

Carbon lost and carbon gained: a study of vegetation and carbon trade-offs among diverse land uses in Phoenix, Arizona

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Abstract. Human modification and management of urban landscapes drastically alters vegetation and soils, thereby altering carbon (C) storage and rates of net primary productivity (NPP). Complex social and ecological processes drive vegetation cover in cities, leading to heterogeneity in C dynamics depending on regional climate, land use, and land cover. Recent work has demonstrated homogenization in ecological processes within human-dominated landscapes (the urban convergence hypothesis) in soils and biotic communities. However, a lack of information on vegetation in arid land cities has hindered an understanding of potential C storage and NPP convergence across a diversity of ecosystem types. We estimated C storage and NPP of trees and shrubs for six different land-use types in the arid metropolis of Phoenix, Arizona, USA, and compared those results to native desert ecosystems, as well as other urban and natural systems around the world. Results from Phoenix do not support the convergence hypothesis. In particular, C storage in urban trees and shrubs was 42% of that found in desert vegetation, while NPP was only 20% of the total NPP estimated for comparable natural ecosystems. Furthermore, the overall estimates of C storage and NPP associated with urban trees in the CAP ecosystem were much lower (8–63%) than the other cities included in this analysis. We also found that C storage (175.25–388.94 g/m²) and NPP (8.07–15.99 g·m⁻²·yr⁻¹) were dominated by trees in the urban residential land uses, while in the desert, shrubs were the primary source for pools (183.65 g/m²) and fluxes (6.51 g·m⁻²·yr⁻¹). These results indicate a trade-off between shrubs and trees in arid ecosystems, with shrubs playing a major role in overall C storage and NPP in deserts and trees serving as the dominant C pool in cities. Our research supports current literature that calls for the development of spatially explicit and standardized methods for analyzing C dynamics associated with vegetation in urbanizing areas.

Key words: carbon storage; desert city; land-use change; net primary productivity; urban forest; urbanization.

INTRODUCTION

If the drivers of urbanization continue into the future, the world may see a fivefold increase in urban land cover in the next 50 years (Angel et al. 2011, Seto et al. 2012). Land-use changes associated with the urbanization process have dramatic effects on ecological structure and function, influencing global biogeochemical cycles (Vitousek et al. 1997, Kaye et al. 2006). Due to the links between modifications to the carbon (C) cycle and global climate change, C dynamics have become a centerpiece in discussions of the impacts of cities (Grimm et al. 2008a, Gurney et al. 2009, Hutrya et al. 2014). Although vegetation in urban areas is not likely to ever offset the CO₂ emissions in cities run on a fossil fuel-based economy (Pataki et al. 2006), mitigation of CO₂ emissions is considered a high priority for local governments around the world, as seen by the proliferation of municipal plans for achieving sustainability (Jo and

McPherson 1995, Pincetl and Gearin 2005, Romero-Lankao et al. 2014). Yet, there are still substantial holes in our understanding of C pools and fluxes associated with cities (Pataki et al. 2006, Hutrya et al. 2014).

Motivated by interests in quantifying the many ecosystem services provided by trees and other green infrastructure in human landscapes, researchers have explored the direct and indirect benefits of the urban forest on local C budgets as outcomes of tree-planting programs in cities (McPherson and Simpson 2001, Nowak and Crane 2002, McPherson et al. 2005, McPherson and Kendall 2014). Urban trees have been shown to reduce CO₂ emissions indirectly by reducing household energy use through shading and blocking wind (McPherson and Simpson 2001, Nowak and Crane 2002, Jenerette et al. 2011, Nelson et al. 2012). However, some urban forests also make unexpected and potentially large contributions to local C budgets based on their ability to sequester and store C in their biomass (e.g., Seattle, Washington, USA and Halifax, Nova Scotia, Canada; Freedman et al. 1996, Hutrya et al. 2011, Weissert et al. 2014).

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Arid and semiarid landscapes cover a third of Earth's terrestrial surface and support expansive agriculture, large cities, and rapid rates of urbanization comparable to other areas of the world (Gober 2010). Despite intensive water consumption and management required by urban forests, cities in water-limited ecosystems continue to establish vigorous tree planting programs to take advantage of the numerous social-ecological benefits conferred by trees (Pataki et al. 2011). Consequently, urbanization in water-limited ecosystems has the potential to influence regional C budgets by creating "hot spots" of vegetation biomass and growth in an otherwise low-productivity environment (Buyantuyev and Wu 2012, Zhang et al. 2013).

The process of urbanization, however, generally is preceded by vegetation clearing in the initial phases of development, then is followed by the replanting and maintenance of trees and lawns in urban landscapes. This practice results in a net loss of C storage in forested cities of mesic climates (Boyle and Lavkulich 1997). In contrast, a net increase in C storage has been predicted for aridland cities, where intensive irrigation and management of the urban forest increases vegetative cover compared to native shrubland (Buyantuyev and Wu 2009). These different analyses of tree cover superficially support the urban homogenization hypothesis: the process by which cities are thought to become more like one another, and less like their surroundings, over time (Pouyat et al. 2006, Grimm et al. 2008b, Groffman et al. 2014).

Measuring and comparing C dynamics associated with vegetation both within and across cities remains challenging. To date, most research is based on plot-level data that are scaled up to local land uses and then to the broader region (Hutyra et al. 2011, Raciti et al. 2012). This methodology is primarily a legacy of how ecologists studied natural ecosystems for decades, but does not allow urban ecologists to capture the heterogeneity that exists in human-dominated environments (e.g., Luck and Wu 2002). For example, although residential areas are structured differently than industrial or commercial lands, significant variation also exists within the residential land-use type that can lead to important differences in the amount and quality of vegetated surfaces (Cadenasso et al. 2007, Walker et al. 2009). In urban ecosystems, C pools and fluxes vary spatially in response to such heterogeneity across a multitude of land uses and several different scales; therefore, it is not surprising that some of the most recent papers on urban C dynamics call for a standardization of methods so that scientists can effectively describe heterogeneity across a city, as well as support applicable comparisons among multiple cities (Pataki et al. 2006, Raciti et al. 2012, Davies et al. 2013, Hutyra et al. 2014, Romero-Lankao et al. 2014).

Statistical techniques for spatial scaling are one way to effectively capture heterogeneity in urban ecosystems while utilizing plot-based data (Majumdar et al. 2008). Kaye et al. (2008) used hierarchical Bayesian scaling to spatially represent soil nutrient pools across central

Arizona and metropolitan Phoenix, yielding regional estimates of soil C, nitrogen, and phosphorus for a variety of land uses that were more accurate than estimates derived from simple up-scaling methods. Although this kind of statistical model is appropriate for understanding soil processes, it is unknown whether these techniques can also be implemented to understand vegetation-related C dynamics.

Recent attempts at standardization have used an increasingly popular tool, the UFORE (Urban FOREst Effects) model (Nowak and Crane 2000), which is distributed as part of the free and easily accessible i-Tree Suite (i-Tree Canopy 2014) marketed by the U.S. Forest Service in partnership with influential urban forestry and arboriculture companies in the United States. Simplifying the process of implementing and analyzing urban forest structure and function has produced a proliferation of studies on ecosystem services associated with urban trees. However, arid and semiarid ecosystems are typically underrepresented in urban ecological data sets and models, despite their global extent. Notably, the UFORE model does not estimate C storage and NPP for shrubs because they are considered to be a very small pool compared to trees and soils in most ecosystems studied (Nowak 1994). Yet the natural woody vegetation in water-limited ecosystems tends to be dominated by shrubs or shrubby trees, and therefore shrubs could make up a relatively large portion of biomass in arid regions, both within and outside of city boundaries.

The goals of this study were to (1) compare C storage and NPP in urbanized and rural lands in an arid, shrub-dominated ecosystem relative to estimates from other cities across diverse biomes; (2) analyze vegetation C storage across different land uses in an arid ecosystem relative to soil C estimates from the same urban area; and (3) test the strength of spatial scaling techniques with plot-level vegetation data across heterogeneous land cover types typical of urban ecosystems. This research tests the hypothesis that in water-limited ecosystems urbanization increases, rather than decreases, C storage, and NPP. Furthermore, as comparative approaches to understanding urban ecosystems are a priority for social-ecological analyses (Grimm et al. 2008b, Groffman et al. 2014), our study informs the ongoing discussion about best practices for quantifying and comparing ecosystem fluxes across heterogeneous urban areas.

METHODS

Study site

The Phoenix metropolitan area (hereafter, Phoenix), also known as the Phoenix–Mesa–Glendale Metropolitan Statistical Area (MSA), comprises portions of Maricopa and Pinal Counties and encompasses the Central Arizona–Phoenix (CAP) Long-Term Ecological Research (LTER) study area. The metro population grew from 3.3 million to 4.2 million people between 2000

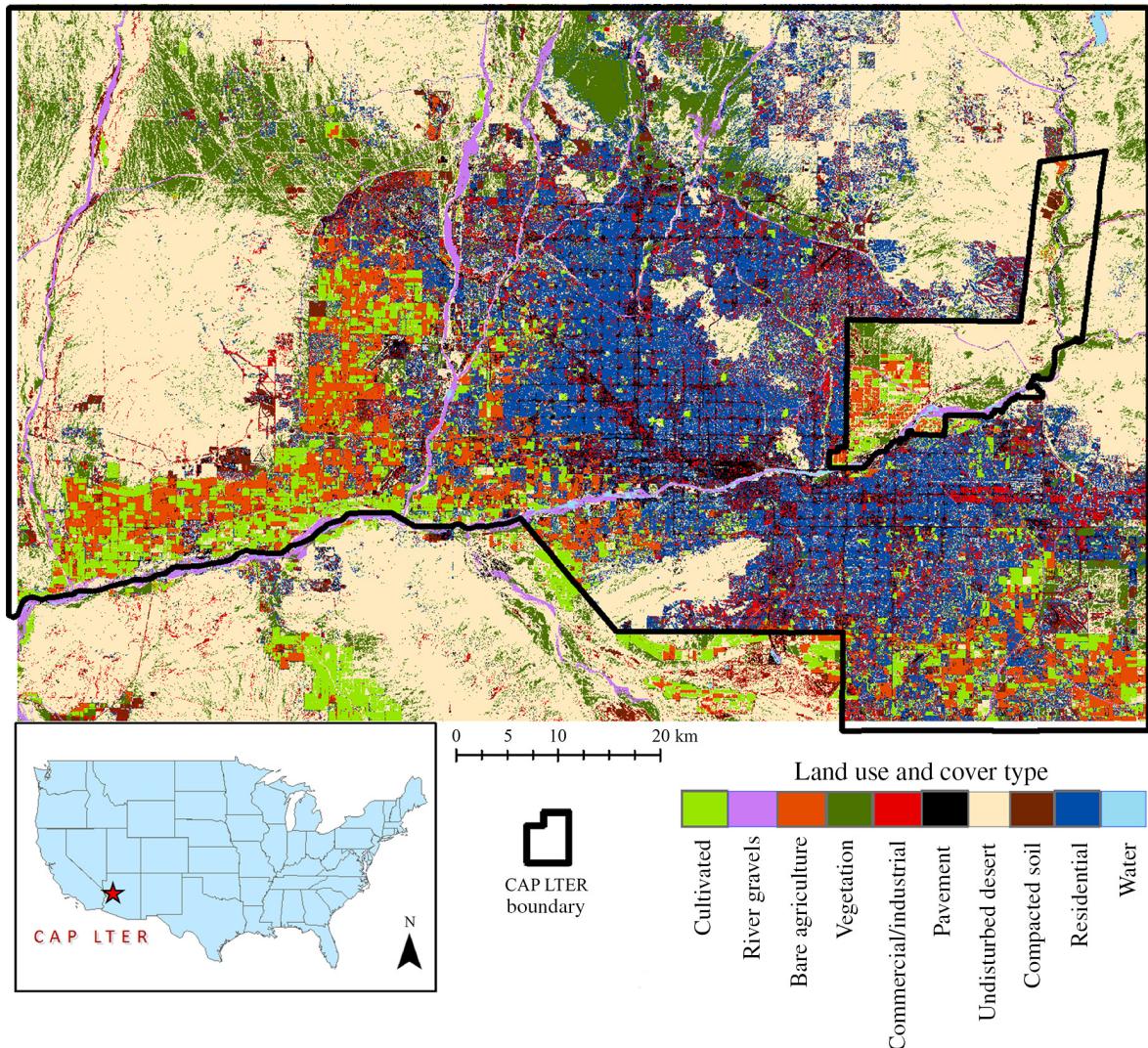


FIG. 1. Land uses in the Central Arizona Phoenix Long-Term Ecological Research area (CAP LTER; land use/cover derived by A. Buyantuyev using Landsat ETM+ images from Spring 2000). [Color figure can be viewed at wileyonlinelibrary.com]

and 2010 (U.S. Census Bureau, 2010), and Phoenix has been one of the fastest growing metro areas in the United States since the 1950s (Gober 2006). Average daily temperature ranges from 15° to 30°C, with an annual rainfall of 193 mm. Although lower Sonoran Desert scrub is the native vegetation type, water and nutrient inputs associated with agricultural, urban, and residential land uses allow for a more extensive coverage of crops, forests, and other vegetation than would typically be supported in this climate zone.

Our study area comprises 7962 km² of the Phoenix MSA, encompassing the city of Phoenix and 24 additional municipalities including five of the six largest cities in Arizona, a matrix of sprawling urban and suburban lands, and a large portion of the surrounding desert and agricultural areas (Moeller 2000; Fig. 1). Here the CAP LTER program has been conducting a longitudinal

survey focusing on land use and land cover, vegetation, and soil characteristics, and other supporting environmental and social variables (Hope et al. 2005a, b). Since 2000, the CAP LTER has collected ecological data every five years at 200 points randomly located within a dual-density grid across the CAP ecosystem. Standard ecological field methods used by the UFORE and other ecological models were employed in this survey to quantify patterns of plant and soil characteristics (Hope et al. 2003, 2005c, Oleson et al. 2006, Zhu et al. 2006, Walker et al. 2009). In particular, Kaye et al. (2008) conducted an analysis on C and other nutrients stored in soil, which will provide comparable data for our results on C dynamics in vegetation.

Since we used the same basic data set that was analyzed in the studies cited in the last paragraph, a more detailed description of the survey design can be found in those

publications. In brief, the 200+ plots (204–207 plots depending on the year) were spatially dispersed according to a random, dual-density, tessellation-stratified design. A grid with 4 km × 4 km cells ensured the plots were distributed across the region. A 30 m × 30 m plot was randomly placed in more than 200 of the grid cells, and the plot location was maintained among all sample years. Plot density was higher in the urbanized area (1 plot/grid cell) compared to the other land uses (desert and agriculture; 1 plot/3 grid cells) to account for the potentially increased heterogeneity in land cover and ecosystem structure and function within the urbanized area.

Plot-level vegetation measurements: trees and shrubs

All vegetation in each 30 m × 30 m plot was identified according to species or genus, mapped, and measured. Tree stems were measured at breast height (DBH, 1.37 m) and/or basal height, depending on their structure and size (i.e., large, single-stemmed or shrubby multiple-stemmed structure). The field team also measured tree height, bole height, crown width, the percentage of the canopy that was missing (from an idealized tree form), tree condition (i.e., poor, fair, good, and excellent based on the amount of dieback), and whether or not the tree was located on a street.

The number of stems and their basal diameters were measured for most shrubs in each plot. In some cases, a few shrubs per plot were not measured directly but were indicated to be similar to other shrubs that were measured in the same plot, a technique used to save time in the field. Shrub height was indicated for every shrub, as well as crown width, determined as the mean of width in north/south and east/west cardinal directions.

Carbon storage and net primary productivity

We used the vegetation variables in the 2005 data set to estimate C storage and NPP associated with both trees and shrubs. Tree data were compiled and analyzed using the UFORE model (now called i-Tree Eco). Detailed methods for the model can be found online (i-Tree Canopy 2014). In general, biomass equations were compiled from the literature (Nowak 1994, Nowak et al. 2002), and whole-tree biomass was calculated using a root to shoot ratio of 0.26 (Cairns et al. 1997). Dry biomass was converted to C storage by multiplying by 0.5 (Forest Products Lab 1952, Chow and Rolfe 1989). It is assumed in the UFORE model that open-grown trees have lower biomass than trees growing in forests (Nowak 1994, i-Tree Canopy 2014); thus the model reduces all biomass estimates for urban trees by 0.8 to account for this difference. McHale et al. (2009) determined that this model assumption is not valid, since most of the species they analyzed exhibited similar or greater biomass when open grown compared to the allometric equations developed for traditional forest trees. Therefore, we added an extra 20% to the C storage values estimated by the UFORE model to erase the 0.8 conversion factor implemented in the model.

Tree NPP, also called annual sequestration in the UFORE model, was determined using DBH and height–growth equations derived from data in northern climate zones (Fleming 1988, Nowak 1994). These growth estimates are likely conservative for the Phoenix area because of its higher mean temperature and longer growing season, but locally based equations were not available. Growth in the Sonoran Desert is likely limited both by available water and nutrients, but in urban areas, additional water is usually supplied and trees are often fertilized. We therefore expect tree growth to vary according to human management decisions, but to largely fall above these UFORE estimates.

Finally, the UFORE model is typically used to analyze 0.1 acre (4,046 m²) circular plots in designated land-use types and then to estimate tree benefits for each land use. Since our plots were 30 m × 30 m, the tree data were processed in the model and reported on a per tree basis, rather than for each plot/land use. We then calculated the total C stored and sequestered by trees in each plot, categorized our plots by land use, and estimated mean C stored and sequestered by trees in each land-use type across the region.

The UFORE model does not estimate C storage and NPP for shrubs because they are considered to be a very small pool compared to trees and soils in most ecosystems studied (Nowak 1994). However, arid and semiarid ecosystems are typically underrepresented in ecological data sets and models despite their global extent (e.g., Davies et al. 2013). Because water-limited ecosystems are often dominated by shrubby vegetation, we used equations from the literature to estimate the shrub C contribution to total change in C storage that occurs with urbanization. To do this, we compiled data from all publications that estimated biomass of shrubs in arid or semiarid areas. Finding usable equations to estimate shrub biomass is challenging because studies tend to use a variety of different predictors (i.e., height, short and long canopy width, canopy area, circumference of crown area, basal area, diameter of the longest stem, number of stems, plant volume, etc.). Overall, we found more than 30 articles on shrub biomass predictions, but only eight met our criteria: the allometric equations had to have (1) usable predictors and (2) been developed for semiarid or arid systems, or (3) utilized species that were found in the CAP ecosystem. Using 16 equations found in the eight publications (Frandsen 1983, Rosenschein et al. 1999, Tietema and Hall 1999, Hierro et al. 2000, Sah et al. 2004, Northup et al. 2005, Allen et al. 2008, Browning et al. 2008), we created a data set predicting biomass from a range of crown areas seen in the field. We developed a general regression model that predicted an average amount of biomass for each DBH and applied this regression to all shrub species found in the CAP plots

$$Y = 0.73A^{1.3204}$$

where Y is shrub biomass (g/m²) and A is crown area (m²). To estimate the amount of C stored in shrub biomass, we multiplied by 0.47 (Browning et al. 2008).

Shrub NPP was calculated using an equation developed by Chew and Chew (1965)

$$\text{NPP} = 0.54S$$

where NPP is in the units of $\text{kg}\cdot\text{ha}^{-1}\cdot\text{yr}^{-1}$, S is shrub cover in m^2/ha , and 0.54 is the average net annual production per m^2 cover of *Larrea*, *Flourensia*, and *Parthenium*. Annual dry mass accumulation (kg) of all shrubs in each plot was calculated from total shrub cover within the plot boundaries. Again, these estimates were then converted to C by multiplying by 0.47 (Browning et al. 2008). Finally, we divided the estimates of aboveground NPP by 0.76 to convert to total NPP ($\text{g C}\cdot\text{m}^{-2}\cdot\text{yr}^{-1}$), accounting for belowground NPP (Chew and Chew 1965, Shen et al. 2005).

Land use and cover

We analyzed data from 207 sample points that were located in a variety of land uses classified according to definitions provided by the Maricopa Association of Governments (1997): desert (73 points), agriculture (16), urban non-residential (41), urban residential land with mesic landscaping (23), urban residential land with xeric landscaping (22), and mixed land use (11). We evaluated land use and cover from aerial photographs for 2005 and noted any significant land-use changes compared to year 2000.

We utilized an aggregated land-cover classification map initially developed from Landsat ETM+ images acquired in year 2000 and employed by Kaye et al. (2008) to calculate the storage of soil C associated with each land use (Fig. 1). The land-cover/land-use classes in this map included urban residential land with xeric landscaping (1607 km^2), urban residential land with mesic landscaping (175 km^2), urban industrial, commercial, and transportation (176 km^2), water and high-density riparian vegetation (169 km^2), Sonoran Desert (4697 km^2), agricultural land (1130 km^2), and unclassified (9 km^2).

We scaled median plot-level C data to regional estimates of vegetation C storage and NPP. For this up-scaling, we needed to resolve two issues. First, the land-cover classification map did not include the "mixed" land-use category associated with our plot-level land-use definitions. Second, although the map included both water and riparian vegetation land uses, none of our plots represented those land-use categories. Therefore, for this analysis we only included urban xeric residential landscapes, urban mesic residential landscapes, and urban non-residential landscapes, as well as the natural desert and agricultural land uses.

Statistical analysis: testing for differences among land uses

We tested for statistical differences among estimates of C and NPP in trees and shrubs for all land uses using JMP (Version 9, SAS Institute, Cary, North Carolina,

USA). The data were not normally distributed so we used the Wilcoxon/Kruskal-Wallis Test (rank sums). C storage and NPP for both trees and shrubs differed significantly among land-use categories ($P < 0.0001$ for all analyses). We therefore conducted nonparametric comparisons for each pair of land-use types using the Wilcoxon method and the Steel-Dwass method, the latter test taking into account multiple comparisons. The Steel-Dwass method found fewer significant differences than the Wilcoxon, as expected for this more conservative test.

We calculated sampling uncertainties around the median C storage and NPP estimates using a bootstrapping analysis (Efron and Tibshirani 1994). To develop 95% confidence limits around each median, one thousand bootstrap samples were drawn with replacement.

Developing the regression models

Of the 207 plots for which we analyzed vegetation variables, 198 had similar independent variables as identified by Kaye et al. (2008) that could be used to predict the variance in C storage and NPP (Table 1). In the Bayesian model, we first set up a regression analysis for two different models. In one, we analyzed NPP and C storage in shrubs and trees as correlated dependent variables, since these two variables showed significant correlation. For similar reasons, we worked with another regression model that considered C storage in shrubs and trees separately. To do the estimation and prediction we used the *spBayes* package in R (Version 2.4.1, 2006). The full versions of each of these models used all the independent variables for regression. Later we dropped some of these variables, since it is customary in statistical analysis to go for model choice based on DIC (Deviance Information Criterion), which assigns a penalty for using too many independent variables in the model. Such "model overfitting" results in poor prediction or inefficient scaling at unobserved locations.

TABLE 1. Variables used in the Bayesian and nonparametric models, their definitions, and units.

Variable	Definition
Y_1	C stored in shrubs (g/m^2)
Y_2	shrub net primary productivity, NPP ($\text{g}\cdot\text{m}^{-2}\cdot\text{yr}^{-1}$)
Y_3	C stored in trees (g/m^2)
Y_4	tree net primary productivity; NPP ($\text{g}\cdot\text{m}^{-2}\cdot\text{yr}^{-1}$)
S	slope (degrees)
E	elevation (m)
δ	agriculture index: 0 if never in agriculture; 1 if ever used for agriculture
P	surface area covered by impervious surfaces (%)
L	pervious area covered by turfgrass lawn (%)
LU	land-use category
T	pervious area covered by tree canopies (%)
W	spatially random effects
j	unique spatial location
β_2	regression coefficients
σ^2	random error variance

Even after fitting the regression model to the data, there may be some spatial variability remaining in the detrended data. Our Bayesian model captured this leftover variability through hierarchical specifications and complex Markov chain Monte Carlo algorithms. However, an assumption underlying this method is that the data are unimodal, symmetrical, and with an approximately normal distribution. Our data met none of these assumptions, nor did log transformation improve data symmetry given a large number of zeros in the underlying data sets. The distribution was clearly bimodal and asymmetric, rendering the Bayesian regression model inappropriate.

Nonparametric regression does not assume any specific pattern or shape to the data and is therefore a more robust method, although it usually leads to larger confidence intervals (more uncertainty in the predictions). We therefore constructed nonparametric regression models to describe the data. Finally, we used the model-choice criterion, Akaike's Information Criterion (AIC), to choose the "best" model, based on package `npreg` in R (Version 2.4.1, 2006).

RESULTS

C storage and NPP in a desert urban ecosystem compared with cities in other biomes

The convergence hypothesis (Pouyat et al. 2006) suggests that urban areas may become more alike over time, diverging in structure and function from their surrounding native ecosystems. Specifically, in terms of C dynamics, this would mean an increase in storage and flux in cities in arid environments, and a decrease in storage and flux in urban systems located in temperate forests, when compared to the surrounding native ecosystem. In short, arid and temperate cities should converge in their C budgets over time. In particular, it is tree planting and care in arid and semiarid cities that serves as a primary driver of C storage convergence, since trees can store more C than other vegetation types that dominate those natural ecosystems. We expected this phenomenon to be especially apparent in Phoenix, where active management of landscapes by people could augment the water and nutrient limitations characteristic of desert ecosystems. Instead, when comparing our results for urban C storage and NPP associated with trees located on CAP's urban land uses (on a per area basis) to estimates for desert vegetation found in the literature, we found that the urban areas in the CAP ecosystem did not have more C or higher NPP than desert ecosystems found in the southwestern United States. At most, C storage in urban trees and shrubs is 42% of that found in vegetation in deserts, while NPP is only 20% of the total NPP estimated for comparable natural ecosystems (Fig. 2B, D). Furthermore, the overall estimates of C storage and NPP associated with urban trees were 8–63% lower than the other cities included in this analysis (Fig. 2A, C).

Comparing CAP metropolitan land-use results and those of other well-cited urban studies to average values

of C storage in natural ecosystems near each city (C estimates were found in the literature), revealed some support for the convergence hypothesis (Fig. 2A, B). Cities located in semiarid ecosystems, including Boulder, Colorado, and Oakland and Sacramento, California, actually stored more C in trees than was found in the vegetation of surrounding native grasslands, as expected according to the convergence hypothesis (Fig. 2B). Both desert and temperate forest cities had smaller C storage pools in trees compared to the vegetation in their surrounding native ecosystems; the latter result for temperate forests is also predicted when considering the homogenization of cities (Fig. 2B). Thus, in at least two of the biomes we considered there was evidence of urban ecosystem convergence based on C storage in trees.

Furthermore, as can be seen by looking specifically at the CAP ecosystem and Seattle, the level of urbanization may have a lot to do with how much C storage is possible in coarse woody vegetation (Fig. 2A). For instance, both the urban non-residential and high-urban areas of Phoenix and Seattle, respectively, had lower C storage than the residential and low-urban areas in these cities. This result lends support to recent research comparing tree cover in cities, which showed that urban morphological characteristics, including residential parcel size and pervious/impervious area, were the most influential drivers of tree cover (Biggs et al. 2014). Although social, cultural, and economic variables also can play a role in how much tree cover there is in urban areas (Iverson and Cook 2000, Grove et al. 2006, Troy et al. 2007), trees need enough space and the appropriate conditions to grow (Lowry et al. 2012, Biggs et al. 2014).

Comparing C storage of trees in cities to the surrounding natural ecosystem C pools can be a useful exercise. As most of the C storage in urban ecosystems is often associated with trees, rather than shrubs and fine herbaceous vegetation, these kind of comparisons can show overall trade-offs in C storage with potential land-use changes. Similar comparisons between trees and total vegetation NPP among the different land uses, however, provided less information about the general trade-offs in productivity with urbanization, because trees only make up a small proportion of the total NPP measured in both urban and natural ecosystems (Fig. 2C). Our comparative analysis of several well-cited results from cities globally shows, however, that the urban sites in the Phoenix metropolitan area have lower tree-based NPP than other urban areas located in less water-limited biomes (Fig. 2C). Also, these comparisons showed that urban trees across all ecosystems typically contribute to less than a quarter of the total NPP associated with all of the vegetation in the surrounding natural ecosystems (Fig. 2D).

Total C storage and NPP in the CAP ecosystem

In the CAP ecosystem as a whole, shrubs (884.29 Gg C) accounted for a slightly larger overall C pool than trees

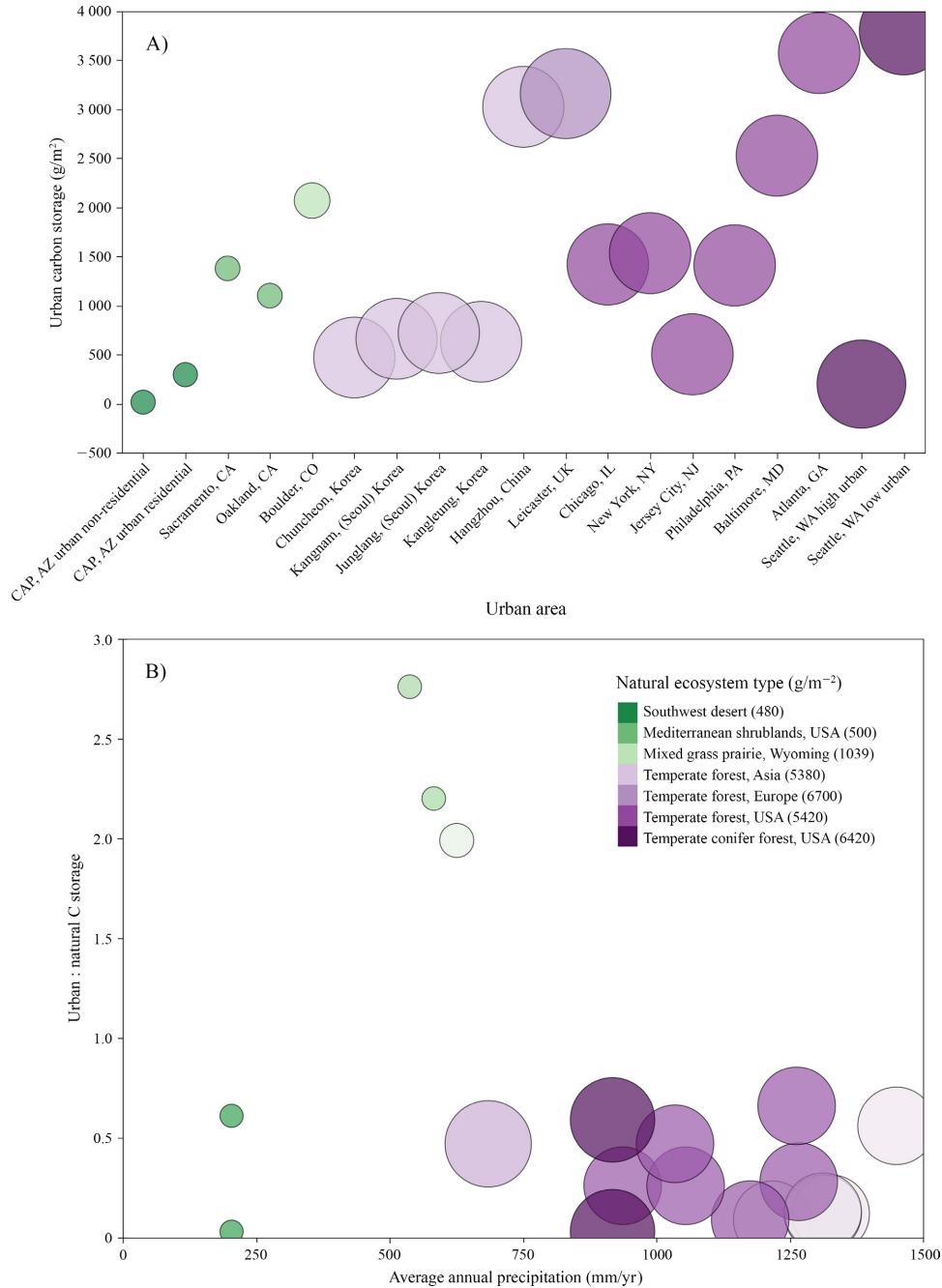


FIG. 2. C storage and NPP in a desert city compared with storage and NPP associated with trees in cities from ecosystems around the world. In all graphs, the size of the bubble represents the amount of C storage or NPP associated with all vegetation in the native ecosystem type nearest to the city. The color of the bubble indicates the native ecosystem type. (A) C storage (g/m²) for each city. C storage in urban course woody vegetation is highly variable among cities, even within the same ecosystem type. (B) Ratio of tree C storage to natural ecosystem C storage along the natural precipitation gradient. Points with values above 1 indicate that there is more C stored in the urban forest trees than has been documented for vegetation in the surrounding native ecosystem type. These data show that cities in semiarid ecosystems exhibit the capacity for increasing C storage in woody vegetation with urbanization. (C) NPP (g·m⁻²·yr⁻¹) for each city. Urban forest NPP is highly variable among cities. Our data indicate that urban trees in the desert cities may have lower NPP than cities in located in other ecosystem types. (D) Ratio of tree NPP in cities to natural ecosystem NPP along a natural precipitation gradient. Urban NPP for trees in all of the cities was less than half of the NPP documented for vegetation in the associated native ecosystem types. Data sources: Oakland and Sacramento, California (CA: Nowak and Crane 2002); Boulder, Colorado (CO: Golubiewski 2006); Korean urban forests (Jo 2002); Hangzhou, China (Zhao et al. 2010); temperate urban forests (Nowak and Crane 2002); Leicester, UK (Davies et al. 2011); Seattle, Washington (WA: Hutryra et al. 2011); southwest desert (Shen et al. 2005, Liu et al. 2012), Mediterranean shrublands (Liu et al. 2012); temperate grasslands (Schuman et al. 1999); temperate forests (Imhoff et al. 2004, Thurner et al. 2014).

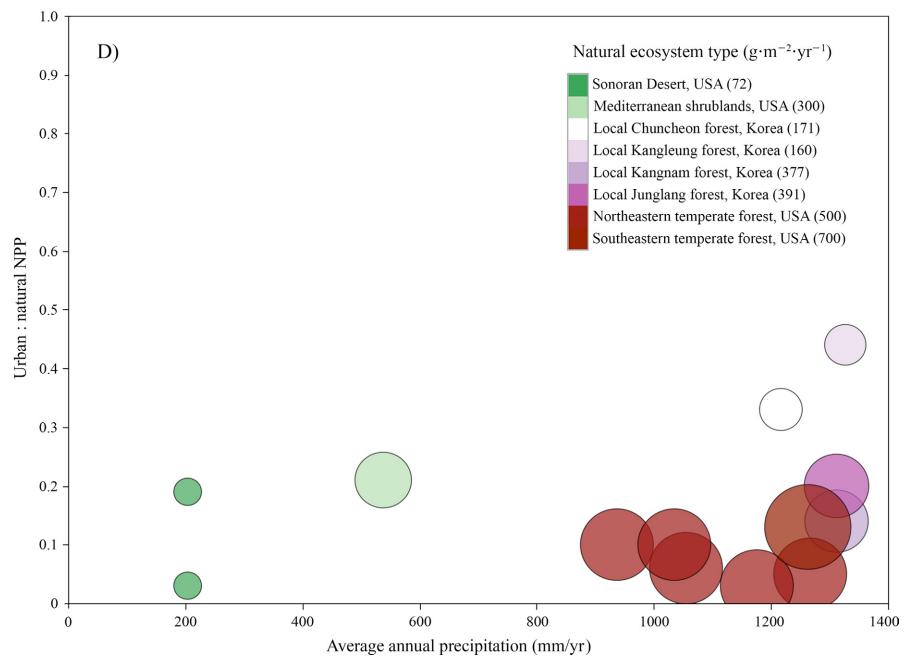
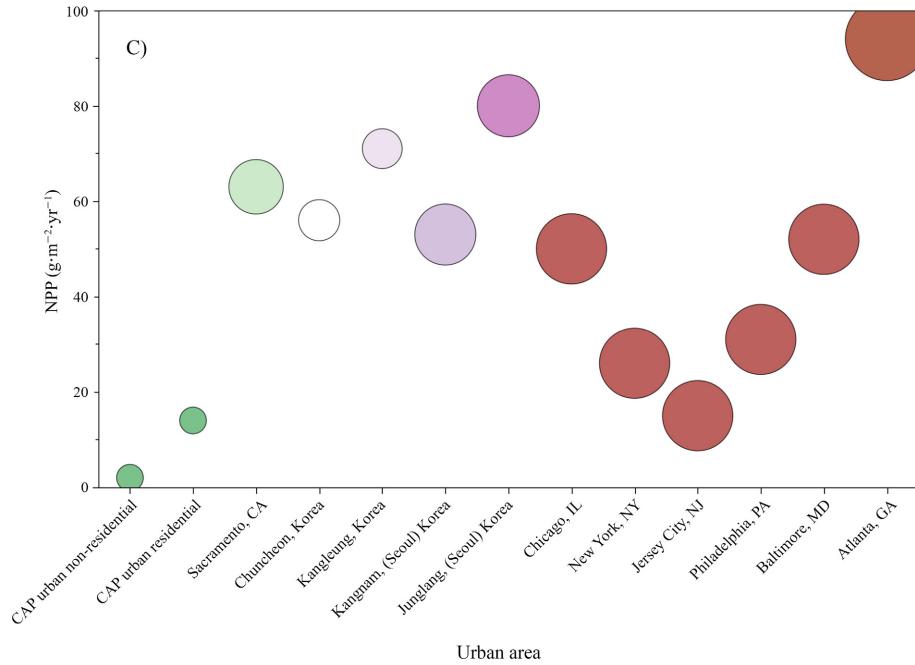


FIG. 2. Continued.

(804.36 Gg; Table 2). This is reasonable, since shrub cover was high in the desert and the desert land use accounts for 59% of the land area in the CAP ecosystem. On the other hand, trees were responsible for a larger proportion of the NPP in the CAP ecosystem (48.61 Gg C/yr for trees vs. 31.71 Gg/yr for shrubs). The young trees that dominated the CAP ecosystem were likely still actively growing and therefore may exhibit higher plant-specific productivity than shrubs. Although the UFORE

model's growth predictions accounted for the age and health of trees, the general equation we utilized for shrub productivity did not address potential variability in productivity due to age or health among shrubs in the CAP ecosystem. Among the urban land uses, xeric residential lands had higher C storage (459.67 Gg C) and NPP (21.69 Gg C/yr) associated with trees than other urban land uses (76.78 Gg C, 2.94 Gg C/yr for mesic residential, and 2.55 Gg C, 0.35 Gg C/yr for urban non-residential), again

TABLE 2. Total C storage and NPP in the Central Arizona Phoenix Long-Term Ecological Research (CAP LTER) study area, estimated using both means and medians associated with the survey data set and total land area of each land use.

Land use	Area (km ²)	Shrub C storage (Gg)		Tree C storage (Gg)		Total storage (Gg)		Shrub NPP (Gg/yr)		Tree NPP (Gg/yr)		Total NPP (Gg/yr)	
		Mean	Median	Mean	Median	Mean	Median	Mean	Median	Mean	Median	Mean	Median
Xeric residential	1606.94	55.39	20.78	1057.46	459.67	1112.85	480.45	1.64	0.93	29.07	21.69	30.71	22.62
Mesic residential	175.04	5.35	0.80	156.23	76.78	161.58	77.58	0.30	0.04	3.99	2.94	4.29	2.98
Urban non-residential	175.66	6.25	0.02	46.71	2.55	52.96	2.57	0.26	0.02	1.49	0.35	1.75	0.37
Desert	4696.64	2329.53	862.54	897.11	265.22	3226.64	1127.76	65.02	30.58	40.56	23.48	105.58	54.06
Agriculture	1129.52	6.85	0.15	18.17	0.15	25.02	0.3	0.40	0.15	1.57	0.15	1.97	0.3
Total	7961.71	2403.36	884.29	2175.68	804.36	4579.04	1688.65	67.62	31.71	76.68	48.61	144.30	80.32

owing to the larger land area associated with xeric landscapes (20% of the total land area).

A majority of the C storage associated with vegetation in the CAP ecosystem is in the desert, because the areal extent of the desert land-cover type is greatest in the study area (Fig. 1). Kaye et al. (2008) studied soil nutrients in the CAP ecosystem and found similar results. Therefore, in terms of both vegetation and organic and inorganic C in soils, the desert and xeric land uses accounted for the bulk of the C storage (Fig. 3). Soil C pools were substantially larger than C found in vegetation. Combining vegetation and soil C pools among all land uses, we show that agricultural lands contributed significantly to regional C storage by virtue of their high soil C (Fig. 3).

C dynamics and land use

Storage of C and NPP associated with trees and shrubs varied significantly with land use. Of 21 comparisons for each variable (e.g., shrub C, shrub NPP, tree C, and tree NPP) the Wilcoxon method produced 12–14 differences and the Steel-Dwass method indicated 8–10 differences. The largest differences were among urban, desert, and agricultural land-use categories; within the urban land-use categories, only a few land uses differed significantly. In general, the data showed low C storage in trees and shrubs in agricultural lands (0.13 g/m² vs. 175.25–388.94 g/m² for trees and 0.13–12.93 g/m² for shrubs in other land uses), and higher C storage in shrubs in the desert (183.65 g/m²) compared to all other land uses (0.13–12.93 g/m²; Table 3). Furthermore, the urban land uses tended to have fewer shrubs than the desert, but more trees. Finally, many of the plots on urban land have sparse vegetation and therefore exhibit relatively low C storage and NPP.

Spatial scaling

Past research on C and nutrient dynamics in soils showed that hierarchical Bayesian statistical modeling provided improved, spatially explicit results across diverse land uses (Kaye et al. 2008). Thus, our goal was to determine if such spatial scaling techniques, when applied to plot-level vegetation data in cities, might be used to display a more nuanced and spatially explicit representation of C storage and NPP across the CAP ecosystem. We compared both Bayesian and nonparametric statistical models to traditional scaling methods, in which means or medians were applied to each land-use category. Our analyses of the mean and median C storage and NPP values for shrubs and trees for each land use in the CAP ecosystem revealed that Bayesian and nonparametric statistical models produced dissimilar results (Table 4). In particular, median values of shrub C and NPP estimated from the Bayesian model did not vary across the different land uses and were much higher than values for the other two models/analyses (Fig. 4). The

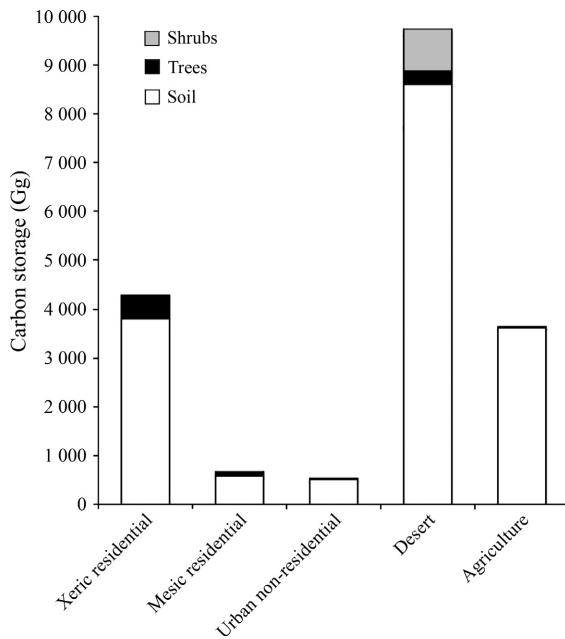


FIG. 3. Total soil and vegetation C density for each land use in the CAP ecosystem, based on combined estimates from this study and that of Kaye et al. (2008).

nonparametric model produced higher predictions of tree C and NPP across all land uses (Fig. 4). Furthermore, the nonparametric predictions estimated large amounts of C and NPP associated with trees in the agricultural and urban non-residential land uses, yet both survey data and land-cover maps show that these land-uses have little woody vegetation cover associated with them.

Since the statistical models were not able to predict vegetation-related C dynamics across the CAP region, traditional scaling approaches were revealed as the most appropriate method for estimating C storage and NPP for the ecosystem. Often means are used to scale up plot-level data; however, the distributions of these particular data make that strategy problematic. Due to the fact that these data are highly variable, and several plots had very little or no shrub or tree cover, median values best characterize the central tendency of this data set. Therefore, the most conservative and representative estimates for both C and NPP across the landscape, for both trees and shrubs, were developed using the medians associated with the survey data (Table 4, Fig. 4).

DISCUSSION

C storage and NPP in a desert urban ecosystem compared with cities in other biomes

Since most of the studies on NPP utilize a combination of remote sensing and modeling methodologies to estimate total productivity of all vegetation in the ecosystem (e.g., Cramer et al. 1999, Imhoff et al. 2004, Buyantuyev and Wu 2009, Zhao et al. 2011), comparing our results for trees and shrubs to other estimates of the effects of urbanization on NPP is tenuous at best. The modeling literature generally shows increases in NPP with urbanization in arid ecosystems and decreases in NPP in temperate ecosystems. For instance, Imhoff et al. (2004) predicted an average annual increase in $25 \text{ g}\cdot\text{m}^{-2}\cdot\text{yr}^{-1}$ in urban areas compared with the surrounding arid ecosystem in the U.S. Southwest. Although the largest annual decreases were

TABLE 3. C storage and NPP for trees and shrubs in different land uses in the CAP ecosystem.

Land use	Tree C (g/m ²)	Tree NPP (g·m ⁻² ·yr ⁻¹)	Shrub C (g/m ²)	Shrub NPP (g·m ⁻² ·yr ⁻¹)
Agriculture	0.13 (0.13–2.69)	0.13 (0.13–0.81)	0.13 (0.13–0.13)	0.13 (0.13–0.13)
Desert	56.47 (2.86–114.76)	5.00 (0.88–7.46)	183.65 (133.47–216.76)	6.51 (5.42–8.58)
Xeric residential	175.25 (1.94–435.19)	8.07 (0.51–20.63)	12.93 (6.52–20.42)	0.62 (0.28–0.87)
Mesic residential	388.94 (109.54–777.47)	15.99 (7.46–24.46)	4.60 (0.14–6.29)	0.20 (0.13–0.33)
Mixed residential	286.54 (100.82–666.99)	14.82 (8.07–27.63)	8.14 (1.58–32.53)	0.31 (0.13–1.57)
Mixture	0.13 (0.13–164.65)	0.13 (0.13–8.71)	9.30 (0.13–48.15)	0.28 (0.13–1.47)
Urban non-residential	14.50 (0.13–223.31)	1.97 (0.13–10.64)	0.13 (0.13–3.77)	0.13 (0.13–0.17)

Note: Median values are reported with 95% confidence intervals in parentheses.

TABLE 4. Total C storage (Gg) and NPP (Gg/yr) for both trees and shrubs across the CAP system estimated using the means and medians associated with the survey data and the nonparametric and Bayesian models.

	Survey data		Nonparametric		Bayesian scaling	
	Mean	Median	Mean	Median	Mean	Median
C storage (Gg)	4579.04	1688.65	3272.13	2605.18	4555.39	1968.15
NPP (Gg/yr)	144.30	80.32	142.56	116.15	118.01	84.84

Notes: These estimates only include the xeric residential, mesic residential, urban non-residential, desert, and agriculture land-use types totaling an area of 7784 km². The entire CAP study area is actually 7962 km² and includes areas that were unclassified in the land-use and cover map. There were no plots from the survey associated with these unclassified land uses and so we did not include that land area in our estimates.

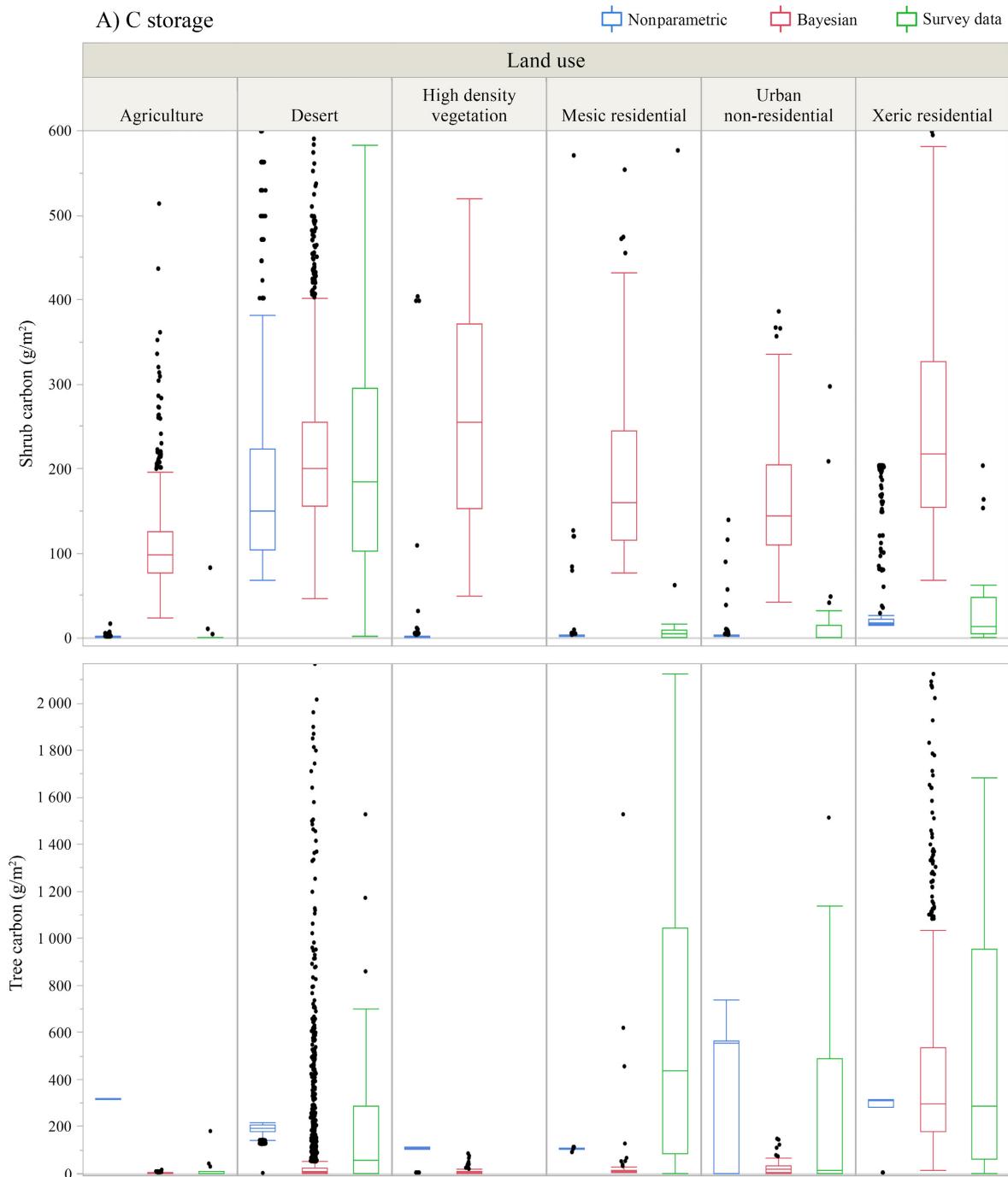


FIG. 4. Median predictions for (A) C storage and (B) NPP using the nonparametric model, Bayesian model, and the survey data. Median values are represented by the lines within each box, while the box ends show the interquartile range. The whiskers on each box indicate upper and lower data point values, and do not include outliers.

associated with urban areas in the northeastern and southeastern United States ($\sim 95 \text{ g}\cdot\text{m}^{-2}\cdot\text{yr}^{-1}$), peri-urban areas across all ecosystem types showed some increases in NPP. Buyantuyev and Wu (2009) did not report NPP in $\text{g}\cdot\text{m}^{-2}\cdot\text{yr}^{-1}$, yet their analysis of NDVI across the CAP ecosystem agrees with Imhoff et al.'s predictions that overall productivity is higher in urban areas compared

with the desert; however, this result was particularly representative of dry years and was reversed during wet years. Similarly, studies in China and Australia report decreased NPP with increasing settlement in temperate regions but increased NPP with rising population in semiarid ecosystems (Luck 2007, Lu et al. 2010, Pei et al. 2013).

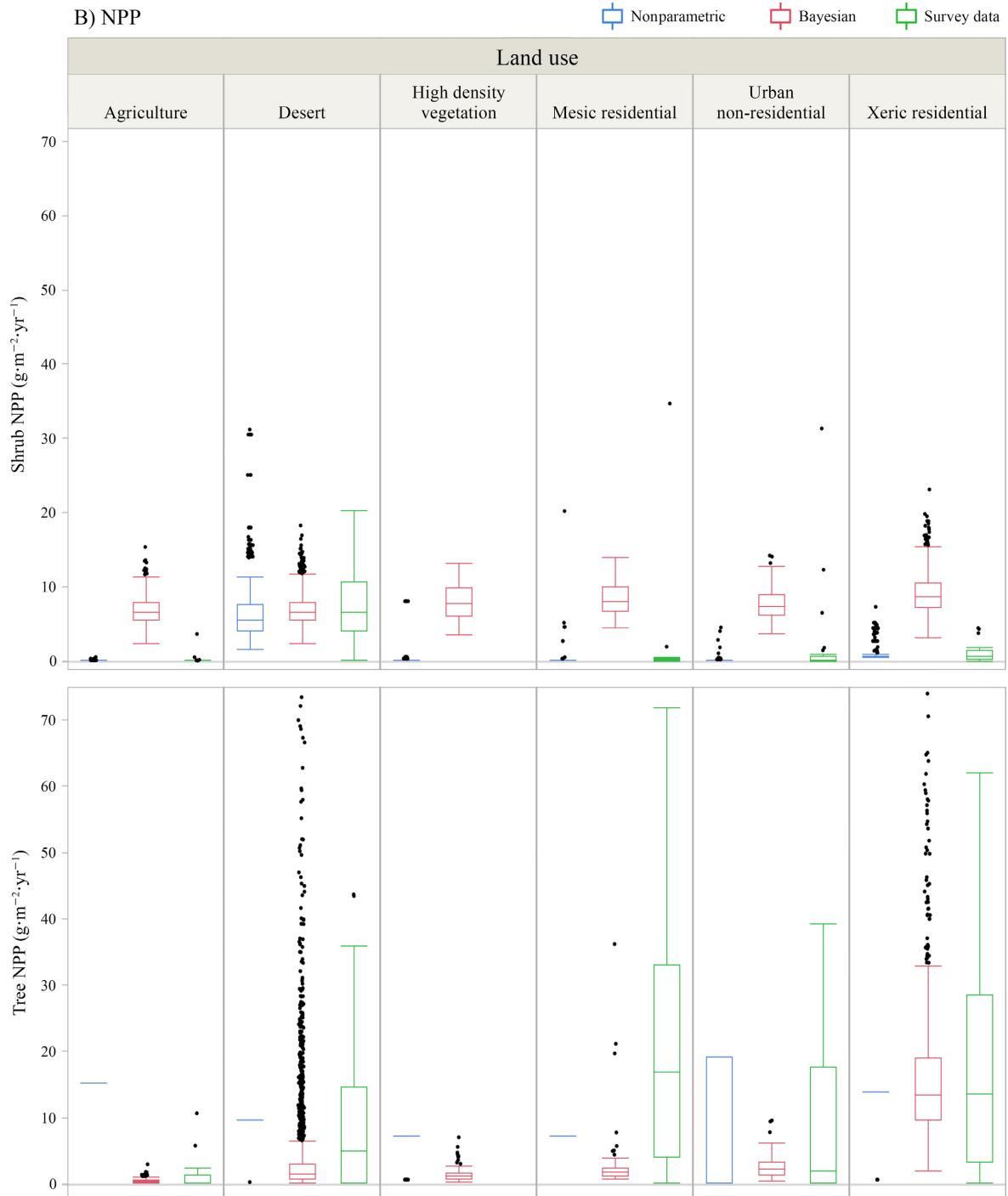


FIG. 4. Continued.

Considering the findings from modeling and remote sensing studies, the potential for a desert city to exceed the C storage and NPP of the surrounding ecosystem should be great, yet our results based on survey data for trees and shrubs in the CAP ecosystem did not meet this expectation. Our result is particularly surprising considering the sprawling nature of residential areas in the

region, since large, open parcels can support more trees (Biggs et al. 2014). It is possible that the low C storage and NPP seen in the Phoenix area could actually be a function of city age: most of the cities in CAP ecosystem are relatively new developments. As the trees that are already in the area continue to grow and more people plant and maintain urban forest canopy, we could see an

increase in C storage and NPP over time. There have been efforts to increase tree cover in the area to reduce heat-related health issues in the region, which would also contribute to an increase in C storage and NPP. This increase in tree canopy cover, however, would come at the cost of another limited resource in the area, water (Pataki et al. 2011). Considering this cost, people in the Phoenix metropolitan area may not want to plant more trees, or they may find xeric landscaping more aesthetically appealing and natural in a desert city (Larsen and Harlan 2006). In fact, it seems like it would be easier to overcome water limitations in a semiarid city compared to a desert city, and the urban areas in Phoenix may not ever achieve the same C storage and NPP status of their semiarid counterparts. Nonetheless, a recent review of the literature (Davies et al. 2013) suggests that there are many more studies on urban tree canopy located in temperate ecosystems than semiarid and arid environments. More work in these water-limited cities is needed before the full potential of C storage and NPP can be assessed, especially with reference to the convergence hypothesis.

Total C storage and NPP in the CAP ecosystem

Utilizing a hierarchical patch mosaic-urban ecosystem model (HPM-UEM), Zhang et al. (2013) estimated that the total C storage in the CAP ecosystem is equal to 16,700 Gg C with 74% (12358 Gg C) associated with soil organic C storage and 21% (3500 Gg C) attributed to vegetation. According to results from Kaye et al. (2008) and our analysis, the HPM-UEM may underestimate total soil organic C storage (organic C storage from Kaye et al. was 18710 Gg C) and overestimate vegetation C storage (tree and shrub C from our analysis was 1688 Gg C). Utilizing median results from our study and those from Kaye et al. (2008) show that trees and shrubs alone may contribute about 8% to the total C storage potential of vegetation and soils in the CAP ecosystem. Including the total estimated inorganic C in the soil in the CAP ecosystem means an even lower percentage of C associated with vegetation (4.7%). One reason for the discrepancy is that the HPM-UEM analysis also includes other vegetation types, including fine vegetation like lawns, herbaceous material, and succulent plants; however, in urban ecosystems, trees and shrubs typically account for a majority of the C storage capacity (Nowak 1994). The estimates of vegetation and soil C storage using the HPM-UEM are particularly difficult to compare with traditional scaling results, however, since Zhang et al. (2013) utilized different estimates of total land area and land area per land use than employed in our study and Kaye et al. (2008). Nonetheless, a spatial and multi-scale analysis utilizing HPM-UEM has potential for mapping and analyzing the distribution of C dynamics across multiple ecosystem types, including metropolitan areas, and the spatial and temporal specificity of this process-based model may be the main reason for differences in the estimates of total C storage.

In terms of NPP, we estimated that shrubs and trees in the CAP ecosystem sequester 80.32 Gg C on an annual basis. In comparison, transportation and energy emissions in Maricopa County amounting to 9599 Gg C were released into the atmosphere in 2008 (data from VULCAN; Gurney et al. 2009). This striking contrast supports the claim that ecosystem productivity in and around urban areas has little promise for offsetting a meaningful amount of CO₂ emissions associated with the activities of people living in cities and towns (Pataki et al. 2006).

Comparisons with other predictions of NPP indicated that our estimated NPP is likely conservative. Although Zhang et al. (2013) did not report total NPP for the region, applying their average NPP rate for the study system (175 g·m⁻²·yr⁻¹) evenly across all land uses implies a total NPP of 1156 Gg C/yr. Although that value is equal to 12% of the energy and transportation emissions for 2008 in the region, it was meant to represent NPP associated with all vegetation. Zhang et al. (2013) estimated that lawn and crop productivity were responsible for a significant portion of the NPP in the CAP ecosystem (59% or 682 Gg C/yr), and productivity associated with herbaceous vegetation and grass does not lead to long-term storage in the ecosystem. Alternatively, the modeling analysis also indicated that trees and shrubs make up 10% and 28% of the total NPP (439 Gg C/yr), which was 5.5 times higher than our estimated NPP values.

Comparing our estimates of rates of NPP for trees and shrubs just within the desert land use (11.51 g·m⁻²·yr⁻¹) to a model that predicts desert ecosystem productivity (Shen et al. 2005) also suggests that our median NPP rates may be conservative. Modeling NPP according to six different plant functional types, Shen et al. (2005) estimated average aboveground NPP (ANPP) of the *Larrea-Ambrosia*-dominated desert communities surrounding the Phoenix Metropolitan area was 72.3 g·m⁻²·yr⁻¹. Although this value for ANPP is five times higher than our estimated total NPP for shrubs and trees in the desert, it also accounts for all of the vegetation in the ecosystem. Furthermore, the model showed that in some years ANPP could be as low as 11.3 g·m⁻²·yr⁻¹ and in other years as high as 229.6 g·m⁻²·yr⁻¹. This large variation in ANPP in the desert can be accounted for in models that utilize climate data to simulate plant growth, since the primary driver of interannual variation is rainfall (Shen et al. 2005). Our shrub NPP predictions were based on one study by Chew and Chew (1965), the only study available that gave an estimate for NPP based on shrub cover for a shrub type found in this desert ecosystem. Scaling up ground-based estimates of NPP is not only laborious, but can lead to erroneous estimates of NPP at large scales (Wu et al. 2006, Buyantuyev and Wu 2009). Similarly, the UFORE model may have underestimated NPP associated with trees, since productivity for the CAP region was based on tree growth in other ecosystems (Nowak and Crane 2000). Furthermore, it seems likely that our NPP estimates were especially low for urban environments, where people add water and nutrients to

their gardens and landscapes. Empirical analyses are needed to decipher which NPP values are most realistic for the region at multiple scales, although it is notoriously difficult to compare methodologies that scale up productivity values to those that either model or measure total ecosystem fluxes (Wu et al. 2006).

C dynamics and land use

Early studies on C dynamics in urban vegetation implied that C stored in shrubs was inconsequential; for example, Nowak (1994) found shrubs stored about 4% of the total C associated with trees and shrubs in Chicago. Therefore, most studies utilizing the i-Tree tool for assessing and managing community forests (i-Tree Canopy 2014), as well as the models leading to the development of the entire i-Tree suite (i.e., the UFORE and STRATUM models), do not include estimates of C associated with shrubs. Our findings support previous indications that C associated with shrubs in cities does not account for a large portion of the total C in woody vegetation; however, when considering land-use changes associated with the urbanization process, the C trade-offs among different vegetation pools need to be addressed. In particular, our results indicated that urban land uses have higher storage and increased productivity associated with trees, while C storage in the desert landscape was predominately associated with the presence of shrubs. If we only measured and analyzed vegetation C for trees, we would have assumed a greater accumulation of C in vegetation during urbanization while not accounting for the shrub C lost when desert ecosystems are urbanized.

Among the urban land uses, we expected greater C storage and higher NPP to be associated with trees on mesic residential sites, due to the water and nutrient inputs typically required to support these landscapes. Our analysis showed that trees in all residential land uses store more C and have higher NPP than non-residential urban lands, but the landscaping regimes (xeric, mesic, and mixed) did not differ significantly from one another. Similar results have been reported for other studies of C storage in vegetation in cities, which show that low C storage is particularly common in non-residential land uses (Nowak and Crane 2002, Hutrya et al. 2011, Strohbach et al. 2012). In the case of Phoenix, the lack of difference among the different landscape choices suggests that individual properties feature a similar number of trees and shrubs regardless of the kind of landscaping option selected. This is true despite the fact that plant species composition differs between mesic and xeric landscaping regimes (Walker et al. 2009). Considering the length and intensity of summer heat in the desert, it may be that many residents are using large woody vegetation to provide some shade on their property, even when attempting to design and manage for a less water-dependent landscape. A recent study bears out the cooling benefit of xeric vegetation types; although

drought-tolerant species typically planted in xeric landscapes are slightly less effective than large shade trees, they nevertheless have significant impacts on local microclimates (Chow and Brazel 2012).

Spatial scaling

Traditional scaling was the most conservative and representative approach for estimating C storage and NPP associated with trees and shrubs across the CAP ecosystem. Although Bayesian scaling provided a valuable method for quantifying spatial distribution of soil nutrient and C pools across the urban landscape (Kaye et al. 2008), the variables utilized in that model, including slope, elevation, and agricultural history, were not useful for predicting C associated with vegetation. This result is likely due to differing drivers for urban vegetation and soil C dynamics (Golubiewski 2006, Pouyat et al. 2006); for example, although landscape choices like fertilization or mulching can have some influence on soil C, planting choices are a primary control on vegetation types in cities. Furthermore, modeling vegetation-associated C spatially across the landscape is more challenging due to the heterogeneous and disconnected nature of tree and shrub cover (Cadenasso et al. 2007). A recent analysis of C storage in urban trees suggests land-cover maps can help improve estimates of C storage (Strohbach et al. 2012). Advances in remote-sensing technology have made high-resolution tree cover classification more popular in cities, and therefore, a spatially informed approach based on land-cover classifications seems viable (Grove et al. 2006, Troy et al. 2007, Bigsby et al. 2014).

Best practices for quantifying and comparing ecosystem fluxes across heterogeneous urban areas

Recent studies suggest that the C storage and NPP estimates from many cities may not be realistically comparable. Raciti et al. (2012) showed that depending on how an urban area is defined, estimates of C stocks could vary significantly, and this is certainly true of our study area, in which over one-half of the total is undeveloped desert. Davies et al. (2013) expanded on the issue of comparability after reviewing all of the literature available on C storage in urban forests. They highlighted that land-use categories, land-tenure status, and resolution of land-use and cover maps can all play a role in uneven and incomparable estimates of C storage. Furthermore, most studies do not actually use biomass equations developed for trees locally, and depending on what biomass estimates are implemented, C storage estimated can vary greatly (McHale et al. 2009). NPP estimates have similar constraints, since most predictions are based on general growth equations. These studies indicate a real need for standardized methodologies of city ecosystem C and nutrient budgets so that researchers can truly compare across ecosystems and assess the validity of theories like urban ecosystem convergence (see also Hutrya et al. 2014).

Nowak et al. (2013) conducted a nationwide analysis of tree C storage and NPP in cities across the United States. They utilized C density per unit of tree canopy cover from several key analyses in a few major biomes, and then scaled up standardized values to acquire citywide and state-level estimates. These kinds of comparisons are valuable because the estimates are based on one unified methodology. However, depending solely on tree-cover measurements can lead to inaccurate assessments of C storage, since other variables like land tenure influence tree species, size, and number, which all have an influence on C dynamics (McPherson and Simpson 2001, Nowak et al. 2002, Hutrya et al. 2011, Strohbach et al. 2012). Incorporating LiDAR data into high-resolution land-cover classifications shows some promise for analyzing the size of trees (i.e., tree height) contributing to canopy cover in cities, and may lead to more accurate assessments of C storage in trees and other vegetation (O'Neil-Dunne et al. 2013, Raciti et al. 2014). Therefore, with the advancement of remote-sensing technologies, one way to ensure the continued use of plot-level data for assessing the spatial heterogeneity of C storage and flux in cities may be to standardize all estimates by land-cover type. With the increasing availability of high-resolution land-cover classifications, cover-based ecosystem processes can be evaluated at multiple scales across cities. Then how researchers define urban or specific land uses becomes less of an issue, because these data can be easily accessed and analyzed according to a multitude of boundaries.

A prime goal of studies of C storage and NPP in cities is a more accurate assessment of the impact of urbanization on the C cycle at multiple scales. Studies on trees and other vegetation are only partially answering the important questions regarding urban C budgets. To date, complete C budgets are rare and the appropriate data for such an analysis are difficult to access (Pataki et al. 2006, Churkina et al. 2010, Zhao et al. 2011). Atmospheric methodologies for assessing C dynamics in cities are underutilized (Hutrya et al. 2014), but at the same time will not provide the detailed information needed to understand the drivers of C fluxes at fine scales (Pataki et al. 2006). Zhao et al. (2011) showed the value of looking at CO₂ emissions and NPP spatially in a study on the state of Florida (USA), yet this kind of analysis is also needed at finer scales to capture the heterogeneity associated with different kinds of development patterns. Emissions data could also be linked to high-resolution land-cover classifications utilizing demographic and land cover-based predictors of emissions. These kinds of spatially explicit analyses will be useful to planners and developers aiming to reduce the footprint of cities.

CONCLUSION

The convergence hypothesis proposes that cities become more like one another and diverge from their surrounding ecosystems. The prediction following from this

hypothesis, that urbanization will increase C storage and NPP in arid ecosystems (due to inputs of water and nutrients provided by humans in cities and towns), was not supported by this study. Comparing C storage and NPP of trees across cities shows that urban areas in the CAP ecosystem have relatively lower C storage and productivity than temperate and semiarid urban ecosystems. To date, there has been very little research on arid cities, and most of the information available comes from broad-scale remote sensing and modeling. As urbanization occurs in the Phoenix area, and specifically when the desert is converted to urban land uses, there is a change in the main C-storage pools, with trees replacing shrubs and tree C exceeding C stored in shrubs. Carbon dynamics associated with shrubs are often left out of most urban analyses and our study indicates that in arid and semiarid areas shrubs may be an important contributor to carbon storage and NPP before urbanization and in the overall ecosystem. Bayesian scaling techniques were not useful for predicting C associated with vegetation spatially across the urban landscape. Modeling C dynamics associated with vegetation may be difficult in urban ecosystems due to the heterogeneous and disconnected nature of tree and shrub cover. However, integrating plot-level data with high-resolution land-cover analyses provides a promising step forward in the search for standardized methodologies for spatially assessing C dynamics across a city, and could lead to more reliable comparisons of carbon dynamics among cities.

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