fragments studied, the urban fragment promoted the highest rainfall interception percentage (54.50%), followed by the periurban (47.65%) and the rural fragments (45.47%).

The number of species and the three main species were different among forest fragments. For the urban fragment, we have found 21 species with the main ones *Parapiptadenia rigida*, *Gymnanthes klotzschiana* and *Bauhinia forficata*. For the periurban fragment, we have found 31 species with the main ones *Parapiptadenia rigida*, *Cinnamodendron dinisii* and *Allophylus edulis*. For the rural fragment, we have found 57 species with the main ones *Araucaria angustifolia*, *Nectandra grandiflora* and *Ilex paraguariensis*. The species *Parapiptadenia rigida*, *Gymnanthes klotzschiana*, *Bauhinia forficata*, and *Allophylus edulis* are deciduous in the Mixed Ombrophilous Forest ecosystem (Carvalho, 1994; Lorenzi, 2000).

Table 4: Analysis of variance for the rainfall interception along the four seasons of the year and for the three forest fragments studied, with the source of variation (SV), degree of freedom (DF) and mean square (MS).

SV	DF	MS	p-value	Fragment	Rainfall interception (mm)
Seasons	3	36.189	0.2485	Urban	7.8270 a
Fragments	2	12.939	0.6076	Perirurban	6.7231 a
Residual	76	25.802		Rural	6.8194 a

Bartlett test: χ^2=8,74, DF=3, p-value=0.0328

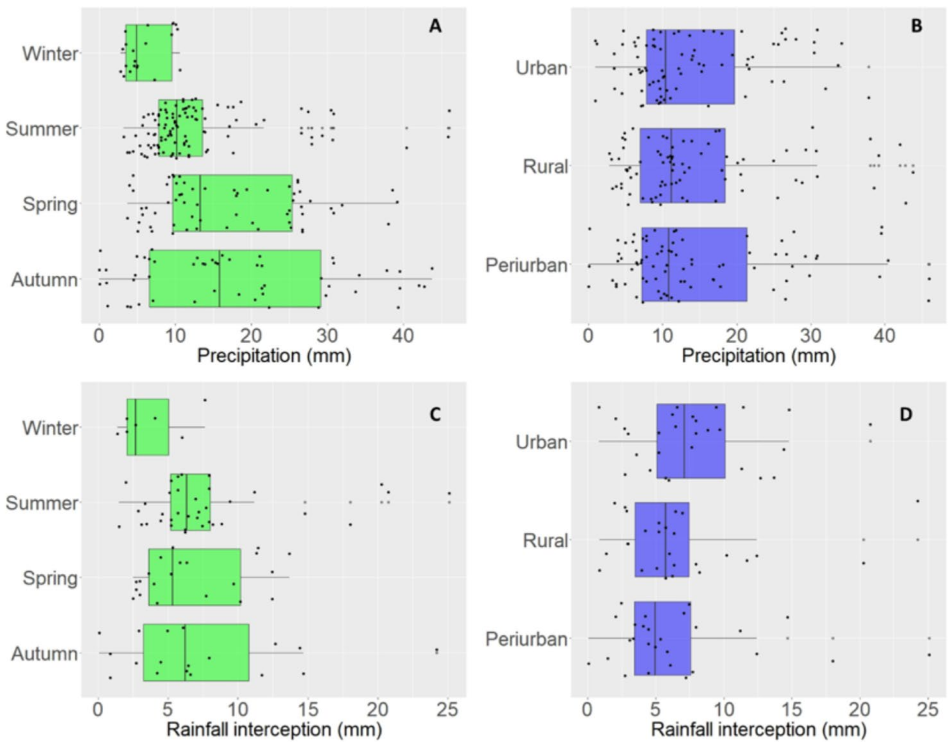

Figure 4: Precipitation (A and B) and rainfall interception (C and D) along the seasons of the year (A and C) and in the forest fragments studied (B and D). Each boxplot bar represents the 25th and 75th percentile, with the median vertical line inside.

DISCUSSION

The significant difference for the results obtained from the models (Tables 2 and 3) can be justified since each model is adjusted for a certain region and forest composition. Therefore, the most suitable model to use for the indirect carbon storage estimation is Ratuchne (2010), as it was adjusted for a database from forest in initial, medium, and advanced successional stages, and can be applied to different areas in the Montane Mixed Ombrophilous Forest ecosystem. The Martins model is also adjusted for the same phytophysionomy, but its adjustment was carried out for forest in the initial stage of secondary succession (Martins, 2011).

The statistical similarity among forest fragments in relation to the above-ground biomass and carbon storage might be related to the fact that the urban area, despite being isolated and inserted in an environment with frequent human activities, does not suffer from frequent management or annual disturbances (Pouyat et al., 2006; Edmondson et al., 2014) providing the ecosystem service of carbon storage in the same way as the conservation unit area (rural fragment).

Generally, carbon storage values are higher in better conserved sites (Muhd-Ekhzarizal et al., 2018; Kendie et al., 2019). The rural area is part of a conservation unit (National Forest category), although the portion studied is bordering an agricultural site, but the area sampled is distant from urbanized areas that poses continuous anthropic pressure. However, urban, and peri-urban areas, even under great anthropic pressure and consequently being characterized as low environmental quality, are storing carbon in the same way as the best-preserved area. This is an important finding since these areas play fundamental roles in the urban environment enabling a diversity of functions or services to be promoted (Dacanal, 2010; Schreyer et al., 2014; Zhang et al., 2015; Kendie et al., 2019).

The landscapes surrounding the forest fragments sampled (Figure 2) are formed by areas of different compositions, from a more urbanized and impervious land cover in the urban environment, to a transition between land covers and not so urbanized in the periurban area to a not urbanized but altered area in the rural place. These different landscapes changes can affect precipitation at local scale (their duration and frequency) by having characteristics of more continuous tree cover to a less continuous one, with more or less impervious surfaces (Torres-Valcárcel et al., 2014; Zhang et al., 2018).

The analysis of variance (Table 4) indicates that there was no significant difference ($p>0.05$) among the seasons of the year, which confirms that there is not a remarkable seasonality in rainfall distribution in the Mixed Ombrofilous Forest ecosystem (Salton et al., 2016; Gonçalves; Back, 2018; Back; Sônego; Pereira, 2020). It was found that in autumn there was the highest average of precipitation among all seasons (17.93 mm). Although the summer is considered the season with the highest precipitation trend in the State of Paraná (Gonçalves; Back, 2018), the season had the second lowest average in this study (12.72 mm). Summer was the season with the highest number of rainy days (12 days in 30 days sampled), but precipitation was well distributed throughout the days, unlike autumn, which had the highest average, because despite having fewer rainy days (6 days in 30 days sampled), precipitation was more intense.

The smaller difference (Figure 5) between rainfall in the border part (mean value = 10.41 mm) and inside the fragments (mean value = 7.80 mm), compared to the total precipitation outside (mean value = 14.36 mm) is justified by the edge effect promoted by the wind movement in the border part, with greater precipitation values than in the fragment center where there is greater canopy density with a not so intense movement due to the wind influence (Van Stan et al., 2015). So, studies aiming to show the effect of forest fragments over the rainfall interception must be performed in the core of the forest fragment avoiding the edge effect promoted by the wind during precipitation events.

For the seasons of the year, it was found that there was greater rainfall interception in the summer (58.65%), which usually is the season with the highest number of rainfall episodes (Salton et al., 2016). This result is extremely important for cities, since the presence of forest cover influences the redistribution of rainfall water, by damping, redirecting, and retaining the rainfall drops that could reach the ground (Oliveira Júnior; Dias, 2005), which consequently promote a positive impact by avoiding the runoff effect and mitigating problems caused by storms in urban areas, such as floods (Silva et al., 2008).

For autumn, there were high averages of precipitation, surpassing the wettest periods of the year, summer, and spring. According to SIMEPAR (2020), the autumn of 2019 suffered an adversity from the El Niño phenomenon, which caused more rainfall during the period than the usual (an increase of 35% to 45% in precipitation). In Paraná State, there is a tendency for a greater quantity of rainfall during the Spring and Summer seasons, but a greater volume and duration of each precipitation event during the Autumn and Winter seasons (Salton et al., 2016), which might have influenced the greater volume of rainfall collected during the Autumn (Figure 4A).

It can be seen that the three forest areas do not show significant differences among them regarding the rainfall interception (Table 4 and Figure 4D), suggesting that besides their different diversity and deciduous trees dominance, they promote the same contribution to this ecosystem service. The presence of tree cover, despite being less dense in urban and peri-urban areas when compared to rural areas, by with high plant density in the regenerating stratum, directly influence the distribution of precipitation and its interception (Fleischbein et al., 2005; Giglio; Kobiyama, 2013).

The importance of green areas within urban centers is not only related to their efficiency in intercepting and retaining rainfall, but also as climate regulators (Martini; Biondi, 2015). In conserved areas, knowledge about interception is essential to support species management rules, while for isolated fragments these studies are critical to support their protection and value which helps to avoid the risk of flooding in cities (Xiao; McPherson, 2003). In general, the results obtained showed the importance of maintaining urban green areas as isolated forest fragments to promote the rainfall interception, which is an important process in the hydrological cycle in forest ecosystems (Wang et al., 2016).

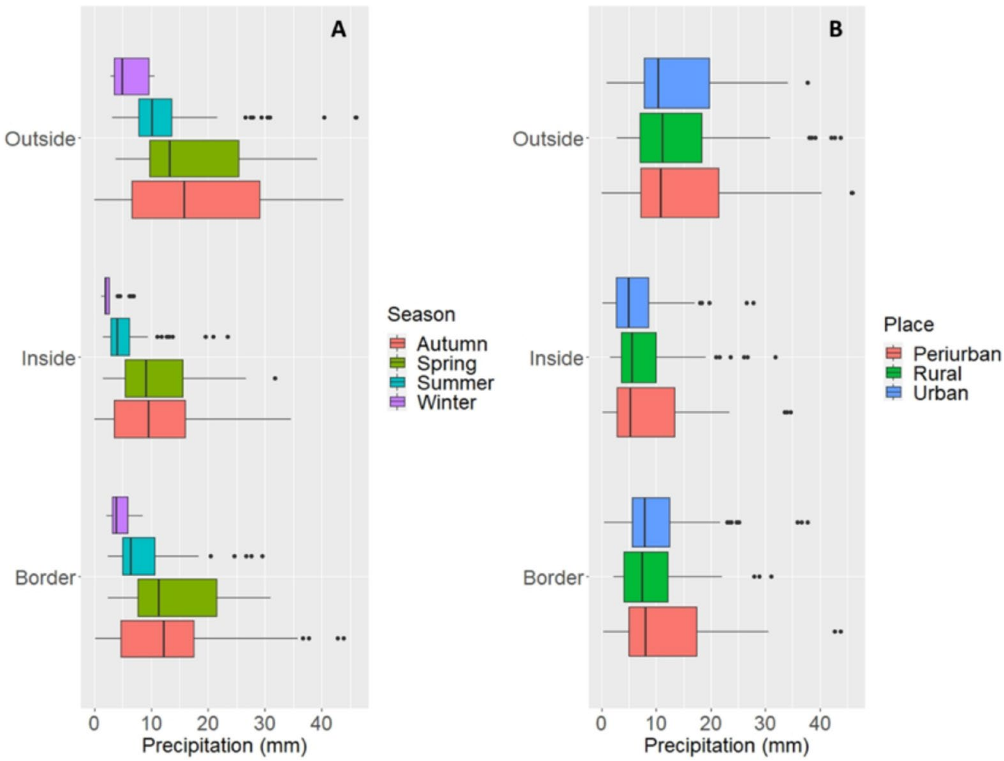

Figure 5: Precipitation distribution at each position considered to collect data (inside, in the border and outside each forest fragment), during the seasons (A) and at each place sampled (B). Each boxplot bar represents the 25th and 75th percentile, with the median vertical line inside.

As exploratory and experimental-based research, this study showed the value of isolated urban forest fragments to promote ecosystem services. Although the small scale of this study could be one limitation to point out and apply results, it brings and highlights new research perspectives to one not new but emerging interest to tackle climate change: the complex urban environment and its nature-based solutions.

CONCLUSIONS

In this exploratory and experimental-based research, the urban forest fragment, although lacking connectivity, showed a similar capacity to provide ecosystem services as other more connected or preserved forest fragments, in the periurban and rural areas. The carbon storage and the rainfall interception were provided in the same way in all forest fragments studied, which highlights the importance of maintaining and encouraging the increase of green areas and more continuous forest cover in cities to tackle climate change problems. As it is not possible or interesting to use direct methods to estimate the carbon storage in urban forest fragments, by cutting, measuring, and weighting whole trees, the use of local equations previously developed to be applied in the forest ecosystem studied promotes more reliable estimations than more general equations as a pantropical model. The position of the rainfall collector in the forest fragment, if in the center or in the border, does really affect the total precipitation evaluation and the rainfall interception estimates.

AUTHORSHIP CONTRIBUTION

Project Idea: RB, JTA

Database: JTA, RB

Processing: JTA, RB

Analysis: JTA, RB, PH, FGKB

Writing: JTA, RB, PH, FGKB

Review: RB, PH, FGKB

ACKNOWLEDGEMENTS

Authors thank the Coordenação de Aperfeiçoamento de Pessoal de Nível Superior - Brasil (CAPES) - Finance Code 001 for the provision of the doctoral scholarship to the first author, and the manager of the Forest Management Laboratory from UNICENTRO (Prof. Dr. Afonso Figueiredo Filho) for the data shared from the experiment in the National Forest of Irati.

REFERENCES

ARROYO-RODRIGUEZ, V.; FAHRIG, L.; TABARELLI, M.; WATLING, J. I.; TISCHENDORF, L.; BENCHIMOL, M. Designing optimal human-modified landscapes for forest biodiversity conservation. Ecology Letters, n.23, p.1404-1420, 2020.

AZEVEDO, A. D.; FRANCELINO, M. R.; CAMARA, R.; PEREIRA, M. G.; LELES, P. S. S. Carbon Stock in Atlantic Forest Restoration Areas. Forest, v.48, n.2, p. 183-194, 2018.

BACK, J. A.; SÔNEGO, M.; PEREIRA, J. R. Concentration indices and rainfall in southern Brazil. Brazilian Journal of Climatology, v. 27, p.57-72, 2020.

BAÚ, A. L.; GOMES, B. M.; QUEIROZ, M. M. F.; OPAZO, M. A. U.; SAMPAIO, S. C. Spatial behavior of probable monthly rainfall in the western mesoregion of Paraná state. Irrigate, v.11, n.2, p.150-168, 2006.

CARVALHO, P. E. R. Espécies florestais brasileiras: recomendações Silviculturais, potencialidades e uso da madeira. Brasília: Emprapa,1994. 640p.

CHAVE, J.; RÉJOU-MÉCHAIN, M.; BÚRQUEZ, A.; CHIDUMAYO, E. Improved allometric models to estimate the aboveground biomass of tropical trees. Global Change Biology, v.20, n.10, p.3177-3190, 2014.

CUNHA, T. J. F.; MADDARI, B. E.; CANELLAS, L. P.; RIBEIRO. et al. Soil Organic Matter and Fertility of Anthropogenic Dark Earths (Terra Preta de Índio) in the Brazilian Amazon Basis. Revista Brasileira de Ciência do Solo, v.33, n.1, p.85-93, 2009.

DACANAL, C.; LABAKI, L. C.; SILVA, T. M. L. Let's go for a walk in the forest! Thermal comfort in urban forest fragments. Built Environment, v.10, n. 2, p.115-143, 2010.

DECOCQ, G.; ANDRIEU, E.; BRUNET, J. et al. Ecosystem services from small forest patches in agricultural landscapes. Current Forestry Reports, v.2, p.30-44, 2016.

DERKZEN, M. L.; VAN TEEFFELEN, A. J. A.; VERBURG, P. H. Quantifying urban ecosystem services based on high- resolution data of urban green space: an assessment for Rotterdam, the Netherlands. Journal of Applied Ecology, v.52, n.4, p.1020-1030, 2015.

EDMONDSON, J. L.; DAVIES, Z. G.; MCCORMACK, S. A.; GASTON, K. J.; LEAKE, J. R. Effects of soil cover on soil organic carbon stocks in a European city. Science of the Total Environment, v.472, p.444-453, 2014

ESTAVILLO, C.; PARDINI, R.; DA ROCHA, P. L. B. Forest loss and the biodiversity threshold: an assessment considering the habitat requirements of species and the use of matrix habitats. Plos One, v.8, n.12, p.e82369, 2013.

FAHRIG, L. Why do several small patches hold more species than few large patches? Global Ecology and Biogeography, v.29, n.4, p.615-628, 2020.

FERRAZ, A. S.; SOARES, V. P.; SOARES, C. P. B.; RIBEIRO, C. A. A. S.; BINOT, D. H. B.; LEITE, H. G. Estimation of Biomass Stock in a Forest Fragment Using Orbital Images. Forest and Environment, v. 21, n.3, p.286-296, 2014.

FLEISCHBEIN, K.; WILCKE, W.; GOLLER, R.; BOY, J.; VALAREZO, C.; ZECH, W.; KNOBLIC, K. Rainfall interception in a lower montane forest in Ecuador: effects of canopy properties. Hydrological Processes, v.19, p.1355-1371, 2005.

FREITAS, J. P. O.; DIAS, H. C. T.; SILVA, E.; TONELLO, K. C. Net precipitation in a semideciduous forest fragment in the city of Viçosa, MG. Árvore, v.40, n.5, p.793-801, 2016.

GIGLIO, J. N.; KOBIYAMA, M. Rainfall interception: a review with emphasis on monitoring in Brazilian forests. Brazilian Journal of Water Resources, v.18, n.2, p.297-317, 2013.

GONÇALVES, F. N.; BACK, A. Analysis of spatial and seasonal variation and trends in precipitation in southern Brazil. Journal of Agricultural Sciences, v.41, n.3, p.592-602, 2018.

GROPPO, J. D.; SALEMI, L. F.; MORAES, J. M.; TREVISAN, R.; SEGHESI, G. B.; MARTINELLI, L. A. Water retention capacity of the vegetative canopy: comparison between Atlantic Forest and eucalyptus forest plantation. Forest Science, v.29, n.1, p.96-104, 2019.

HILLEBRAND, H.; BLASIUS, B.; BORER, E. T.; CHASE, J. M.; DOWNING, J. A.; ERIKSSON, B. K. Biodiversity change is uncoupled from species richness trends: Consequences for conservation and monitoring. Journal of Applied Ecology, n.55, p.169-184, 2017.

GELMI-CANDUSSO, T. A.; HÄMÄLÄINEN, A. M. Seeds and the City: The Interdependence of Zoochory and Ecosystem Dynamics in Urban Environments. Frontiers in Ecology and Evolution, v.7, p.1-19, 2019.

HODGES, M. N.; MCKINNEY, M. Urbanization impacts on land snail community composition. Urban Ecosystems, v.21, n.4, p.721-735, 2018.

IVES, C. D.; LENTINI, P. E.; THRELFALL, C. G. et al. Cities are hotspots for threatened species. Global Ecology and Biogeography, v.25, n.1, p.117-126, 2016.

LIANG, J.; CROWTHER, T. W.; PICARD, N. et al. Positive biodiversity-productivity relationship predominant in global forests. Science, v.354, n.6309, 2016.

LIU, J.; ZHANG, Z.; ZHANG, M. Impacts of forest structure on precipitation interception and runoff generation in a semiarid region in northern China. Hydrological Processes, v.32, n.15, p.2362-2376, 2018.

LORENZI, H. Árvores brasileiras: manual de identificação e cultivo de plantas arbóreas nativas do Brasil. 3 ed. Nova Odessa: Instituto Plantarum, 2000. 351p.

LUEDERITZ, C.; BRINK, E.; GRALLA, F. et al. A review of urban ecosystem services: six key challenges for future research. Ecosystem Service, v.14, p.98-112, 2015.

MADRE, F.; VERGNES, A.; MACHON, N.; CLERGEAU, P. Green roofs as habitats for wild plant species in urban landscapes: first insights from a large-scale sampling. Landscape and Urban Planning, v.122, p.100-107, 2014.

MARTINI, A.; BIONDI, D. Microclimate and Thermal Comfort of an Urban Forest Fragment in Curitiba, PR. Forest and Environment, v.22, n.2, p.182-193, 2015.

MARTINS, P. J. Biomassa vegetal, estoque de carbono e dinâmica em um fragmento de Floresta Ombrófila Mista Montana. Dissertação (Mestrado em Agronomia) - Universidade Estadual do Centro-Oeste, Guarapuava, 2011, 86f.

MORI, A. S.; LERTZMAN, K. P.; GUSTAFSSON, L. Biodiversity and ecosystem services in forest ecosystems: a research agenda for applied forest ecology. Journal of Applied Ecology, v. 54, n.1, p.12-27, 2017.

MUHD-EKHZARIZAL, M.; MOHD-HASMADI, E. I.; HAMDAN, O.; MOHAMAD-ROSLAN, M. K.; SHAILA, N. S. Estimation of aboveground biomass in mangrove forests using vegetation indices from SPOT-5 image. Journal of Tropical Forest Science, v.30, n. 2, p.224-233, 2018.

NEWBOLD, T.; HUDSON, L. N.; HILL, S. L. L. et al. Global effects of land use on local terrestrial biodiversity. Nature, n.520, p.45-50, 2015.

OLIVEIRA JÚNIOR, J. C.; DIAS, H. C. T. Effective precipitation in a secondary fragment of the Atlantic Forest. Tree Magazine, v.29, n.1, p.9-15, 2005.

OLIVEIRA, L. Determinação da interceptação da chuva pelo dossel de mata ripária no Distrito Federal. Trabalho de Conclusão de Curso (Bacharelado em Gestão Ambiental), Universidade de Brasília, Planaltina, 2014, 20 f.

PFEIFER, M.; LEFEBVRE, V.; PERES, C.; BANKS-LEITE, C.; WEARN, O. R.; MARSH, C. J. The creation of forest edges has a global impact on forest vertebrates. Nature, v.551, n.7679, p.187-191, 2017.

POUYAT, R. V.; YESILONIS, I. D.; NOWAK, D. J. Carbon storage in urban soils in the United States. Journal of Environmental Quality, v.35, n.4, p.1566-1575, 2006.

RATUCHNE, L. C. Ajuste e seleção de equações alométricas para a estimativa de biomassa, carbono e nutrientes em uma Floresta Ombrófila Mista. Dissertação (Mestrado em Agronomia) – Universidade Estadual do Centro-Oeste, Guarapuava, 2010, 98f.

ROVEDA, M.; PELISSARI, A. L.; GENÚ, A. M.; FIGUEIREDO FILHO, A.; ROIK, M. Modelagem geoestatística da estrutura espacial arbórea e dos atributos do solo em Floresta Ombrófila Mista. Floresta, v.48, n.1, p.67-76, 2018.

SALTON, F. G.; MORAIS, H.; CARAMORI, P. H.; BORROZZINO, E. Climatology of Precipitation Episodes in Three Locations in the State of Paraná. Brazilian Journal of Meteorology, v.31, n.4, p.626-638, 2016.

SAWCZUK, A. R.; FIGUEIREDO FILHO, A.; DIA, A. N.; WATZLAWICK, L. F.; STEPKA, T. F. Alterations in the horizontal structure, in the period of 2002-2008, in Mixed Ombrophylous Forest in the center-south of the state of Paraná. Forest Science, v.24, n.1, p.149-160, 2014.

SCHREYER, J.; TIGGES, J.; LAKES, T.; CHURKINA, G. Using Airborne LiDAR and QuickBird Data to model carbon storage of urban trees and their distribution - a case study from Berlin. Remote Sensing, v.6, n.11, p.10636-10655, 2014.

SIMEPAR - SISTEMA DE TECNOLOGIA E MONITORAMENTO AMBIENTAL DO PARANÁ. Boletim climatológico. 2020. Available at: <http://www.simepar.br/prognozweb/simepar/timeline/boletim_climatologico>. Accessed in: May 23th 2020.

STROHBACH, M. W.; HAASE, D. Above-ground carbon storage by urban trees in Leipzig, Germany: analysis of patterns in a European city. Landscape and Urban Planning, v.104, n.1, p.95-104, 2012.

SUTTON, P. C.; ANDERSON, S. J. Holistic valuation of urban ecosystem services in New York City's Central Park. Ecosystem Services, v.19, p.87-91, 2016.

TORRES-VALCÁRCEL, A.; HARBOR, J.; GONZÁLEZ-AVILÉS, C.; TORRES-VALCÁRCEL, A. Impacts of Urban Development on Precipitation in the Tropical Maritime Climate of Puerto Rico. Climate, v.2, n.2, p.47-77, 2014.

VAN STAN, J. T.; LEVIA, J. R.; JENKINS, R. B. Loss of interception of the forest canopy on time scales: implications for urban afforestation initiatives. The professional geographer, v.67, n.1, p.41-51, 2015.

XIAO, Q.; McPHERSON, E. G. Rainfall interception by Santa Monica's municipal urban forest. Urban Ecosystems, v.6, p.291-302, 2003.

ZHANG, D.; ZHENG, H.; REN, Z.; ZHAI, C.; SHEN, G.; MAO, Z.; WANG, P.; HE, X. Effects of forest type and urbanization on carbon storage of urban forests in Changchun, northeastern China. Journal of Geographical., v.25, p.147-158, 2015.

ZHANG, Y.; PANG, X.; XIA, J.; SHAO, Q.; YU, E.; ZHAO, T.; SHE, D.; SUN, J.; YU, J.; PAN, X.; ZHAI, X. Regional Patterns of Extreme Precipitation and Urban Signatures in Metropolitan Area. Journal of Geophysical Research Atmospheres, v.124, n.2, p.641-663, 2018.

WANG, C.; YANG, J.; MYINT, S. W.; WANG, Z.; TONG, B. Empirical modeling and spatiotemporal patterns of urban evapotranspiration for the metropolitan area of Phoenix, Arizona. GIScience & Remote Sensing, v.53, n.6, p.778-792, 2016.

WATZLAWICK, L. F.; MARTINS, P. J.; RODRIGUES, A. L.; EBLING, A. A.; BALBINOT, R.; LUSTOSA, S. B. C. Carbon contents in species from the Mixed Ombrophylous Forest and effect of ecological group. Cerne, v.20, n.4, p.613-620, 2014.