

Impact of the Urbanisation Process in the Availability of Ecosystem Services in a Tropical Ecotone Area

Lucianna Marques Rocha Ferreira,¹ Luciana S. Esteves,^{2*}
Enio Pereira de Souza,³ and Carlos Antonio Costa dos Santos³

¹Centre of Technology and Natural Resources, Federal University of Campina Grande, Campina Grande 58429-140, Brazil; ²Faculty of Science and Technology, Bournemouth University, Poole BH12 5BB, UK; ³Department of Atmospheric Sciences, Federal University of Campina Grande, Campina Grande 58429-140, Brazil

ABSTRACT

Urbanisation has been a main cause of land use land cover (LULC) change worldwide, often with irreparable consequences to the provision of ecosystem services. Despite an increase in quantitative assessments of ecosystem service values (ESV) related to LULC changes, data are scarce for ecotones, such as the agreste in northeast Brazil (a transitional area between the Atlantic Forest and the Caatinga biomes). The benefit transfer method was used to quantify changes in ESV between 1989, 2007 and 2014 due to urbanisation in the microwatershed Riacho das Piabas (3660 ha) in the agreste of Paraíba. Remote sensing techniques and a geographic information system were used to quantify LULC changes. Loss of arboreal vegetation (covering 46% of the study area in 1989 and 5% in 2014) was the key factor driving the 73.2% decline in the total ESV (from US\$ 13.7 million to US\$ 3.7 million in 2017 values). LULC changes resulted in losses of 89% in the estimated value of eight

ecosystem services, including climate regulation, water flow regulation, moderation of disturbance, nutrient cycling and biological control, which are critical locally when considering the regional trend towards aridification and the existing pressures on water resources. Ecosystem functions loss and climate change impacts may lead to a shift in ecotone boundaries favouring the semiarid Caatinga vegetation. These results urge the implementation of ecosystem-based spatial planning, focusing on urban green infrastructure and restoration of natural habitats and their connectivity, to prevent further ecosystem service losses. Local estimates of ESV required to inform the suggested policy actions are identified.

Key words: microwatershed; land use land cover; urbanisation; natural resources; ecosystem services valuation; agreste; Brazil.

Received 25 January 2018; accepted 20 May 2018;
published online 5 June 2018

Electronic supplementary material: The online version of this article (<https://doi.org/10.1007/s10021-018-0270-0>) contains supplementary material, which is available to authorized users.

Authors' Contribution LMRF and EPS designed the study; LMRF and LSE undertook most of the data processing and drafting the article; LMRF, EPS, LSE and CACS contributed to data analyses and crafting the article.

*Corresponding author; e-mail: lesteves@bournemouth.ac.uk

INTRODUCTION

Ecosystem dynamics are influenced by land use land cover (LULC), especially in the ecological functions that reflect into ecosystem services (Kindu and others 2016; Tolessa and others 2017). Ecosystem services benefit humans, directly or indirectly, through the supply of goods (for example, water, food and raw material), life support,

physical, mental and spiritual well-being and development of economic activities (MEA 2005; Hails and Ormerod 2013; Costanza and others 2014).

Urbanisation is one of the main drivers of LULC changes globally (Elmqvist and others 2013), resulting in long-lasting (for example, McKinney 2002) or irreversible (for example, Seto and others 2011) environmental impacts. Increasing population and urbanisation causes pressure on natural resources and high demand for ecosystem services, which combined can lead to critical environmental degradation, such as water crisis, air pollution, microclimatic alteration and collapse of natural resources (Seto and others 2011; Solecki and others 2013). Loss of ecosystem services due to human-driven conversion of vegetated areas into urban land (for example, Tianhong and others 2010; Mendoza-González and others 2012; Estoque and Murayama 2013; Crespín and Simonetti 2016; Yi and others 2017) has been widely reported in developing countries, where urban centres are largely (and increasingly) dependent on the ecological integrity of surrounding rural areas (Hails and Ormerod 2013).

It is estimated that between 2000 and 2030 urban areas will expand by around 200% and approximately 5 billion people will be living in cities at the end of the period (Fragkias and others 2013). In this context, Groffman and others (2017) highlighted two challenges for the science of urban ecosystems: (1) the ability to predict and explain structural and functional patterns of ecosystems under altered conditions (for example, urbanisation); and (2) assimilating the understanding of an integrated socio-ecological system, in which humans are an inseparable part of ecosystems.

The identification and valuation, monetary or otherwise, of ecosystem services are increasingly the focus of research worldwide (Nelson and others 2009; Balvanera and others 2012; McDonough and others 2017). These studies have quantified gains and losses of ecosystem services brought about by spatial and temporal changes (Kreuter and others 2001; Mendoza-González and others 2012; Tolessa and others 2017; Yi and others 2017); applied the existing knowledge to inform decision-making and policy development (Green and others 2016); and raised public awareness through environmental education (de Groot and others 2012; Tolessa and others 2017). Some biomes and types of ecosystems (for example, coastal and inland wetlands and tropical forests) seem to have attracted most of the attention, whereas others are still poorly studied. For example, in *The Economics of Ecosystems and*

Biodiversity (TEEB) database (Van der Ploeg and de Groot 2010), there are 68 entries for monetary value estimates of ecosystem services in coastal wetlands, 61 in tropical forest and only one in urban settings, two in deserts and nine in marine ecosystems.

Very little is known about ecosystem services in ecotones or how their availability is affected by urbanisation or other LULC changes. Ecotones are often neglected in terms of conservation despite being important buffer zones to adjacent biomes (Scarano 2009) and of relative high biodiversity (Bueno and others 2017). Fragmentation of habitat and biodiversity loss are known effects of urbanisation, and of great importance for ecotones, such as the *agreste*. This article presents the first quantitative assessment of loss in ecosystem services values (ESV) associated with LULC changes in the *agreste*. The *agreste* is an ecotone between the Atlantic Forest and the *Caatinga*, two Brazilian biomes (included in the World Network of Biosphere Reserves) under great threat from human and climate pressures. Besides being neglected in conservation efforts, the *Caatinga* (and its ecotones) is one of the least studied biomes and one of the most threatened natural vegetation in Brazil (for example, Moro and others 2016).

The assessment focuses on changes in ESV associated with rapid urbanisation occurring over a 25-year period (1989, 2007 and 2014) in the microwatershed Riacho das Piabas (MWRP), state of Paraíba, Brazil. ESV is used here as a proxy to quantify changes in ecosystem services availability, rather than an accurate estimate of their monetary value. Watersheds are recognised as the spatial planning unit for water resources management by the National Policy for Water Resources (Law 9433, 8 January 1997). The MWRP supported and was greatly affected by the growth of Campina Grande, a city known as the technology centre of the *agreste* and for its contribution to the regional economy.

First, this article summarises the key characteristics of the study area and its relevance to the *agreste*. Then the remote sensing techniques used to classify LULC (detailed in the online supplementary material S1) and the benefit transfer method applied to estimate ESV for each LULC are described. The quantification of LULC and ESV changes are presented and discussed, including an evaluation of the suitability of the monetary estimates for ecosystem services (coefficient values in US\$ ha⁻¹ a⁻¹) calculated by de Groot and others (2012) to reflect ESV changes in the study area. The discussion offers suggestions of management

strategies that can be applicable to the agreste and other areas of similar ecological importance undergoing rapid and disorganised urbanisation.

STUDY AREA

The MWRP extends over 3659.82 ha within the Paraíba River Basin, state of Paraíba, northeast Brazil, spreading across three municipalities, mostly within Campina Grande (Figure 1). The study area is dominated by semideciduous and deciduous forest (greatly deforested) and xerophytic flora, reflecting an ecological tension between open ombrophylous forest and savanna-steppe (Moro and others 2016). Natural vegetation covered 67.4% of the Caatinga biome in 1990 declining to 63.2% in 2010 at average annual rates of -0.19% between 1990 and 2000 and -0.44% between 2000 and 2010 (Beuchle and others 2015). Data on LULC changes specific to the agreste were not found, but rates of vegetation cover loss are assumed to be similar or higher than reported for the Caatinga.

The climate of this region is hot and humid with a dry season in the summer, type As' according to the Köppen-Geiger classification (Kottek and others 2006). The annual rainfall is approximately 800 mm with the wettest period occurring between March and June (Macedo and others 2011). The combination of climate and geomorphology resulted in a region dominated by non-cultivable land, with limitations to permanent crops and steep lands susceptible to erosion (AESA 2010). Climate predictions for the semiarid Northeast Brazilian indicate increased temperatures and decreased rainfall in the twenty-first century, leading to increased risk of desertification and important socioeconomic impacts (for example, Marengo and others 2017; Vieira and others 2015).

The 'Trapeiros da Borborema' (traditional travelling traders of the region) settled along the banks of the Piabas creek attracted by the easy access to drinking water and the availability of pasture for horses and donkeys, important assets in the agreste. These settlements gave rise to Campina Grande, the most important cotton growing area in Brazil in the



Figure 1. The microwatershed Riacho das Piabas is located in the state of Paraíba, Northeast Brazil.

early 1900s and now the most important city in the agreste. Droughts are frequent and cause critical impacts in the region. The great drought of 1824–1828 resulted in the transformation of the Piabas creek into a large water reservoir (called Açude Velho) to supply Campina Grande. Due to degradation of water quality, the Açude Velho is no longer used as a supply for human consumption, but through time it became the city's cultural heritage and iconic landmark (Câmara 2006).

Similar to other locations in Brazil and other developing countries, human occupation in the MWRP was mainly unplanned and disregarded the impacts on local ecological dynamics. The middle and downstream sectors of the MWRP are urbanised or channelled, surrounded by illegal housing built in designated Permanent Preservation Areas, which are protected by Brazilian legislation. Water contamination is an important issue, aggravated by an inadequate sewage system. The upstream sector of the MWRP, although mostly rural, is impacted by unregulated development within private properties, including construction of small dams, removal of riparian vegetation and farming activities.

MATERIALS AND METHODS

LULC Classification

The LULC classification was based on the normalised difference vegetation index (NDVI), which is used for both monitoring and interannual comparisons of vegetation cover (Jensen 2006). The NDVI was obtained from the analysis of Landsat/TM 5 (Land Remote Sensing Satellite Thematic Mapper) images taken in 1989 and 2007 and Landsat 8/Operational Land Imager and Thermal Infrared Sensor images taken in 2014. The satellite images used in this work were downloaded from the United States Geological Survey (USGS 2014) Global Visualization Viewer. The satellite images were georeferenced to UTM WGS 84, Zone 25 South using the orthorectified Landsat 8 image as reference. Ground-truth validation of the LULC classification was performed on 60 geographic control points using a GPSMAP Garmin 64S tied to the Global Navigation Satellite System.

The vegetation in the study area is dominantly deciduous, with plant species that grow their leaves after a few days of rainfall and shed them during dry periods. Therefore, a careful selection of images is essential to ensure consistency in the spectral response of vegetation and the associated range of NDVI values. Only images with low cloud cover captured on days preceding precipitation events

were used in the analysis (Table 1). Digital image processing, calculations of the NDVI and the thematic maps were produced using the Geographic Information System QGIS 2.8.3. Image processing to generate the NDVI from Landsat 5 images included radiometric calibration and monochromatic reflectance as described in Waters and others (2002). To obtain NDVI from the Landsat 8 image, the reflectance at the top of atmosphere was calculated according to USGS (2016). The NDVI was then calculated using equation (1) (Rouse and others 1974).

$$\text{NDVI} = \left(\frac{\rho_{\text{nir}} - \rho_r}{\rho_{\text{nir}} + \rho_r} \right) \quad (1)$$

where ρ_{nir} is the radiant flux reflected in the near-infrared and ρ_r is the radiant flux reflected in red (in J s^{-1}). The NDVI values range from -1 to 1 , so that pixel values closer to 1 represent greater vegetation vigour (Jensen 2006). Further details of the calculations to obtain the NDVI from the satellite images are presented in the online supplementary material (S1).

Based on the NDVI values, six LULC classes were identified in the study area: water (includes natural and artificial water bodies), grasslands (areas dominated by grasses, including cultivated land), shrublands (dominated by scrublands and savanna-type vegetation), arboreal vegetation (dominated by trees), urban area and bare lands (Table 2). In the study area, cultivated land tends to be in small plots and used seasonally (regulated by rainfall and water availability). As the LULC were classified from imagery obtained on 'dry conditions', cultivated areas are depicted as grasslands. Urban areas correspond to areas characterised by impermeable surfaces (for example, houses, buildings and paved roads or streets), whereas bare lands include areas without vegetation and dirt roads. Green urban areas large enough to be resolved by the satellite images are classified within one of the vegetated LULC.

Estimating the Ecosystem Services Value

The total ESV for the study area was estimated using the benefit transfer method (Figure 2), which has been widely used (for example, Kreuter and others 2001; Estoque and Murayama 2013; Crespin and Simonetti 2016; Tolessa and others 2017; Yi and others 2017) to assist assessments in areas where local valuations are lacking (Mendoza-Gonzalez and others 2012; Rolfe and others 2015; Kindu and others 2016). Here the coefficient values ($\text{US\$ ha}^{-1} \text{ a}^{-1}$) calculated by de Groot and others

Table 1. Date and Specification of the Satellite Images Analysed in This Study. *Source:* USGS 2014

Imagery date	Path/raw	Satellite/sensor	Radiometric/space resolution
10/07/1989	214/65	Landsat 5/TM	8 bits/30 m (bands 1, 2, 3, 4, 5 e 7) e 120 m (band 6)
29/08/2007	214/65	Landsat 5/TM	8 bits/30 m (bands 1, 2, 3, 4, 5 e 7) e 120 m (band 6)
26/04/2014	214/65	Landsat 8/OLI and TIRS	16 bits/15 m (band 8), 30 m (bands 1, 2, 3, 4, 5, 6, 7 e 9) and 100 m (bands 10 e 11)

Table 2. LULC Classes, Their Respective NDVI Range, Equivalent Biome and Value Coefficient (VC)

LULC classes	NDVI range	Equivalent biome	VC _{LULC} (2017 US\$ ha ⁻¹ a ⁻¹)
Water	– 1–0	Rivers and lakes	5199.86
Urban area	0.01–0.3	–	–
Bare lands	0.31–0.4	–	–
Grasslands	0.41–0.5	Grasslands	3499.89
Shrublands	0.51–0.6	Woodlands	1935.19
Arboreal vegetation	0.61–0.9	Tropical forests	6413.61

VC from de Groot and others (2012) adjusted to the Consumer Price Index of November 2017.

(2012) were used to quantify relative gains or losses in ESV due to LULC changes in the MWRP. The value coefficient of the biome identified by de Groot and others (2012) best matching each LULC class identified in the study area (Table 2) was used as a proxy for the local ESV. The description of the 'biome' woodland (in de Groot and others 2012) includes vegetation types such as savannas, shrublands and scrublands, which are a good representation of the vegetation found in the LULC shrublands. The tropical forests 'biome' includes deciduous/semideciduous tropical forests, which more closely relate to the vegetation types found in the arboreal forest LULC.

The match between the LULC classes and the biomes represented in the study of de Groot and others (2012) was not perfect. Therefore, sensitivity analyses were conducted to determine whether variations in the coefficient values would result in unacceptable uncertainties associated with the unit value transfer. The coefficient values used to estimate the ESV of the four LULC classes (water, grassland, shrublands and arboreal vegetation) were adjusted by 50%, and the coefficient sensitivity (CS) was calculated using equation (2) following the standard economic concept of elasticity (Mansfield 1985), as proposed by Kreuter and others (2001) and applied by many (for example, Li and others 2007; Hu and others 2008; Crespin and Simonetti 2016; Kindu and others 2016).

$$CS = \frac{(ESV_j - ESV_i)/ESV_i}{(VC_{jk} - VC_{ik})/VC_{ik}} \quad (2)$$

where ESV is the estimated total ecosystem service value, VC is the unit value coefficient (in US\$ ha⁻¹ a⁻¹), *i* and *j* represent the initial and adjusted values, respectively, and *k* is the LULC class. CS indicates the proportion of change in ESV relative to the proportion of change in VC. If CS is greater than 1, the estimated total ESV is considered elastic or very sensitive to the VC, suggesting that a more accurate value coefficient is needed (Kreuter and others 2001). If CS is less than 1, the estimated total ESV is inelastic and robust, indicating that the VC is acceptable even if not very accurate (Kindu and others 2016).

To identify the ecosystem services most affected by land use change in the MWRP, the value of each service associated with the LULC class was calculated using the average monetary value estimated by de Groot and others (2012) of that ecosystem service for the equivalent biome. All monetary values and VC estimated by de Groot and others (2012) were adjusted to November 2017 values (Table 3) using the Consumer Price Index inflation calculator of the *US Bureau of Labor Statistics* inflation calculator (available at https://www.bls.gov/data/inflation_calculator.htm).

Similar to the approach taken by others (for example, Mendoza-González and others 2012;

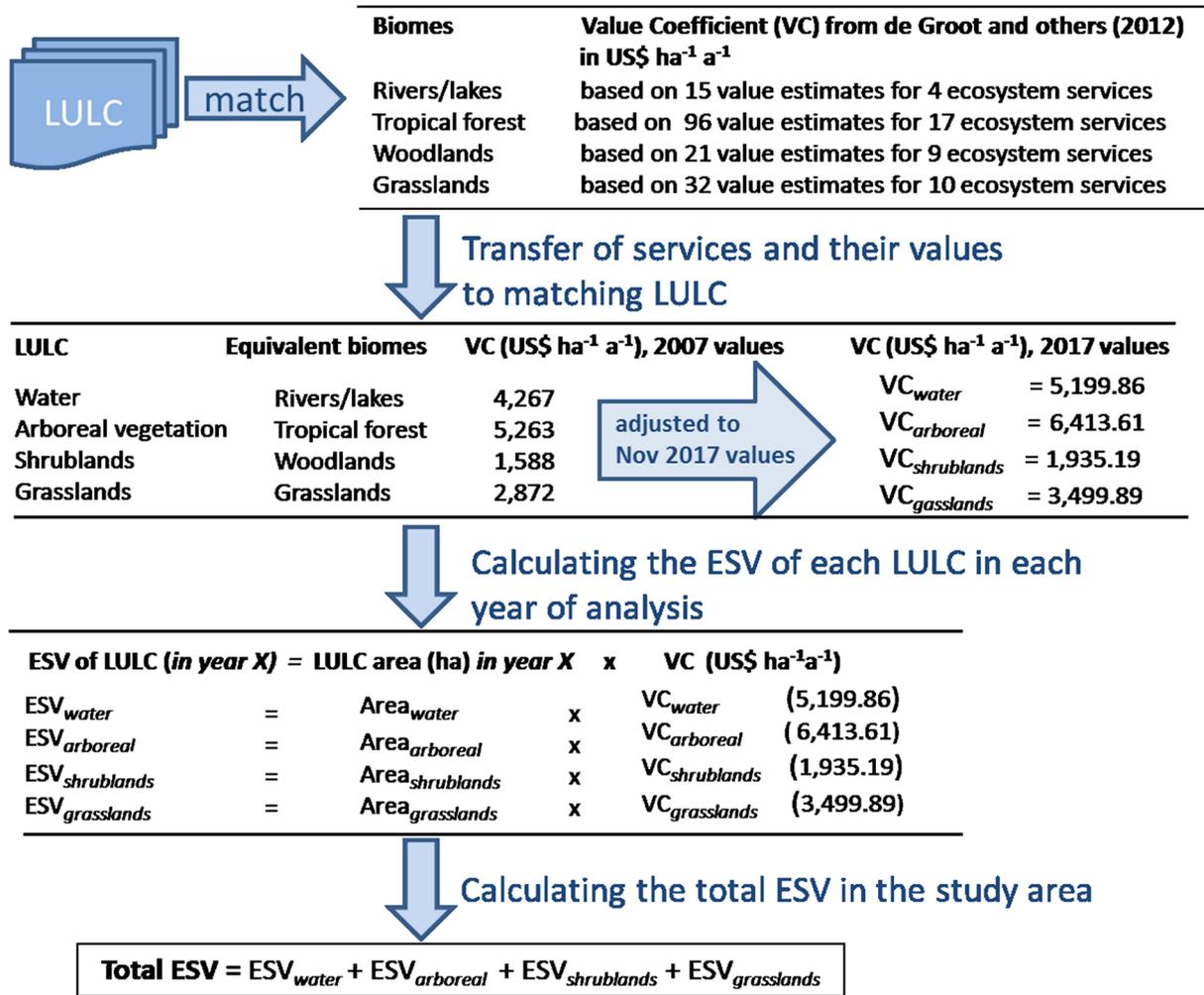


Figure 2. Steps of the benefit transfer method used to estimate the ecosystem service values in the study area.

Crespin and Simonetti 2016), the LULC classes of bare lands and urban areas were excluded from the calculations of total ESV. The key objective here is to quantify the impact of urbanisation on the provision of ecosystem services using the relative change in ESV as an indicator. In the study area, bare lands and urban areas expanded at the expense of the natural vegetation, resulting in net loss of ecosystem services; therefore, justifying that their contribution to local ESV is considered negligible for the purpose of this study. Note that the ESV calculations include the contribution of green urban areas, as these are classified within one of the vegetated LULC.

RESULTS

Land Use Change

There were considerable changes in LULC between 1989 and 2014 in the MWRP, with marked differences in spatial and temporal distribution (Figure 3). In this period, the largest relative changes (in percentage of initial area) were a 465% increase of urban area and a 89% decrease in arboreal vegetation (Table 4). In 1989, arboreal vegetation was the dominant class covering 46% of the study area (Table 4), mainly in the northern and southern sectors of the study area (Figure 3). In contrast, this class covered less than 5% of the MWRP in 2014 (Table 4) and was substituted at average rates of 61 ha a⁻¹, mainly by grasslands and shrublands

Table 3. Ecosystem Services and Their Monetary Value (2017 US\$ ha⁻¹a⁻¹) for Each Biome Equivalent to Local LULC

Ecosystem service	Rivers and lakes	Grasslands	Woodlands	Tropical forest
<i>Provisioning services</i>				
Food	129.17	1452.60	63.37	243.72
Water	2203.27	73.12		32.90
Raw materials		64.59	207.17	102.36
Genetic resources				15.84
Medicinal resources		1.22		1832.81
Ornamental resources			39.00	
Sum	2332.44	1591.53	309.54	2227.63
<i>Regulating services</i>				
Air quality regulation				14.62
Climate regulation		48.74	8.53	2490.87
Moderation of disturbance				80.43
Water flow regulation				416.77
Waste treatment	227.88	91.40		7.31
Erosion prevention		53.62	15.84	18.28
Nutrient cycling				3.66
Pollination			37.78	36.56
Biological control				13.40
Sum	227.88	193.76	62.15	3081.90
<i>Support services</i>				
Nursery service			1551.31	19.50
Genetic diversity		1479.41	3.66	28.03
Sum		1479.41	1554.97	47.53
<i>Cultural services</i>				
Aesthetics information		203.51		
Recreation	2639.54	31.68	8.53	1056.55
Sum	2639.54	235.19	8.53	1056.55
ESV _{LULC}	5199.86	3499.89	1935.19	6413.61

Values from de Groot and others (2012) adjusted to the Consumer Price Index of November 2017.

in the northern sector and by urban areas elsewhere (Figure 3). Urban areas covered less than 10% of the study area in 1989 and 56% in 2014 (Table 4), at an average rate of 67 ha a⁻¹, reflecting the rapid growth of Campina Grande in the central sector of MWRP (Figure 3). The rate of urban sprawl declined from an average of 74 ha a⁻¹ between 1989 and 2007 to 49 ha a⁻¹ between 2007 and 2014, expanding mainly in the southern sector in the latter period (Figure 3).

In the period 1989–2014, bare lands and grassland showed a small increase in area (3.8 and 3.4%, respectively), but changes were variable through time (Table 4). Between 1989 and 2007, there was a reduction in the area of these LULC classes mainly due to urban encroachment in the central sector. The increase in bare lands and grasslands areas observed between 2007 and 2014 resulted from degradation and substitution of vegetated areas, particularly in the north sector (Figure 3). Evidence of this degradation is provided by

the contrasting changes in the extent of shrublands, which increased 3% between 1989 and 2007 and decreased 44% between 2007 and 2014 (Table 4). The data show a clear pattern of arboreal vegetation being substituted by shrublands, which in turn were later changed to grasslands and these were then replaced by urban areas and bare lands.

Despite the importance in triggering land use changes in the study area, water bodies have a modest presence, occupying only 0.1% of the MWRP in 1989 and 0.5% in 2014 (Table 4). The increase of 399% between 1989 and 2007 resulted from the creation of a reservoir to control the water flow of Açude Velho.

Changes in the Availability of Ecosystem Services

Between 1989 and 2014, the total ESV in the study area decreased 73% (from US\$ 13.7 million to US\$ 3.7 million) mainly due to losses of arboreal vegetation (Table 5). The rate of average annual loss

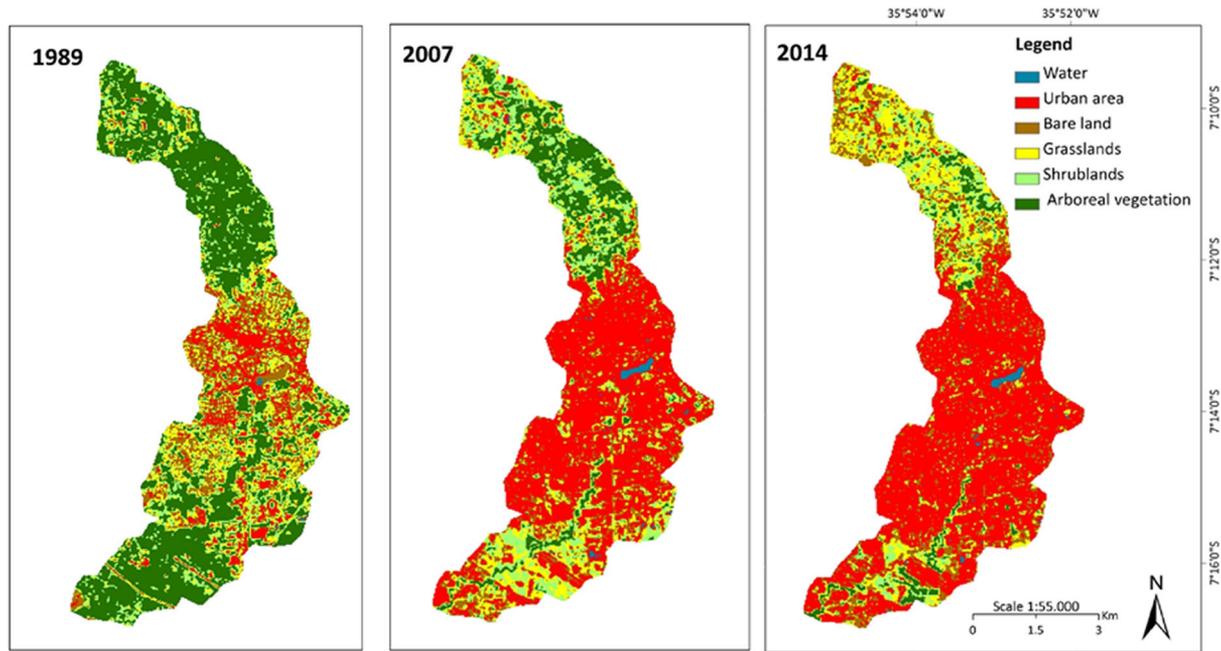


Figure 3. Land use land cover in the microwatershed Riacho das Piabas (Northeast Brazil) in 1989, 2007 and 2014.

Table 4. Extent (ha) of Each LULC Class in 1989, 2007 and 2014, the Respective Cover (%) Relative to the Study Area and Land Use Change (%) in the Microwatershed Riacho das Piabas, Brazil

LULC classes	1989		2007		2014		Change (%)		
	Area (ha)	%	Area (ha)	%	Area (ha)	%	1989–2007	2007–2014	1989–2014
Water	4.31	0.12	21.52	0.59	19.21	0.52	399.30	– 10.73	345.71
Urban area	361.85	9.89	1702.57	46.52	2042.94	55.82	370.52	19.99	464.58
Bare lands	580.81	15.87	439.54	12.01	602.70	16.47	– 24.32	37.12	3.77
Grasslands	504.1	13.77	475.70	13.00	521.00	14.24	– 5.63	9.52	3.35
Shrublands	513.63	14.03	526.78	14.39	292.56	7.99	2.56	– 44.46	– 43.04
Arboreal vegetation	1695.12	46.32	493.71	13.49	181.42	4.96	– 70.87	– 63.25	– 89.30
Sum	3659.82	100	3659.82	100	3659.83	100			

Table 5. Total Ecosystem Service Value (2017 US\$) in 1989, 2007 and 2014 per LULC Classes in the Microwatershed Riacho das Piabas (Brazil) and the Respective Change Through Time (in US\$ and % of Initial Value)

LULC classes	ESV _{LULC} (2017 US\$)			ESV _{LULC} (2017 US\$)					
	1989	2007	2014	1989–2007	%	2007–2014	%	1989–2014	%
Water	22,411	111,901	99,889	89,490	399.3	– 12,012	– 10.7	77,478	345.7
Grasslands	1,764,295	1,664,898	1,823,443	– 99,397	– 5.6	158,545	9.5	59,148	3.4
Shrublands	993,972	1,019,419	566,159	25,448	2.6	– 453,260	– 44.5	– 427,812	– 43.0
Arboreal vegetation	10,871,839	3,166,463	1,163,557	– 7,705,375	– 70.9	– 2,002,906	– 63.3	– 9,708,281	– 89.3
Total ESV	13,652,516	5,962,681	3,653,048	– 7,689,835	– 56.3	– 2,309,633	– 38.7	– 9,999,468	– 73.2

reduced from US\$ 427,213 between 1989 and 2007 to US\$ 329,947 between 2007 and 2014. This reduction reflects the decrease in the average annual loss of ESV_{arboreal} in the latter period (US\$ 286,129) when compared with the former (US\$ 428,076). Changes in the extent of arboreal vegetation have the greatest influence on the total ESV values of any LULC; mainly because it is the local LULC class with the highest coefficient value (Table 2) and was by far the most dominant in 1989 (covering 46% of the study area). Therefore, the total ESV declined considerably as arboreal vegetation was lost to urban areas (which are considered to have negligible contribution to the total ESV for the purpose of this study).

Variations in the extent of other LULC classes have less effect on total ESV because their value coefficient (for example, shrublands), extent (for example, water) or change in area (grasslands) is relatively small. Although these other LULC had minor influences on variations in total ESV, their relative contribution to it increased through time (Figure 4). In the context of decapitalisation of total ESV, the relative importance of arboreal vegetation decreased from 80% to 32% between 1989 and 2014, whereas the share of other LULC increased, particularly grasslands, which represented 13% of the total ESV in 1989 and 50% in 2014 (Figure 4). In absolute values, ESV_{water} increased 346% tripling its contribution to total ESV from 0.2 to 2.7%, whereas $ESV_{\text{grasslands}}$ increased only 3% but its share of the total ESV increased four times. On the other hand, in 2014 $ESV_{\text{shrublands}}$ was only 57% of

its value in 1989 and its share of the total ESV more than doubled in the period, increasing from 7.3 to 15.5% (Figure 4).

Results indicate an overall reduction in the value of 18 out of the 19 ecosystem services included in the calculations of ESV between 1989 and 2014 (Table 6). The only exception was the cultural service 'aesthetic information', which increased by 3.4%. This increase mimics the variation in the extent of the grasslands LULC, the only class in the study area for which a coefficient value for the service 'aesthetic information' is provided by de Groot and others (2012). Between 1989 and 2014, nine ecosystem services had a reduction of over 85% in their value (Table 6): genetic resources, medicinal resources, air quality regulation, climate regulation, moderation of disturbance, water flow regulation, nutrient cycling, biological control and recreation. Considering only these nine services, the estimated loss reaches US\$ 8.93 million or 89% of the total ESV loss in the period. Six of these services (genetic resources, air quality regulation, moderation of disturbance, water flow regulation, nutrient cycling and biological control) account for 82% (US\$ 8.2 million) of the total ESV loss and this is solely due to losses of arboreal vegetation. See the online supplementary material (S2) for a breakdown of the values of each ecosystem service per LULC in 1989, 2007 and 2014.

In 1989, the ecosystem services climate regulation (US\$ 4.2 million), medicinal resources (US\$ 3.1 million) and recreation (US\$ 1.8 million) were the largest contributors to the total ESV (Table 6),

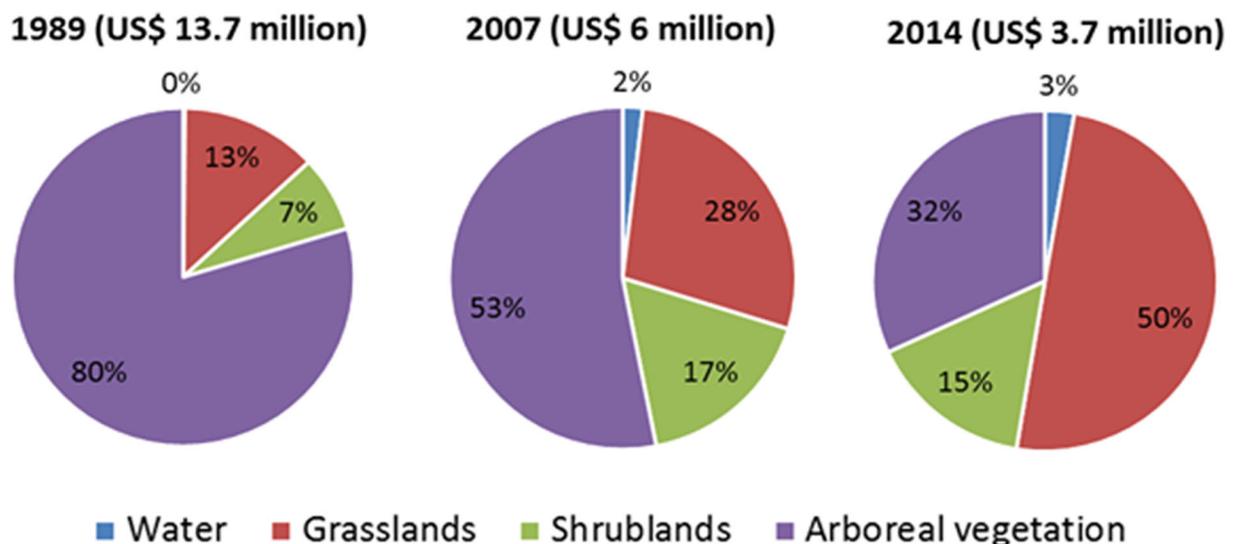


Figure 4. Relative contribution of the different LULC classes to the total ESV (in brackets) in the microwatershed Riacho das Piabas in 1989, 2007 and 2014.

Table 6. Estimated Value of the 19 Ecosystem Services (2017 US\$) in 1989, 2007 and 2014 Associated with the LULC Observed in the Microwatershed Riacho das Piabas (Brazil) and Their Respective Change Through Time (in US\$ and % of Initial Value)

Ecosystem service	Value (US\$)			Change (in US\$ and %)					
	1989	2007	2014	1989–2007	%	2007–2014	%	1989–2014	%
<i>Provisioning services</i>									
Food	1,178,496	847,491	822,041	– 331,005	– 28.1	– 25,449	– 3.0	– 356,455	– 30.2
Water	102,125	98,441	86,389	– 3,685	– 3.6	– 12,052	– 12.2	– 15,736	– 15.4
Raw materials	312,481	190,395	112,831	– 122,086	– 39.1	– 77,563	– 40.7	– 199,650	– 63.9
Genetic resources	26,851	7820	2874	– 19,030	– 70.9	– 4947	– 63.3	– 23,977	– 89.3
Medicinal resources	3,107,448	905,457	333,144	– 2,201,991	– 70.9	– 572,313	– 63.2	– 2,774,304	– 89.3
Ornamental resources	20,032	20,544	11,410	513	2.6	– 9135	– 44.5	– 8622	– 43.0
<i>Regulating services</i>									
Air quality regulation	4783	7218	2652	– 17,565	– 70.9	– 4566	– 63.3	– 22,130	– 89.3
Climate regulation	4,251,275	1,257,446	479,783	– 2,993,828	– 70.4	– 777,664	– 61.8	– 3,771,492	– 88.7
Moderation of disturbance	136,339	39,709	14,592	– 96,629	– 70.9	– 25,117	– 63.3	– 121,747	– 89.3
Water flow regulation	706,475	205,764	75,610	– 500,712	– 70.9	– 130,153	– 63.3	– 630,865	– 89.3
Waste treatment	59,448	51,992	53,323	– 7456	– 12.5	1331	2.6	– 6125	– 10.3
Erosion prevention	66,153	42,876	35,887	– 23,276	– 35.2	– 6990	– 16.3	– 30,266	– 45.8
Nutrient cycling	6204	1807	664	– 4397	– 70.9	– 1143	– 63.3	– 5540	– 89.3
Pollination	81,379	37,952	17,686	– 43,427	– 53.4	– 20,266	– 53.4	– 63,693	– 78.3
Biological control	22,715	6616	2431	– 16,099	– 70.9	– 4185	– 63.3	– 20,284	– 89.3
<i>Support services</i>									
Nursery service	829,854	826,826	457,389	– 3028	– 0.4	– 369,437	– 44.7	– 372,465	– 44.9
Genetic diversity	795,165	719,522	776,929	– 75,643	– 9.5	57,407	8.0	– 18,236	– 2.3
<i>Cultural services</i>									
Aesthetic information	102,589	96,810	106,029	– 5780	– 5.6	9219	9.5	3439	3.4
Recreation	1,822,707	597,996	261,386	– 1,224,711	– 67.2	– 336,610	– 56.3	– 1,561,321	– 85.7
Total ESV	13,652,516	5,962,681	3,653,048	– 7,689,835	– 56.3	– 2,309,633	– 38.7	– 9,999,468	– 73.2

corresponding to 67% of the total value. In 2014, these same three ecosystem services represented only 29% of the total ESV and have lost between 85.7% (recreation) and 89.3% (medicinal resources) of their value in 1989. In 2014, the three ecosystem services contributing the most to the total ESV in the study area were food (US\$ 0.82 million), genetic diversity (US\$ 0.78 million), and

climate regulation (US\$ 0.48 million), representing 57% of the total value.

Ecosystem Service Sensitivity Analysis

The sensitivity analysis considering $\pm 50\%$ of the coefficient values of each LULC used to estimate the total ESV resulted in coefficients of sensitivity

Table 7. Estimated Total Ecosystem Service Values Adjusted (ESVa in 2017 US\$) to $\pm 50\%$ of Ecosystem Service Valuation Coefficients (VC), the Relative Change in the ESVa Through Time (in % of Initial Value) and the Coefficient of Sensitivity (CS)

LULC classes VC	Total ESVa			%			CS		
	1989	2007	2014	1989– 2007	2007– 2014	1989– 2014	1989	2007	2014
Water + 50%	13,663,722	6,018,632	3,692,993	– 56.0	– 38.6	– 73.0	+ 0.002	+ 0.02	+ 0.03
Water – 50%	13,641,310	5,906,731	3,593,104	– 56.7	– 39.2	– 73.7	– 0.002	– 0.02	– 0.03
Grasslands + 50%	14,534,663	6,795,130	4,554,770	– 53.2	– 33.0	– 68.7	+ 0.129	+ 0.28	+ 0.5
Grasslands – 50%	12,770,369	5,130,233	2,731,327	– 59.8	– 46.8	– 78.6	– 0.129	– 0.28	– 0.5
Shrublands + 50%	14,149,502	6,472,391	3,921,128	– 54.3	– 39.4	– 72.3	+ 0.073	+ 0.17	+ 0.15
Shrublands – 50%	13,155,530	5,452,972	3,364,969	– 58.5	– 38.3	– 74.4	– 0.073	– 0.17	– 0.15
Arboreal vegetation + 50%	19,088,435	7,545,913	4,224,827	– 60.5	– 44.0	– 77.9	+ 0.796	+ 0.53	+ 0.32
Arboreal vegetation – 50%	8,216,597	4,379,450	3,061,270	– 46.7	– 30.1	– 62.7	– 0.796	– 0.53	– 0.32

(CS) lower than 1 (Table 7). The lowest and the highest CS were obtained for the water LULC (± 0.002 for 1989 and ± 0.03 for 2014) and arboreal vegetation (± 0.796 for 1989 and ± 0.32 for 2014), respectively. Hence, the total ESV estimated for the MWRP are relatively inelastic, that is, they show relative low sensitivity to variations of up to 50% in the value coefficients proposed by de Groot and others (2012). Thus, the estimated ESV is considered to be reasonably acceptable.

The relative differences between adjusted total ESV (ESVa, Table 7) and the total ESV calculated using the value coefficients of de Groot and others (2012) are lower for water LULC (largest differences are + 1% and – 1.7% of the ESV in 2014) and higher for arboreal vegetation (largest differences are around $\pm 39.9\%$ of the ESV in 1989). The CS is dependent on the LULC's value coefficient (VC) and extent; therefore, the largest differences between ESVa and total ESV occur due to changes in the LULC showing the highest VC and largest area. Consequently, a greater relative contribution of the LULC to the total ESV will result in a larger difference between ESVa and ESV. For example, the difference between the ESVa and ESV calculated considering $\pm 50\%$ VC_{grasslands} increased from $\pm 6.5\%$ in 1989 to $\pm 25\%$ in 2014, following an increase in its relative contribution to the total ESV from 13 to 50%, respectively. On the other hand, there was a reduction in the difference between the ESVa and ESV calculated considering $\pm 50\%$ VC_{arboreal} from $\pm 39.8\%$ in 1989 to $\pm 15.9\%$ in 2014, as its relative contribution to the total ESV decreased from 80 to 32%, respectively.

In terms of decapitalisation in the period 1989–2014, ESVa losses are within $\pm 2.3\%$ or less of the US\$ 10 million loss (Table 6) calculated using the

VC from de Groot and others (2012), except when considering $\pm 50\%$ VC_{arboreal}. In this case, the difference is around $\pm 48.6\%$. Despite this considerable difference in absolute values, the relative loss in ESVa between 1989 and 2014 is – 62.7% and – 77.9% for – 50% VC_{arboreal} and +50% VC_{arboreal}, respectively (Table 7), quite similar to the – 73.2% estimated without the adjustment (Table 6).

DISCUSSION

Implications of Ecosystem Services Loss to the Agreste Ecotone

The effects of rapid and disorderly urbanisation (commonly observed in developing countries) caused a reduction of vegetated areas (that is, shrublands and arboreal vegetation) from 60% to just less than 13% of the study area between 1989 and 2014. The serious effects of urbanisation and the decline in vegetated areas on water resources (Hümann and others 2011; Schneider and others 2012); microclimate regulation (Kalnay and Cai 2003; Schneider and others 2012) and fragmentation of habitats and biodiversity (Seto and others 2012; Newbold and others 2015) are well described in the literature and have critical implications in semiarid locations such as the agreste.

A global scale assessment indicates that the ecoregion where the MWRP is located is amongst the most vulnerable to climate change, as both low climate stability and degradation of vegetated areas are contributing to biodiversity and ecosystem functions loss (Watson and others 2013). According to Vieira and others (2015), 94% of the Northeast region of Brazil shows moderate to high susceptibility to desertification. Predictions indicate a large increase in temperature and reduction in precipi-

tation, with a trend for longer dry spells (Marengo and others 2017), increasing evaporation and the pressure on freshwater water resources (Gutiérrez and others 2014). The region is affected by serious water shortages caused by droughts, the last persisting since 2012 (Marengo and others 2017) and resulting in almost three years in which the domestic water supply was restricted to a few days per week and irrigation was prohibited in rural areas. Increased temperature and drought will intensify socioeconomic impacts related to water scarcity, already a major issue in the region (for example, Marengo and others 2017).

The climate trend of increased aridity can create conditions that are more favourable to the semiarid Caatinga vegetation, shifting or shrinking the ecotone boundaries likely to increase pressure on the humid Atlantic Forest. Such a shift could aggravate the decline in total ESV and lead to loss of ecosystem services critical to climate and water flow regulation (greatly provided by forests) and maintenance of biodiversity (for example, pollination and biological control). The great reduction in these important services (for example, 78% of pollination and 89% of climate regulation) observed in the 25 years analysed here enhances the ecotone's vulnerability to climate change, likely contributing to the desertification trend (for example, accelerating its effects and/or expanding the area affected). It is important to mention that in tropical areas, loss of naturally vegetated areas currently is, and is likely to be in the future, a greater threat to ecosystem degradation or species extinctions than climate change (Watson and others 2013), making nature conservation and restoration interventions ever more important.

Spatial Planning and Environmental Management Implications

The temporal changes in LULC driven by the urban development of Campina Grande clearly reflect the management (or political) priority for the potential socioeconomic benefits of the built environment without measuring the consequences of environmental degradation. As a result, the city became increasingly dependent on the provision of ecosystem services (for example, water supply and a great diversity of raw material and food) and the ecological integrity of the surrounding rural areas (Hails and Ormerod 2013). Predictions of climate change impacts in the agreste urge implementation of more sustainable management of water resources able to prevent LULC changes leading to further loss of key ecosystem services (for example,

climate regulation, moderation of disturbance and water flow regulation). Studies estimating ESV and their changes through time often claim that results provide useful evidence to guide policy and management decisions, but they rarely provide an indication on how or where this may be applicable. The results obtained here provide context to identify policy actions that can reduce or prevent impacts from LULC (Table 8).

Maintaining or creating green infrastructure (for example, interconnected green spaces and habitat restoration) is increasingly used as an adaptive management strategy to reduce vulnerability to climate extremes and other environmental disruptions (Green and others 2016; Watson and others 2013; Silva and others 2017). In urban and peri-urban areas, investment in interconnected green infrastructure, in private and public land, combined with restoration of vegetated areas (for example, riparian vegetation) and environmental education are a solution to maintain and enhance the functioning of ecosystems services. Such investment in green infrastructure can be justified by the socioeconomic benefits that can be attained through the positive impact on human health (for example, Tzoulas and others 2007) or disaster reduction (for example, Dhyani and others 2018).

Payment for Environmental Services (PES) should be considered as one investment option (for example, Balvanera and others 2012), which could benefit the MWRP. Law 10165 (of 25 November 2013) established a Policy for PES in the State of Paraíba (Brazil), but no schemes have been implemented within the MWRP so far, probably due to financial constraints, lack of public awareness and/or technical capacity. It is important to emphasise that decision-making, particularly that focused on specific ecosystem services and involving PES, should be based on data validated for local conditions (Nelson and others 2009). Implementation of ecosystem-based spatial planning (for example, Brussard and others 1998) through Strategic Environmental Assessment (for example, Rozas-Vásquez and others 2018) could help identifying strategic areas where PES and other mechanisms could help reduce the ecosystem services loss associated with urbanisation (for example, Dhyani and others 2018).

There is a growing interest in incorporating ecosystem-based management in spatial planning (for example, Balvanera and others 2012; Rozas-Vásquez and others 2017). Incorporating ecosystem services in spatial planning decisions is still limited by lack of clear guidelines, poor understanding of suitable governance mechanisms (for example,

Table 8. Policy Actions to Minimise the Relative Losses of ESV Due to LULC Changes

Policy action	Objectives	Where it is applicable
Stimulate nature conservation	Avoid the environmental and socioeconomic impacts of the resulting LULC changes	Where natural environment is still present or little altered
Regulate or control types and rates of occupation	Reduce ecosystem services losses and the magnitude of investment that may be required to replace them with alternative options	Where urbanisation is favoured or needed
Promote creation of habitats or guide nature restoration efforts	Gain or enhance the provision of ecosystem services most needed locally and reduce the costs associated with environmental degradation	Where human occupation has caused undesirable or unacceptable environmental or socioeconomic impacts, including where environmental compensation is required
Establishing payment for ecosystem services schemes	Prevent losses of ecosystem services more at risk or in greater demand by reducing detrimental, or promoting favourable, LULC changes	Where provision of ecosystem services can be maintained or enhanced through financial incentives to owners/managers of relevant areas

Strategic Environmental Assessment) and the need for methodological support (Mascarenhas and others 2015; Rozas-Vásquez and others 2017). However, these limitations can be overcome when practitioners and academics work together to develop methods that are robust but simple enough for practical applications. For example, the Secretary of Environment and Urbanism of the city of Natal (capital of Rio Grande do Norte state, Northeast Brazil) has identified the potential to enhance ecosystem services provision to support the creation of a corridor of green urban areas connecting two important areas of conservation (SEMURB 2017). The active engagement of practitioners from this Secretary with researchers (from Bournemouth University and the University Federal of Rio Grande do Norte) throughout the project *Valuation of Environmental Services Applied to Coastal Areas* (CAPES/PVE 88881.068064/2014-01) was key for the development and implementation of methods.

Caveats of the Study

The limitations of transferring a general unit coefficient to represent the local ESV are well described (for example, Nelson and others 2009; Richardson and others 2015; Rolfe and others 2015) and recognised here. Although the absolute ESV obtained from benefit transfer must be considered with caution, they are used here to give an indication of magnitudes of change, alerting to the ecosystem services most affected, which should be prioritised for local valuations that can inform policy and management decisions (Kreuter and others 2001; Tianhong and others 2010; Mendoza-

González and others 2012; Richardson and others 2015; Tolessa and others 2017; Yi and others 2017). Nevertheless, it is important to illustrate some of the issues related to the application of the benefit transfer method in the study area.

ESV data from ecotones, and semiarid conditions, are scarce (or non-existent) making difficult the application of coefficients that would be more representative of local biophysical settings. The characteristics of the study area are very geographically specific, a transition area between two unique and threatened biomes (the Caatinga and the Atlantic Forest). Using transfer values from areas of similar socioeconomic characteristics (for example, Latin America) may be more representative if they reflect similar biophysical conditions. Most data from Latin America were obtained from locations (for example, the Amazon) that are considerably different (both in biophysical conditions and in type and intensity of use) from the semiarid ecotone in the study area. Additionally, using data from Latin America only would limit both the number of 'biomes' and the ecosystem services that could be assessed. When data from equivalent sites are not available (as it is the case here), using generalised values are more likely to reduce biases (Richardson and others 2015; Crespin and Simonetti 2016).

As exemplified by the results in this study, ESV estimates can be greatly influenced by the dominant LULC class if: (a) its area is substantially larger than other LULC classes; (b) its area changes considerably through time; and (c) it shows the highest ESV_{LULC} . Although the sensitivity analysis indicates that the total ESV estimated for the MWRP is robust (that is, $CS < 1$), its value varies $\pm 40\%$ and

the decapitalisation $\pm 49\%$ when calculations consider adjustments of $\pm 50\%$ VC_{arboreal} . The ESV_{LULC} depends on each ecosystem service's monetary value and the number of services that are included in the calculations. The VC (from de Groot and others 2012) used in this study accounts for a different number of ecosystem services to calculate each ESV_{LULC} (17 for ESV_{arboreal} , 10 for $ESV_{\text{grasslands}}$, nine for $ESV_{\text{shrubland}}$ and four for ESV_{water} (Figure 2). The issue lies where ecosystem services exist but have been omitted due to the lack of valuation studies in some of the biomes/equivalent LULC, creating a discrepancy in the ESV_{LULC} used to assess impacts of LULC changes. This discrepancy and the indiscriminate use of generalised coefficients can lead to misconstrued knowledge and misinformed decision-making and have been identified as weaknesses of the benefit transfer method (for example, Nelson and others 2009).

The discrepancy in VC described in the previous paragraph will gradually be minimised as more valuation studies are added to global databases filling the existing gaps (Richardson and others 2015). Wherever possible, local valuations should be preferred, particularly on assessment of services considered locally valuable. In the study area, cultural services, such as aesthetic information, illustrate these limitations. Valuation of aesthetic information is only available for grasslands (Table 3), whereas very likely water would have the highest value if local valuation was available. The main course of Riacho das Piabas, particularly the Açude Velho, is an iconic landmark of Campina Grande (the image most used to reflect the city's identity in postcards, paintings and advertising) and the location selected to house sculptures, historic monuments and the Museum of Popular Art of Paraíba. Valuation of cultural ecosystem services (for example, aesthetic information) should be stimulated at the local level as their value is very site-specific and they are underrepresented in global databases.

Estimating ESV using a constant value coefficient irrespective of variations in quality and/or how the market value may have changed through time is a limitation of the benefit transfer method (Nelson and others 2009; Richardson and others 2015; Rolfe and others 2015). In the study area, changes in the ecosystem service provision of water (15.4% over the 25-year period) is likely to be underestimated due to two main reasons: (a) the large difference in the value of this service attributed to $LULC_{\text{water}}$ and the other LULC; and (b) the reduction in water quality through time in the study area. The service of water provision is 67 times

higher for the $LULC_{\text{water}}$ (US\$ 2203) than for the $LULC_{\text{arboreal}}$ (US\$ 33); therefore, small changes in the extent of $LULC_{\text{water}}$ can have an important relative contribution to this ecosystem service. Although there was an increase in $LULC_{\text{water}}$ area in the MWRP, in recent years part of it provides water for secondary use only (for example, irrigation of urban green areas) due to water quality issues. The importance of water supply, the environmental and socioeconomic costs of existing and future pressures (for example, climate change impacts) and the level of investment required in the MWRP urge valuation efforts at the local level.

CONCLUSION

This study adds to the current knowledge of impacts from LULC changes on the provision of ecosystem services by providing the first assessment of temporal changes in total ESV in an area of the Brazilian agreste, a tropical ecotone between the Atlantic Forest and the Caatinga biomes. The benefit transfer method was used to quantify ecosystem services losses between 1989, 2007 and 2014 due to urbanisation in the microwatershed Riacho das Piabas state of Paraíba, northeast Brazil. Considering the lack of local data, the benefit transfer method proved useful to identify: (a) the ecosystem services that were most affected by urbanisation; and (b) the local valuations that could contribute the most to support policy development and management decisions.

Urbanisation caused great reduction of vegetation cover which led to a generalised loss of 18 out of the 19 ecosystem services analysed and a reduction of 73.2% of the total estimated ESV. Considering the existing pressure on water resources and the regional trend towards desertification, urbanisation has likely increased the ecotone's vulnerability to climate change through losses of key ecosystem services (for example, biological control water flow and regulation of climate and water flow). The combination of urbanisation and climate change impacts may lead to the ecotone to shrink or shift boundaries favouring the semiarid Caatinga and increasing pressure on the humid Atlantic Forest. Better understanding of the LULC changes influencing water quality and availability and local valuation of related ecosystem services (for example, water provision and water flow regulation) would be most useful to guide policy and decision-making actions. Impacts of current environmental degradation and predicted climate change on the agreste ecotone urge the implementation of ecosystem-based spatial plan-

ning (for example, through Strategic Environmental Assessment) to prevent further ecosystem services loss. Investment should prioritise urban green infrastructure, restoration of natural habitat and payment for ecosystem services schemes more likely to promote the recovery of the identified key ecosystem services lost.

ACKNOWLEDGEMENTS

The authors Ferreira and Souza acknowledge the financial support to the PhD scholarship and Research Productivity award, respectively, received from the National Council for Scientific and Technologic Development (CNPq/Brazil). Esteves collaboration was facilitated by and resulted from work developed within the project VALSA (*Valuation of Environmental Services Applied to Coastal Areas* (funded by CAPES/PVE 88881.068064/2014-01).

OPEN ACCESS

This article is distributed under the terms of the Creative Commons Attribution 4.0 International License (<http://creativecommons.org/licenses/by/4.0/>), which permits unrestricted use, distribution, and reproduction in any medium, provided you give appropriate credit to the original author(s) and the source, provide a link to the Creative Commons license, and indicate if changes were made.

REFERENCES

- Agência Executiva de Gestão das Águas do Estado da Paraíba (AESPA). 2010. <http://www.aesa.pb.gov.br/>. Accessed 17 Oct 2017.
- Balvanera P, Uriarte M, Almeida-Leñero L, Altesor A, Declerck F, Gardner T, Hall J, Lara A, Laterra P, Peña-Claros M, Matos DMS, Vogl AL, Romero-Duque LP, Arreola LF, Caro-Borrero AP, Gallego F, Jain M, Little C, Xavier RO, Paruelo JM, Peinado JE, Poorter L, Ascarrunz N, Correa F, Cunha-Santino MB, Hernández-Sánchez A, Vallejos M. 2012. Ecosystem services research in Latin America: the state of the art. *Ecosyst Serv* 2:56–70.
- Beuchle R, Grecchi RC, Shimabukuro YE, Seliger R., Eva HD, Sano E, Achard F. 2015. Land cover changes in the Brazilian Cerrado and Caatinga biomes from 1990 to 2010 based on a systematic remote sensing sampling approach. *Appl Geogr* 58:116–27.
- Brussard PF, Reed JM, Tracy CR. 1998. Ecosystem management: What is it really? *Landsc Urban Plan* 40(9–20):1998.
- Bueno ML, Rezende VL, Pontara V, Oliveira-Filho AT. 2017. Floristic distributional patterns in a diverse ecotonal area in South America. *Plant Ecol* 218:1171–86.
- Câmara E. 2006. Os alicerces de Campina Grande: esboço histórico-social do povoado e da vila (1697–1864). 3rd edn. Campina Grande: Editora Caravela.
- Costanza R, de Groot R, Sutton P, van der Ploeg S, Anderson SJ, Kubiszewski J, Farber S, Turner RK. 2014. Changes in the global value of ecosystem services. *Global Environ Chang* 26:152–8.
- Crespin SJ, Simonetti JA. 2016. Loss of ecosystem services and the decapitalization of nature in El Salvador. *Ecosyst Serv* 17:5–13.
- de Groot RS, Brander L, Van der Ploeg S, Costanza R, Bernard F, Braat L, Christie M, Crossman N, Ghermandi A, Hein L, Hussain S, Kumar P, McVittie L, Portela R, Rodriguez LC, Brinkm P, Van Beukering P. 2012. Global estimates of the value of ecosystems and their services in monetary units. *Ecosyst Serv* 1:50–61.
- Dhyani S, Lahoti S, Khare S, Pujari P, Verma P. 2018. Ecosystem based disaster risk reduction approaches (EbDRR) as a prerequisite for inclusive urban transformation of Nagpur City, India. *International Journal of Disaster Risk Reduction*, <http://doi.org/10.1016/j.ijdr.2018.01.018>.
- Elmqvist T, Fragkias M, Goodness J, Güneralp B, Marcotullio PJ, McDonald RI, Parnell S, Schewenius M, Sendstad M, Seto KC, Wilkinson C. 2013. Urbanisation, biodiversity and ecosystem services: challenges and opportunities. Dordrecht: Springer. p 775p.
- Estoque RC, Murayama Y. 2013. Landscape pattern and ecosystem service value changes: implications for environmental sustainability planning for the rapidly urbanizing summer capital of the Philippines. *Landsc Urban Plan* 116:60–72.
- Fragkias M, Güneralp B, Seto KC, Goodness J. 2013. A synthesis of global urbanisation projections. Elmqvist T, Fragkias M, Goodness J, Güneralp B, Marcotullio PJ, McDonald RI, Parnell S, Schewenius M, Sendstad M, Seto KC, Wilkinson C, Eds. *Urbanisation, biodiversity and ecosystem services: challenges and opportunities*. Dordrecht: Springer, pp 409–35.
- Green TL, Kronenberg J, Andersson E, Elmqvist T, Gómez-Baggethun E. 2016. Insurance value of green infrastructure in and around cities. *Ecosystems* 19:1051–63.
- Groffman PM, Cadenasso ML, Cavender-Bares J, Childers DL, Grimm NB, Grove JM, Hobbie SE, Hutry LR, Jenerette GD, McPhearson T, Pataki DE, Pickett STA, Pouyat RV, Rosi-Marshall E, Ruddell BL. 2017. Moving towards a new urban systems science. *Ecosystems* 20:38–43.
- Gutiérrez APA, Engle NL, De Nys E, Molejon C, Martins ES. 2014. Drought preparedness in Brazil. *Weather Clim Extrem* 3:95–106.
- Hails RS, Ormerod SJ. 2013. Ecological science for ecosystem services and the stewardship of natural capital. *J Appl Ecol* 50:807–11.
- Hu H, Liu W, Cao M. 2008. Impact of land use and land cover changes on ecosystem services in Menglun, Xishuangbanna, Southwest China. *Environ Monit Assess* 146:146–56.
- Hümann M, Schüler G, Müller C, Schneider R, Johst M, Caspari T. 2011. Identification of runoff processes—the impact of different forest types and soil properties on runoff formation and floods. *J Hydrol* 409:637–49.
- Jensen JR. 2006. Remote sensing of the environment: an earth resource perspective. 2nd edn. Saddle River: Pearson Prentice Hall. p 608p.
- Kalnay E, Cai M. 2003. Impact of urbanization and land-use change on climate. *Nature* 423:528–31.
- Kindu M, Schneider T, Teketay D, Knoke T. 2016. Changes of ecosystem service values in response to land use/land cover dynamics in Munessa-Shashemene landscape of the Ethiopian highlands. *Sci Total Environ* 547:137–47.

- Kottek M, Grieser J, Beck C, Rudolf B, Rubel F. 2006. World Map of the Köppen–Geiger climate classification updated. *Meteorologische Zeitschrift* 15:259–63.
- Kreuter UP, Harris HG, Matlock MD, Lacey RE. 2001. Change in ecosystem service values in the San Antonio area, Texas. *Ecol Econ* 39:333–46.
- Li RQ, Dong M, Cui JY, Zhang LL, Cui QG, He WM. 2007. Quantification of the impact of land-use changes on ecosystem services: a case study in Pingbian County, China. *Environ Monit Assess* 128:503–10.
- Macedo MJH, Guedes RVS, Sousa FAZ. 2011. Monitoramento e intensidade das secas e chuvas na cidade de Campina Grande/PB. *Revista Brasileira de Climatologia* 8:105–17.
- Mansfield E. 1985. *Microeconomics: theory and applications*. 5th edn. New York: W.W. Norton and Company.
- Marengo JA, Torres RR, Alvez LM. 2017. Drought in Northeast Brazil—past, present, and future. *Theor Appl Climatol* 124(3–4):1189–200.
- Mascarenhas A, Ramos TB, Haase D, Santos R. 2015. Ecosystem services in spatial planning and strategic environmental assessment—a European and Portuguese profile. *Land Use Policy* 48:158–69.
- McDonough K, Hutchinson S, Moore T, Hutchinson JMS. 2017. Analysis of publication trends in ecosystem services research. *Ecosyst Serv* 25:82–8.
- McKinney ML. 2002. Urbanization, biodiversity, and conservation. *BioScience* 52:883–90.
- Mendoza-González G, Martínez ML, Lithgow D, Pérez-Maqueo O, Simonin P. 2012. Land use change and its effects on the value of ecosystem services along the coast of the Gulf of Mexico. *Ecol Econ* 82:23–32.
- Millennium Ecosystem Assessment (MEA). 2005. *Ecosystems and human well-being: biodiversity synthesis*. Washington: World Resources Institute. p 408p.
- Moro MF, Lughadha EN, Araújo FS, Martins FR. 2016. Phyto-geographical Metaanalysis of the Semiarid Caatinga Domain in Brazil. *Botanical Review* 82:91–148.
- Nelson E, Mendoza G, Regetz J, Polasky S, Tallis H, Cameron DR, Chan KMA, Daily GC, Goldstein J, Kareiva PM, Lonsdorf E, Naidoo R, Ricketts TH, Shaw MR. 2009. Modeling multiple ecosystem services, biodiversity conservation, commodity production and tradeoffs at landscape scales. *Front Ecol Environ* 7:4–11.
- Newbold T, Hudson LN, Hill SLL, Contu S, Lysenko I, Senior RA, Boeger L, Bennett DJ, Choimes A, Collen B, Day J, Palma A, Diaz S, Echeverria-Londono S, Edgar MJ, Feldman A, Garon M, Harrison MLK, Alhusseni T, Ingram DJ, Itescu Y, Kattge J, Kemp V, Kirkpatrick L, Kleyer M, Correia DLP, Martin CD, Meiri S, Novosolov M, Pan Y, Phillips HRP, Purves DW, Robinson A, Simpson J, Tuck SL, Weiher E, White HJ, Ewers RM, Mace GM, Scharlemann JPW, Purvis A. 2015. Global effects of land use on local terrestrial biodiversity. *Nature* 520:45–50.
- Richardson L, Loomis J, Kroeger T, Casey F. 2015. The role of benefit transfer in ecosystem service valuation. *Ecol Econ* 115:51–8.
- Rolfe J, Windle J, Johnston RJ. 2015. Applying benefit transfer with limited data: unit value transfers in practice. In: Johnston RJ, Rolfe J, Rosenberger RS, Brouwer R, Eds. *Benefit transfer of environmental and resource values: a guide for researchers and practitioners*. Dordrecht: Springer. p 141–62.
- Rouse JW, Haas RH, Schell JA, Deering DW. 1974. Monitoring vegetation systems in the great plains with ERTS. *Proceedings, Third Earth Resources Technology Satellite, in 1 Symposium, Greenbelt: NASA SP*. pp 3010–17.
- Rozas-Vásquez D, Fürst C, Geneletti D, Almendra O. 2018. Integration of ecosystem services in strategic environmental assessment across spatial planning scales. *Land Use Policy* 71:303–10.
- Rozas-Vásquez D, Fürst C, Geneletti D, Muñoz F. 2017. Multi-actor involvement for integrating ecosystem services in strategic environmental assessment of spatial plans. *Environ Impact Assess Rev* 62:135–46.
- Scarano FR. 2009. Plant communities at the periphery of the Atlantic rain forest: rare-species bias and its risks for conservation. *Biol Conserv* 142:1201–8.
- Schneider A, Logan KE, Kucharik CJ. 2012. Impacts of Urbanisation on ecosystem goods and services in the U.S. Corn Belt. *Ecosystems* 15:519–41.
- SEMURB (Secretary of Environment and Urbanism of the city of Natal). 2017. *Projeto Trampolim Ecológico*. Report from the Secretaria Adjunta de Informação, Planejamento Urbano e Gestão Ambiental, Setor de Áreas Verdes, Natal/RN: SEMURB (June 2017), 18p.
- Seto KC, Fragkias M, Güneralp B, Reilly MK. 2011. A meta-analysis of global urban land expansion. *PLoS ONE* 6(8):e23777.
- Seto KC, Güneralp B, Hutyrá LR. 2012. Global forecasts of urban expansion to 2030 and direct impacts on biodiversity and carbon pools. *PNAS* 109(40):16083–8.
- Silva R, Lithgow D, Esteves LS, Martinez ML, Moreno-Casasola P, Martell R, Pereira P, Mendoza E, Campos-Cascaredo A, Winckler Grez P, Osorio AF, Osorio-Cano JD, Rivillas GD. 2017. Coastal risk mitigation by green infrastructure in Latin America. *Marit Eng* 170:39–54.
- Solecki W, Seto KC, Marcotullio PJ. 2013. It's time for an urbanisation science. *Environ Sci Policy Sustain Dev* 55:12–17.
- Tianhong L, Wenkai L, Zhenghan Q. 2010. Variations in ecosystem service value in response to land use changes in Shenzhen. *Ecol Econ* 69:1427–35.
- Tolessa T, Senbeta F, Kidane M. 2017. The impact of land use/land cover change on ecosystem services in the central highlands of Ethiopia. *Ecosyst Serv* 23:47–54.
- Tzoulas K, Korpela K, Venn S, Yli-Pelkonen V, Kaźmierczak A, Niemela J, James P. 2007. Promoting ecosystem and human health in urban areas using Green Infrastructure: a literature review. *Landsc Urban Plan* 81(3):167–78.
- United States Geological Survey (USGS). 2014. *Global Visualization Viewer*. Available in: <http://glovis.usgs.gov/>. Accessed 17 Sept 2016.
- United States Geological Survey (USGS). 2016. *Using the USGS Landsat 8 Product*. Available in: <http://landsat.usgs.gov/using-usgs-landsat-8-product/>. Accessed 3 Dec 2017.
- Van der Ploeg S, de Groot RS. 2010. *The TEEB valuation database—a searchable database of 1310 estimates of monetary values of ecosystem services*. Wageningen, The Netherlands: Foundation for Sustainable Development.
- Vieira RMSP, Tomasella J, Alvalá RCS, Sestini MF, Affonso AG, Rodriguez DA, Barbosa AA, Cunha APMA, Valles GF, Crepani E, de Oliveira SBP, de Souza MSB, Calil PM, de Carvalho MA, Valeriano DM, Campello FCB, Santana MO. 2015. Identifying

- areas susceptible to desertification in the Brazilian northeast. *Solid Earth* 6:347–60.
- Waters R, Allen R, Tasumi M, Trezza R, Bastiaanssen W. 2002. SEBAL Surface Energy Balance for Land. Idaho implementation: Advanced Training and User's Manual. Version 1.0. 97p.
- Watson JEM, Iwamura T, Butt N. 2013. Mapping vulnerability and conservation adaptation strategies under climate change. *Nat Clim Change* 3:989–94.
- Yi H, Güneralp B, Filippi AM, Kreuter UP, Güneralp İ. 2017. Impacts of land change on ecosystem services in the San Antonio River Basin, Texas, from 1984 to 2010. *Ecol Econ* 135:125–35.