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Current and historical land use influence soil-based ecosystem services in an urban landscape

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Abstract. Urban landscapes are increasingly recognized as providing important ecosystem services (ES) to their occupants. Yet, urban ES assessments often ignore the complex spatial heterogeneity and land-use history of cities. Soil-based services may be particularly susceptible to land-use legacy effects. We studied indicators of three soil-based ES, carbon storage, water quality regulation, and runoff regulation, in a historically agricultural urban landscape and asked (1) How do ES indicators vary with contemporary land cover and time since development? (2) Do ES indicators vary primarily among land-cover classes, within land-cover classes, or within sites? (3) What is the relative contribution of urban land-cover classes to potential citywide ES provision? We measured biophysical indicators (soil carbon [C], available phosphorus [P], and saturated hydraulic conductivity [K_s]) in 100 sites across five land-cover classes, spanning an ~125-year gradient of time since development within each land-cover class. Potential for ES provision was substantial in urban green spaces, including developed land. Runoff regulation services (high K_s) were highest in forests; water quality regulation (low P) was highest in open spaces and grasslands; and open spaces and developed land (e.g., residential yards) had the highest C storage. In developed land covers, both C and P increased with time since development, indicating effects of historical land-use on contemporary ES and trade-offs between two important ES. Among-site differences accounted for a high proportion of variance in soil properties in forests, grasslands, and open space, while residential areas had high within-site variability, underscoring the leverage city residents have to improve urban ES provision. Developed land covers contributed most ES supply at the citywide scale, even after accounting for potential impacts of impervious surfaces. Considering the full mosaic of urban green space and its history is needed to estimate the kinds and magnitude of ES provided in cities, and to augment regional ES assessments that often ignore or underestimate urban ES supply.

Key words: carbon; ecosystem services; historical ecology; land-use change; phosphorus; runoff regulation; saturated hydraulic conductivity; soil; urban ecosystems; water quality.

Introduction

Urban areas are complex mosaics of land-cover types with different land-use histories and vegetation conditions (Zhou et al. 2017), all of which can influence ecosystem services (ES; Gaston et al. 2013): the benefits people receive from ecosystems (MA 2005). Urban ecosystems are temporally and spatially heterogeneous at fine scales, have distinct climates, and can differ greatly in their biodiversity compared to surrounding natural and rural ecosystems (Forman 2014). Moreover, as urban areas expand, new green space is incorporated into existing cities, often on former agricultural land. Thus, cities are ideal for exploring the sensitivity of ES to current and past drivers (Dallimer et al. 2015), and assessing the degree to which land-use legacies may be unappreciated but important influences on ES (Ziter et al. 2017). However, studies of urban ES rarely address the high spatial heterogeneity and complex land-use histories of cities, particularly when considering multiple services (Gaston et al. 2013, Haase et al. 2014, Ziter 2016).

Many studies rely on land-cover-based proxies to map ES (Haase et al. 2014, Ziter 2016) because of data constraints, and this can be an important first step towards inclusion in management or policy (Chan et al. 2006, Raudsepp-Hearne et al. 2010). However, land-cover-based proxies are often poor surrogates for ES (Eigenbrod et al. 2010). This may be particularly true in urban ecosystems, given the lack of empirical data for many services, and high within-land-cover variability driven by differences in land-use and management (Loram et al. 2008). For example, various types of green space (i.e., non-impervious areas) within a city may be classified as the same land cover, but differ in plant community composition, number and size of trees, and soil conditions, leading to differences in ES. In Leicester, UK, researchers observed higher soil carbon stocks in residential yards than in public green space and under trees and shrubs compared to other vegetation (Edmondson et al. 2014). In Melbourne, Australia, degree of habitat complexity in urban parks influenced hydrological ES such as runoff regulation (Ossola et al. 2015). Yet such ecological differences are rarely considered in a land-cover-based approach (Haase et al. 2014). A better understanding of the extent to which ES indicators vary among and within land-cover classes is needed to assess potential for current ES provision and to determine the degree to which changes in management might alter ES in heterogeneous urban landscapes.

Approaches based on current landscapes also ignore the role of past land-use in explaining contemporary distribution of ES. Time lags and land-use legacies are widespread ecological phenomena (Foster et al. 2003, Burgi et al. 2017), with

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historical land-use affecting ecosystem functions for decades to millennia (Dambrine et al. 2007, Rhemtulla et al. 2009). Despite this knowledge, ES research has only recently begun to adopt temporal approaches (but see Díaz-Porras et al. 2014, Dallimer et al. 2015, Renard et al. 2015, Tomscha and Gergel 2016). Urban areas have undergone shifts in land use/land cover during development, and thus can be excellent systems for assessing effects of land-use legacies on ES. Indeed, in one of the few studies incorporating historical land use, five of eight ES in the city of Sheffield, UK were better predicted by past land use than by current patterns, with lag times ranging from 20 to 100 yr (Dallimer et al. 2015). Understanding such legacies is central to understanding the extent that ES can be inferred from contemporary landscape pattern, as well as how ES and relationships between them may shift over time as landscapes continue to change (Ziter et al. 2017).

Urban ES dependent on soil and soil–vegetation relationships may be particularly susceptible to legacy effects and time lags in ES supply, as these services are underlain by slow ecological processes (Ziter et al. 2017). This is especially relevant in former agricultural areas (Raciti et al. 2011), which characterize many contemporary urban landscapes in North America. Agricultural activities (e.g., tillage, fertilization) fundamentally change soil nutrient pools such as carbon (C), nitrogen, and phosphorus (P). These legacies can last for decades to millennia following cessation of agriculture and re-establishment of forests or grasslands (Fraterrigo et al. 2005, McLauchlan 2007). However, little data exists regarding legacy effects in areas that subsequently urbanized (but see Golubiewski 2006, Pouyat et al. 2008, Raciti et al. 2011), with existing studies often focusing on soil C in residential lawns. Urban lawns in both Colorado's Front Range and the city of Baltimore, Maryland, USA store more soil C than native grassland or forest ecosystems, and older lawns store more C than young lawns (Golubiewski 2006, Raciti et al. 2011). Phosphorus has been hypothesized to follow the opposite trend, with soil P levels expected to decrease following agricultural conversion. However, legacy effects on urban soil P have been studied to a lesser extent than C, and the relationship is less clear (Bennett 2003, Kara et al. 2011), despite known implications for water quality (Hobbie et al. 2017, Motew et al. 2017). Urban ES research would benefit from further understanding of the effects of agricultural legacies on a greater number of services, across different types of urban green space.

We measured multiple soil-based indicators across five land-cover classes in a historically agricultural urban landscape to determine the extent to which current and past drivers influence three ES, carbon storage, water quality regulation, and runoff regulation (Table 1), that are expected to be sensitive to patterns in land use/land cover and land-use legacies. These regulating services are socio-culturally valuable (Martín-López et al. 2012), and are among the most frequently researched ES in urban systems globally (Ziter 2016); yet to our knowledge, they have not previously been measured in the same study. We asked (1) How do indicators of soil-based urban ecosystem services vary with contemporary land cover and time since development? (2) Do indicators of soil-based urban ecosystem services vary primarily among land-cover classes, within land-cover classes, or within sites? (3) What is the relative contribution of urban land-cover classes to soil-based ecosystem services at the citywide scale?

We expected agricultural legacies to be an important driver of ES, but that effects would differ among services. We anticipated that C storage and water quality regulation, ES based on soil pools (C and P) known to change slowly over time, would be more susceptible to legacy effects than runoff regulation. We expected runoff regulation to be more responsive to current land cover, however, as we did not anticipate a strong legacy effect on soil texture, which is a key component of saturated hydraulic conductivity. We also expected the scale of variability to differ among soil properties, as well as between more and less developed land-cover types. Regarding soil properties, we expected C storage to show high within-site variability due to close association with litter inputs and thus fine-scale patterns of vegetation, and available P to show lower within-site, but high among-site, variability due to agricultural legacies of fertilizer application (which tends to be applied evenly within fields, but vary between farms/farmers; e.g., Bennett et al. 2005). We expected saturated hydraulic conductivity to show high among-land-cover variability, as we anticipated differences in soil texture and compaction to be more pronounced at this scale. We also anticipated more developed land-cover types (e.g., residential yards) to show higher within-site variability in soil properties, due to higher anticipated fine-scale heterogeneity in vegetation and management than in

Table 1. Ecosystem services, biophysical indicators, and rationale for three soil-based ecosystem services quantified across land-cover types and historical gradients in Madison, Wisconsin, USA.

Ecosystem service	Indicator	Description/Rationale
Carbon storage	soil carbon (C)	Soil carbon storage typically accounts for the majority of carbon storage in urban systems. Soil C acts not only as a climate regulation service, but is also indicative of general soil quality, underlying a range of benefits
Water quality regulation	available phosphorus (P)	Phosphorus is a key driver of surface water quality in the Madison region (Motew et al. 2017). Soil phosphorus levels are indicative of potential P runoff into water bodies. While available P (Bray 1-P) is not a direct measure of total P or P runoff, available P is closely correlated with total P in our study area (Bennett 2003), and is also comparable with historically available data.
Runoff regulation	saturated hydraulic conductivity (K_s)	Soil condition is a key factor linking land use and runoff/ flood regulation (Depietri et al. 2011). K_s, derived from soil texture, organic matter, and bulk density, is a key control over potential infiltration rates.

semi-natural (e.g., forest, grassland) sites, where we anticipated higher between-site variability.

Methods

Study area description

This study was conducted during summer 2015 in Madison, Wisconsin, USA (Fig. 1), which encompasses 244 km^2 (199 km^2 land, 45 km^2 water) (U.S. Census Bureau 2010) and is bordered by several suburbs (city population 245,000; urban area population 407,000). The climate is continental, characterized by warm humid summers and cold winters (1981–2010 mean temperature 22°C July, −7°C January, annual precipitation of 87.6-cm; National Climatic Data Center; *data available online*).[2] Situated in an urbanizing, agricultural watershed, much of the City of Madison was developed on former farmland, and the surrounding county remains primarily in agricultural production. Current land cover in the city is dominated by low- and medium-density developed land and open space, but also includes deciduous forest, high-density developed land, and grassland (Table 2). The remainder of the city is made up of agricultural land at the city's outskirts, wetlands, and barren land.

As in many urban landscapes, cultural and socioeconomic variability in the city's many distinct neighborhoods underlies management and aesthetic decisions that contribute to ecological variability of yards and other private green space. Madison also has a strong legacy of appreciation for urban natural areas, including a network of 250 parks and over 700 ha of conservancy land (Park Division 2012). Bolstered by the University of Wisconsin Arboretum and Lakeshore Preserve, this wealth of green space places Madison among the nation's highest ranked cities for per capita parkland. Combined with the considerable area of private yards and gardens in this mid-sized city, this variety and abundance of green space provides ample opportunity to study urban ES.

Soils in the region developed under the influence of past glaciation, which left behind a sandy-loam, calcareous till in the eastern two-thirds of Dane County. Soils in the greater Madison area are predominantly Alfisols (Hapludalfs in particular), and to a lesser extent Mollisols (predominantly Argiudolls and Endoaquolls), which are associated with the forest and oak–savanna vegetation that dominated the region prior to European settlement, along with pockets of wet sedge meadows (Bockheim and Hartemink 2017, Appendix S1: Fig. S1). Soils underlying study sites are largely Typic Hapludalfs (50% of sites, predominantly Dodge, McHenry, St. Charles series) and Mollic Hapludalfs (21% of sites, predominantly Batavia series). Comprising a further 20% of sites are Typic Argiudolls (7%, predominantly Plano series), Typic Endoaquolls (7%, predominantly Colwood series), and Udollic Endoaqualfs (6%, Virgil series) (Appendix S1: Table S1; data available from the Soil Survey Geographic [SSURGO] database; *available online*).[3]

[2] www.ncdc.noaa.gov/cdo-web/datatools/normals
[3] https:/websoilsurvey.nrcs.usda.gov

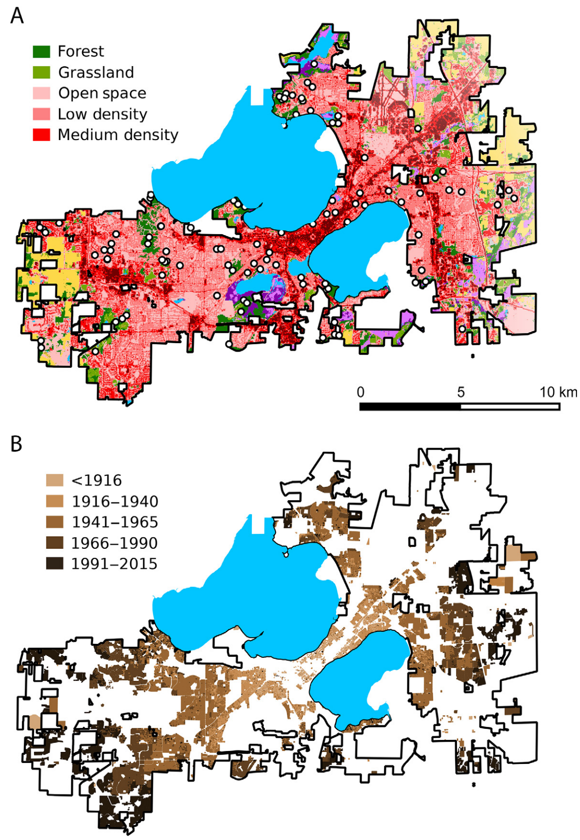

Fig. 1. Map of study area. (A) Land cover in the City of Madison, Wisconsin, USA. Legend indicates land cover classes included in the present study. White circles indicate site locations. (B) Year of development for residential lots within the City of Madison, Wisconsin. Blue areas in panels A and B are lakes.

Study design

We used a stratified design including five land-cover classes, forest, grassland, open space, and low- and medium-density developed land (Table 2), with sites spanning an approximately 125-year gradient of time since development within each land-cover class. A total of 100 sites (n = 20 within each land-cover class) were selected to span these gradients and cover the geography of the city (Fig. 1A). Sites within each land-cover class were well distributed geographically throughout the city to avoid bias due to underlying soil type or other edaphic features. We obtained landowner permission for all sites prior to sampling.

Selected land-cover classes represent a continuum from semi-natural to more developed green space, and include both private and public land. Given the predominantly agricultural history of the region, we chose time since development as a proxy of time since conversion from agricultural use. For low- and medium- density developed land covers, composed of residential property, this gradient was based on the year built (City of Madison Assessor's Office; Fig. 1B). We identified suitable candidate sites using GIS software, and canvassed for permission. To again avoid underlying spatial patterns that may bias our results, we deliberately ensured that sites of similar ages were geographically distributed throughout the city to the extent possible.

TABLE 2. Five land-cover classes encompassing the majority of urban green space in Madison, Wisconsin, including the proportion of each land-cover class composed of building cover, non-building impervious cover, and non-impervious cover.

			Land-cover (%)		
Land cover	Classification description (from Gillon et al. 2016)	Madison landscape (%)	Building cover	Non-building impervious cover	Non-impervious cover (i.e., green space)
Deciduous forest	Dominated by trees generally >5 m tall, and >20% of total vegetation cover. More than 75% of tree species shed foliage simultaneously in response to seasonal change.	6.83	1.24	1.31	97.45
Grassland	Dominated by graminoid or herbaceous vegetation, generally >80% of total vegetation. Sites in our study include unmowed meadows, as well as restored prairies.	4.55	0.12	1.27	98.61
Open space	A mixture of constructed materials, but mostly vegetation in the form of lawn grasses. Impervious surfaces account for <20% of total cover. Sites in our study include city parks (n = 17), golf courses (n = 2), and cemeteries (n = 1).	18.09	3.68	9.95	86.37
Low-density developed	A mixture of constructed materials and vegetation. Impervious surfaces account for 20% to 49% percent of total cover. Sites in our study are residential lots.	30.06	11.18	23.99	64.83
Medium-density developed	A mixture of constructed materials and vegetation. Impervious surfaces account for 50–79% of the total cover. Sites in our study are residential lots.	17.08	13.96	41.92	44.12

Similar methods were followed for open space and grassland sites, however, the temporal gradient was based on available historical records, as well as consultations with the City of Madison parks department and private land managers. For forest sites, we used three sets of aerial photos (from 1937, 1974, and 2013) to choose sites that were in forest cover prior to 1937, between 1937 and 1974, and post-1974.

Field sampling

Each site consisted of a 30 × 30 m area, within which we established four 5 × 5 m subplots (one in each quarter) to assess the effect of fine-scale heterogeneity on soil properties. In each subplot, we conducted vegetation surveys to account for variability in vegetation type and structure within and among sites, with the specific assessment dictated by the land-cover class (Appendix S1).

We collected soil samples for chemical analysis (C, P) at 0–5, 5–10, and 10–25 cm fixed soil-depth intervals. Three 2-cm diameter cores were collected from each 5 × 5 m subplot, then composited into one sample per depth per subplot (i.e., four samples per depth per site). We also collected one 10 mcm depth bulk density (g/cm^3) core from each 5 × 5 m subplot using a 184-cm^3 stainless steel cylinder (4.8 cm inner diameter) connected to a slide hammer. Logistical and safety concerns precluded deeper sampling in an urban context, where buried utility or irrigation lines, etc., are a concern.

Soil preparation and analysis

Composite samples for chemical analysis were air dried, followed by oven drying for 24 h at 60°C. Samples were mechanically crushed to pass through a 2-mm sieve, and visible plant roots and residue discarded. For C analysis, soil samples were ground to a fine powder, and chemical analysis conducted on finely ground subsamples. Total C was analyzed by high-temperature catalytic combustion (Carlo-Erba Model NA 1500 C and N analyzer; CE Instruments, Milan, Italy). Percent C was converted to C mass for each site using soil bulk density. For P analysis, dried and ground subsamples were sent to the Soils and Plant Analysis Lab at the University of Wisconsin-Madison, where a Bray 1 extract procedure was used to analyze available P (Bray and Kurtz 1945). The Bray 1 method was generally developed as an estimate of P available to plants in agricultural systems, rather than as a reflection of P storage in soil or P runoff. However, Bray 1 P is related to dissolved P in runoff in several systems (Sharpley 1995, Ebeling et al. 2008, Wang et al. 2010), Bray 1 P is well correlated with total P in our system (Bennett 2003). Thus, Bray 1 P is likely to provide a good estimate of the sorbed P that accumulates in soils. Bulk density samples were air dried, followed by oven drying at 70°C for 48 h, and weighed to determine soil mass on a dry mass basis. Following bulk density determination, we analyzed samples for soil particle size. Sand, silt, and clay content were measured using the hydrometer method, which uses sedimentation rates based on Stoke's law (Bouyoucos 1962) (Appendix S1: Table S2), and soil texture assigned according to the USDA major textural classes (Appendix S1: Fig. S2). A pedotransfer function (Saxton and Rawls 2006) was used to estimate K_s from soil texture, bulk density, and organic matter (calculated from soil C measurements, under the assumption that 58% of soil organic matter is soil organic C).

Data analysis

To evaluate effects of land cover and time since development (Q1), we used linear mixed models (using R's lme function in the nlme package) to test for significant differences in soil C (kg/m^2), P (ppm), and K_s (mm/h; significance level α = 0.05). For all models, subplots were nested within sites as a random effect. First we tested for significant differences in each ES indicator among all five land-cover classes, with C and P aggregated over depth (R Core Team 2012). Second, we tested for significant effects of land-cover class, time since development, and depth (for C and P) on soil C, P, and K_s among the four land-cover classes for which the year of development was available (i.e., excluding forests, where limited temporal data was available to assess trends). Third, we modeled each land-cover class separately to assess the effects of time since development, depth, and their interaction on soil properties within each land-cover class. Model residuals were visually inspected for normality using diagnostic plots, and log transformations were performed as appropriate to improve normality. For open space sites, we also assessed historical aerial photos to determine whether or not the sites had been in agricultural use prior to development, and tested for significant differences in soil properties between those sites previously used for agriculture compared to those that had not been.

To account for potential spatial patterns beyond those accounted for by city structure, we re-ran all models as generalized additive mixed models (using the R package mgcv) with latitude/longitude added as a smoothing term. We used a penalized latitude/longitude term, so that variance was first attributed to the fixed effects (i.e., land cover, time since development), and then to latitude/longitude if not captured by the fixed effects. The spatial term was non significant in almost all cases, and did not substantively alter results after accounting for land cover and time since development (Appendix S1). Thus, we present results of the more easily interpretable, and slightly more conservative, mixed models here. Results were qualitatively similar for all soil properties. Additionally, analyses of percent C rather than mass showed qualitatively similar results, and are not presented here.

To identify the scales at which variance was most pronounced for each soil property (question 2), we used variance partitioning to assess relative variance in soil C, P, and K_s at three levels: among land-cover classes, among sites, and within individual sites (using VarCorr in R's lme4 package). We also conducted this analysis separately by land-cover class to quantify relative variance in each soil property at two levels: among sites and within sites.

To estimate the relative contribution of each land-cover class to citywide ES (question 3), we accounted for the role of impervious surfaces, under various assumptions representing the effect of impervious cover on soil properties, followed by area-based extrapolations. Impervious surface was split into building cover and non-building impervious (e.g., roads, sidewalks, driveways, etc.), and calculated for each land-cover using QGIS. Impervious surfaces were quantified using NLCD percent impervious data (Xian et al. 2011, Haase et al. 2014), with building cover quantified using Lidar derived-building footprint data for the City of Madison. For all soil properties, we assumed a value of zero under building footprints (sensu Edmondson et al. 2014). Basements are very common in our study region, which will preclude substantial soil nutrient pools under buildings at the depths considered here. For C under non-building impervious surfaces, we considered three alternatives to assess the sensitivity of our results to varying assumptions: (1) assume equivalent soil C as in non-impervious surfaces (consistent with findings by Edmondson et al. 2012); (2) assume 50% C (consistent with studies finding lower, but still substantial C pools under non-building impervious (Raciti et al. 2012, Wei et al. 2013), and (3) assume zero C under non-building impervious (a common assumption for urban areas, and the most conservative possible bound). For P under non-building impervious surfaces, we considered two alternatives: (1) assume equivalent soil P as in non-impervious surfaces (consistent with Wei et al. 2013) and (2) assume zero P under non-building impervious surfaces (i.e., assume any P under impervious surfaces does not influence water quality). For K_s, we assumed zero infiltration for any impervious surfaces.

Results

Urban vegetation

Vegetation differed among land cover categories (Appendix S1), but these differences were not strongly correlated with any measured soil properties.

Variation in ES indicators with land cover and time since development (Question 1)

Differences among land cover classes.—All three ES indicators differed among land-cover classes (C, $F_{4,95}$ = 7.16, P < 0.0001; P, $F_{4,95}$ = 2.96, P < 0.05; K_s, $F_{4,95}$ = 4.19, P < 0.01; Fig. 2). Mean soil C (summed to 25 cm depth) ranged from 6.4 to 9.3 kg/m^2 (Table 3, original field values). Carbon density was highest in more developed land covers, including both open space and residential areas (i.e., medium- and low-density developed), and lowest in semi-natural land covers (forests, grassland; Fig. 2A). Mean available P (averaged over 25-cm depth) ranged from 38.6 to 62 ppm (Table 3, original field values). Phosphorus was highest in residential areas, lowest in open space and grasslands, and intermediate in forested areas (Fig. 2B). Mean saturated hydraulic conductivity ranged from 34.6 to 67 mm/h (Table 3, original field values). K_s was highest in forests, likely driven by relatively lower bulk density and lower clay content (Appendix S1: Table S2), followed by grasslands. K_s was lowest in open space, with residential areas intermediate (Fig. 2C).

Differences with time since development and soil depth.—Soil C density was consistently higher in shallow soils ($F_{2,859}$ = 892.94, P < 0.05; Fig. 3A–D). Soil C also varied with time since development, but that relationship differed among land-cover classes and with depth (Fig. 3A–D). Soil C increased over time in low-density developed sites, and the relationship was stronger in deeper soils (>5 cm), where C increased by ~3.3–4.4% per decade of development (Fig. 3C). Soil C also increased over time in medium-density developed sites, again showing a depth-by-time interaction, and increasing by ~3.4% per decade of development (Fig. 3D).

Soil P concentration was consistently higher in shallow soils ($F_{2,873}$ = 63.31, P < 0.0001). Soil P concentration decreased or remained constant with time since development in grassland (Fig. 3E) and open space (Fig. 3F) sites, although open spaces used for agriculture prior to development had higher P than those that were not previously in agriculture (46 ± 8.8 ppm [mean ± SE] vs. 26 ± 8.7 ppm, respectively; $F_{1,18}$ = 6.48, P < 0.05). In contrast, soil P increased with time since development in low- (by 7.4% per decade; Fig. 3G) and medium-density developed sites (by 4.9% per decade of development; Fig. 3H).

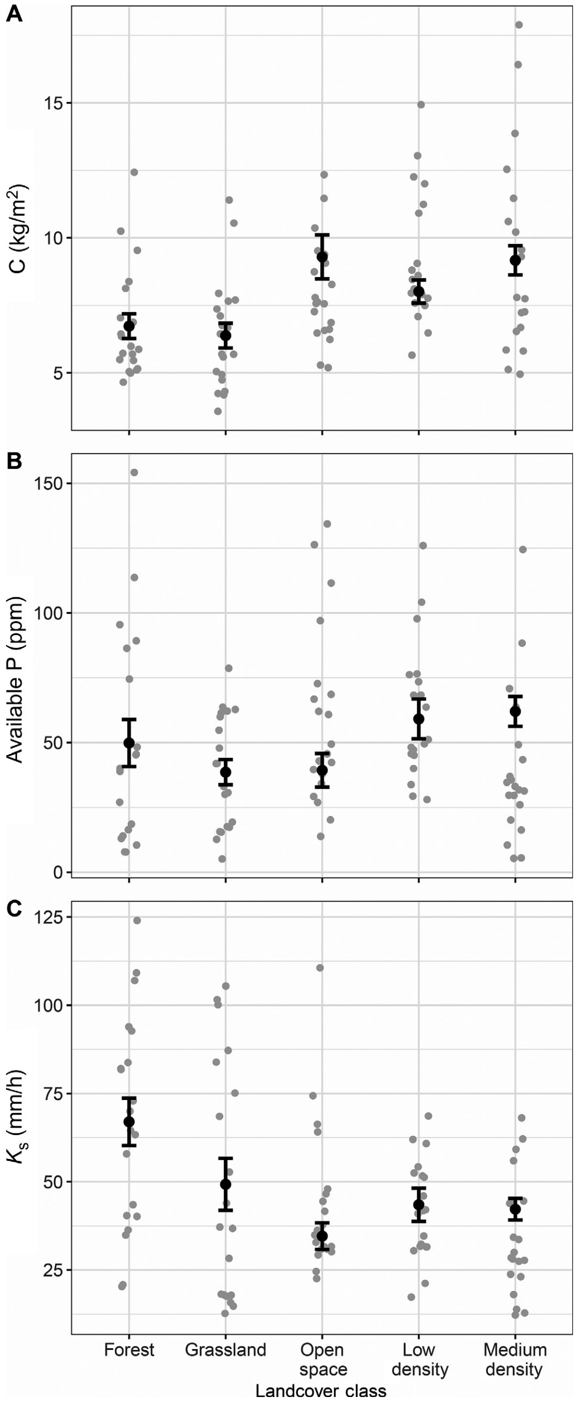

FIG. 2. Variation in biophysical indicators of soil-based ecosystem services (ES) among land-cover classes (forest, grassland, open space developed, low-density developed, medium-density developed), in Madison, Wisconsin. (A) Mean soil C (kg/m^2) summed to 25 cm depth. (B) Mean available soil P (ppm) averaged over 25 cm depth. (C) Mean saturated hydraulic conductivity K_s (mm/h) over 10 cm depth. Bars represent standard error. Gray points represent mean value of each of the 20 sites sampled per land-cover class (mean of four subplots per site).

Saturated hydraulic conductivity increased with time since development in open space sites (Fig. 3J), but showed no relationship with time since development in grasslands (Fig. 3I), low- (Fig. 3K), or medium-density developed sites (Fig. 3L). In open space sites, K_s increased by 8.3% per decade of development (Fig. 3J).

Scale of variance in ES indicators (Question 2)

Most variation in ES indicators occurred among sites of the same land-cover class, followed by within site, and last among land covers (Fig. 4A). However, variance was partitioned at different scales among land-cover classes. In forest, grassland, and open space land-cover classes (e.g., non-residential areas), most variance was among sites (broader scale) rather than within sites. In contrast, in low- and medium-density developed lands, most variance was within sites (finer scale) rather than among sites (Fig. 4B–D). These trends were consistent for all three soil properties.

Relative contribution of land-cover classes to citywide ES (Question 3)

The five land-cover classes considered encompass 76% of the City of Madison, and include the major areas of urban green space. However, land-cover classes differed in amount of impervious surface, which varied from 1% to 56% (Table 2). The relative contribution of each land-cover class to citywide ES depended on the assumed effect of impervious surfaces. Under the assumption that C is not present under buildings, but is stored in equivalent amounts under green space and non-building impervious surfaces, the relative contributions of the five land-cover classes remained consistent with measured field values (Table 3). Under the assumption that C is reduced by up to one-half under non-building impervious surfaces, open space remained the highest contributor to soil C stocks, and low- and medium density developed areas stored approximately equal C per unit area to semi-natural areas (6.0–6.2 kg/m^2 for residential areas vs. 6.3–6.6 for grasslands and forests; Table 3). Only under the most conservative scenario (assuming zero soil C stocks under any type of impervious surface), did semi-natural land covers store more C per unit area than residential areas, although open space areas still stored the most C of any land-cover class (Table 3). Similarly for P, assuming equivalent P under non-building impervious surfaces resulted in minimal change to the relative contributions of different land-cover classes. Assuming zero P under any impervious surfaces altered the relative contributions, with residential areas then storing less P per unit area than forests (27.4–38.3 ppm for low/medium-density residential vs. 48.6 ppm in forests), and medium-density developed areas in particular switching from the highest P values to the lowest (Table 3). K_s was the ES indicator most strongly influenced by impervious surfaces, with residential areas, and particularly medium-density developed demonstrating substantially lower K_s per unit area due to high impervious areas that provide no infiltration capacity (Table 3).

Accounting for the area of the city in each land-cover class, low-density developed areas account for the majority of C and P storage (~36% and 42%, respectively), and provide the greatest overall infiltration (36%) across the city (Table 4). Semi-natural areas contribute disproportionately to infiltration services, with grasslands and forests supplying 9% and 19% of K_s, respectively, despite composing only 6% and 9% of considered land area (Table 4).

Table 3. Mean values of ecosystem service (ES) indicators for five land-cover classes, under varying assumptions for the impact of impervious surfaces on ES.

	C (kg/m^2, to 25 cm depth)				P (ppm, from 0 to 25 cm depth)			K_s (mm/h, from 0 to 10 cm depth)	
Land cover	Original field values	100% under NB-IMP	50% under NB-IMP	Zero under NB-IMP	Original field values	100% under NB-IMP	Zero under NB-IMP	Original field values	0 under NB-IMP
Forest	6.7 (0.46)	6.6 (0.45)	6.6 (0.45)	6.6 (0.45)	49.8 (9.05)	49.2 (8.94)	48.6 (8.82)	67.0 (6.72)	65.3 (6.55)
Grassland	6.4 (0.46)	6.4 (0.46)	6.3 (0.45)	6.3 (0.45)	38.6 (4.87)	38.5 (4.86)	38.1 (4.80)	49.3 (7.35)	48.6 (7.24)
Open space	9.3 (0.82)	8.9 (0.79)	8.5 (0.74)	8.0 (0.71)	39.3 (6.48)	37.9 (6.24)	33.9 (5.59)	34.6 (3.75)	29.9 (3.24)
Low density	8.0 (0.43)	7.1 (0.38)	6.2 (0.33)	5.2 (0.28)	59.1 (7.69)	52.5 (6.83)	38.3 (4.99)	43.5 (4.72)	28.2 (3.06)
Medium density	9.2 (0.54)	7.9 (0.46)	6.0 (0.35)	4.0 (0.24)	62.0 (5.75)	53.4 (4.95)	27.4 (2.54)	42.2 (3.06)	18.62 (1.35)

Notes: All assume zero ES under buildings. Sub-headings refer to the assumed percentage of ES under non-building impervious surfaces (NB-IMP). Original field-derived values for each service (i.e., assuming no effect of impervious surfaces on ES) are included for reference. Values are means with SE in parentheses.

Discussion

Biophysical indicators of three soil-based ES, carbon storage, water quality regulation, and runoff regulation, revealed considerable potential for ES supply in cities. Even developed land has substantial potential for ES provision. Urban soils stored more C on average than the agricultural soils that dominate the surrounding landscape (Kucharik and Brye 2013), and less P than agricultural soils (Bennett et al. 2005) (although urban soil P was higher than in nearby remnant prairies; Bennett et al. 2005). Soil properties also varied with time since development, indicating that land-use legacies can influence contemporary

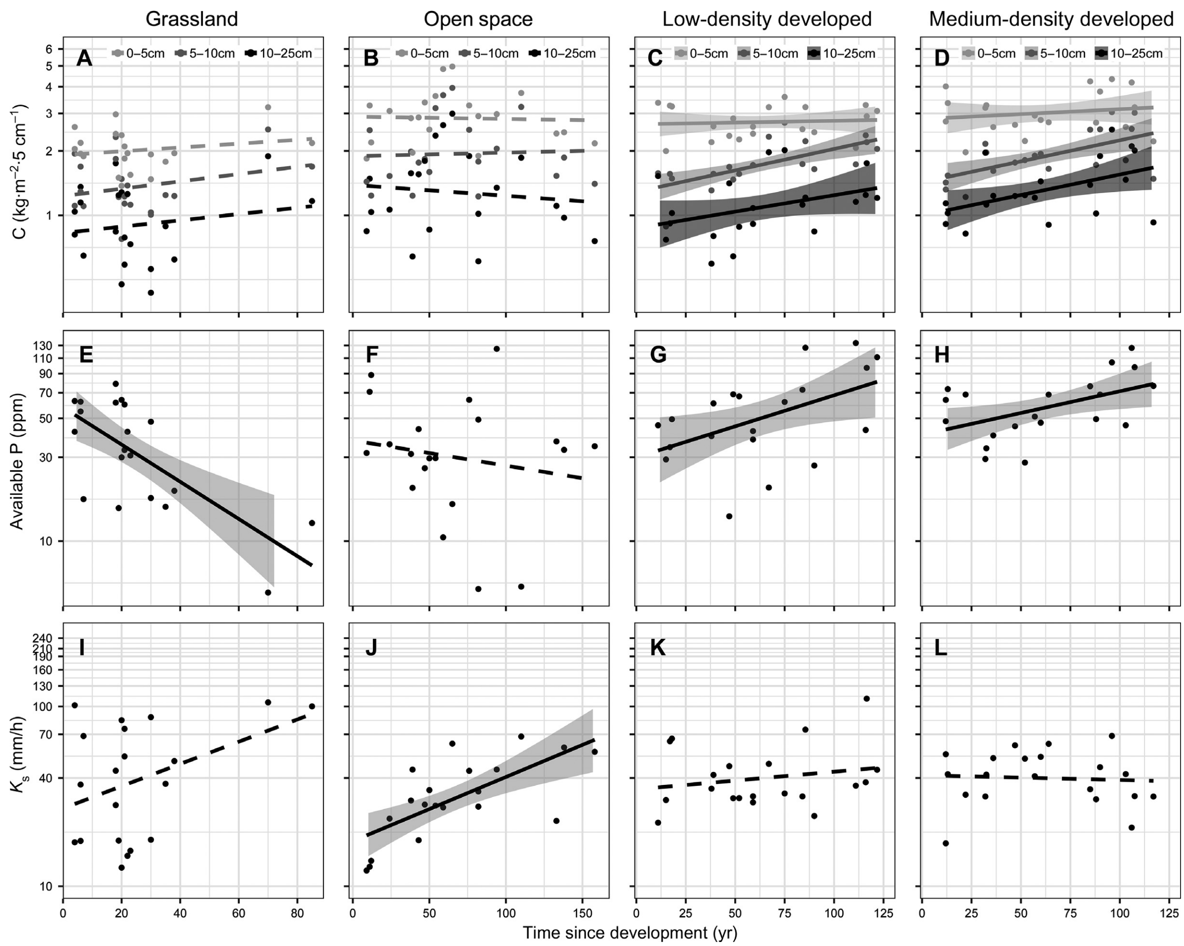

Fig. 3. Effect of time since development on three biophysical indicators of ES, carbon (C) for each 5 cm increment, available phosphorus (P) averaged over 25 cm depth, and saturated hydraulic conductivity (K_s) over 10 cm depth, across four land-cover classes in urban Madison, Wisconsin. Points represent the mean of four composite soil samples taken from each 30 × 30 m site (n = 20 sites). Lines and shading indicate the smoothed conditional mean and 95% confidence interval for the mean, respectively. Solid lines represent statistical significance. Relationships that are not statistically significant are represented with dashed lines.

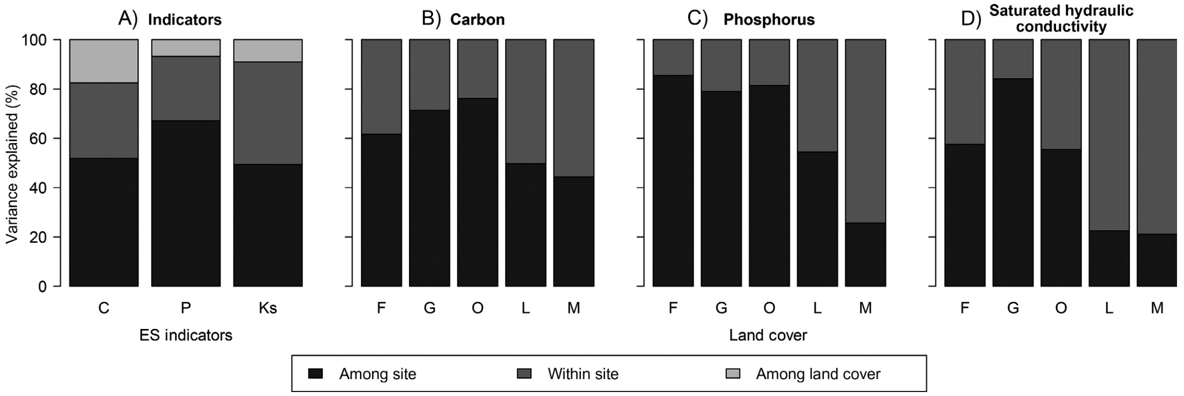

FIG. 4. Percentage of total variance in biophysical indicators of ecosystem services accounted for at scales of measurement. (A) Percent of total variance accounted for at the among land cover, among-site, and within-site scale for each of three ecosystem service indicators: soil carbon (C), soil available phosphorous (P), and saturated hydraulic conductivity (K_s). Percentage of total variance accounted for at the among site vs. within site scale for each of 5 land-cover classes for (B) carbon, (C) phosphorus, and (D) saturated hydraulic conductivity. Land-cover classes include forest (F), grassland (G), open space developed (O), low-density developed (L), and medium-density developed (M).

ES in cities. Given that urban areas comprise a substantial proportion of many landscapes (e.g., 25% of the Yahara watershed), excluding or undervaluing urban ES may lead to underestimates of regional ES.

Differences in ES indicators among land-cover classes

The higher C stocks in open spaces and residential areas relative to forests and grasslands (Fig. 2A) is consistent with previous work in urban and suburban landscapes (Golubiewski 2006, Raciti et al. 2011, Groffman et al. 2017). However, soil C did not differ between open spaces and residential areas in Madison, in contrast to other studies (Pouyat et al. 2006, Edmondson et al. 2014). High soil C in the three most developed land-cover classes studied may reflect high productivity of managed turfgrass systems (Poeplau et al. 2016), as well as irrigation and fertilization of residential yards (Groffman et al. 2016). Urbanization-driven increases in soil C stocks have the potential to increase ecosystem C stocks at regional through continental scales as cities expand (Groffman et al. 2014, 2017); however, life cycle analyses are needed to determine whether increased C stocks are offset by C emissions from management practices (e.g., mowing, fertilizer production; Strohbach et al. 2012).

The high P stocks measured in residential areas compared to other urban land covers may also be a function of household-level management (Fig. 2B). Phosphorus fertilizer was banned within Madison city limits in 2005 (Dane County Code of Ordinances, Chapter 80), but soil P from decades of prior use on residential lawns still persists in the contemporary landscape (see *Effects of land-use legacies on ES indicators*). Household pets can also increase soil P; nutrient inputs from pet waste contributed up to 76% of total P inputs in residential watersheds in Minnesota (Hobbie et al. 2017). Thus, lower P stocks in open spaces and grasslands may be explained in part by less fertilization and more removal of pet waste compared to residential yards. Exceptions to these general patterns, such as intensively managed golf courses, contribute to the high variability in open space P (Fig. 2B, gray dots). That soil P is highest in residential areas is worrisome for water quality, as the impervious cover and street density that characterize such areas can lead to high P in stormwater runoff (Pfeifer and Bennett 2011; Hobbie et al. 2017).

The relatively high K_s observed in forests and grasslands compared to developed land-cover classes is consistent with impairment of infiltration associated with urban development (Gregory et al. 2006, Zhou et al. 2008, Woltemade

TABLE 4. Relative contribution of each land-cover class to ES considering varying assumptions of the impact of impervious surfaces on ES indicators.

		Portion of total ES contributed by each land-cover class (%)					
		Carbon storage (Soil C)			Water quality regulation (Soil P)		Runoff regulation (K_s)
Land cover	City area considered (%)	100% under NB-IMP	50% under NB-IMP	Zero under NB-IMP	100% under NB-IMP	Zero under NB-IMP	Zero under NB-IMP
Forest	8.9	7.8	8.8	10.1	9.1	12.1	18.8
Grassland	5.9	4.9	5.6	6.4	4.8	6.3	9.3
Open space	23.6	27.7	29.8	32.7	18.6	22.4	22.8
Low density	39.3	36.6	36	35.2	42.8	42.1	35.7
Medium density	22.3	23	19.8	15.6	24.7	17.1	13.4

Notes: All assume zero ES under buildings. Sub-headings refer to the assumed percentage of ES under non-building impervious surfaces (NB-IMP). Percent of area considered represents percent of total city area of each land-cover class divided by the total area covered by the five considered land-cover classes, such that all columns sum to 100%. Soil properties used as ES indicators are indicated in parentheses under each ES.

2010) and higher average hydraulic conductivity compared to pasture, cropland, or urban land (Zimmermann et al. 2006, Zhou et al. 2008, Horel et al. 2015). In our system, the high K_s in forests is likely attributable to the somewhat lower bulk density and clay content in forest soils relative to other land covers (Appendix S1: Table S2). Similarly, the lower bulk density of many prairie restorations, particularly older sites that had not been tilled, may explain the higher average Ks compared to more developed areas. Semi-natural areas are less likely to have undergone the compaction that often accompanies development of homes or parks.

Differences in soil properties among urban land-cover classes were sometimes counterintuitive and associated with surprising tradeoffs. For example, open spaces had both the highest C storage (corresponding to high organic matter) and the lowest Ks (Fig. 2A,C), presenting a potential ES tradeoff where a synergy might have been expected due to the influence of organic matter on K_s (Fig. 2A,C). Additionally, residential areas were high in both C and P, presenting a C storage/water quality trade-off, but open spaces were high in C but low in P, indicating a synergy between these ES indicators (Fig. 2A,B). These results emphasize the difficulty of making assumptions about ES across land-cover types, even when vegetation is similar (e.g., residential lawn vs. public park), and point to the role of local land-use decisions in driving ES. Further, supply of a given ES does not necessarily extend to other services of interest (Bennett et al. 2009, Nelson et al. 2009, Qiu and Turner 2013). Thus, management for multiple ES in cities requires considering of the full mosaic of urban green space, including public and private land.

Effects of land-use legacies on ES indicators

Contemporary land cover was alone insufficient to predict soil-based ES in cities; historical land use had a strong effect, particularly in developed land covers. While land-use legacies are widely recognized in studies of land abandonment followed by succession, relatively few studies have considered effects of historical land use in cities (but see Dallimer et al. 2015). The increased C observed in older residential soils may represent C built up following cessation of agriculture (Raciti et al. 2011), as cultivation reduces soil C (Guo and Gifford 2002). Recovery from physical disturbance associated with construction may also play a role, with faster vegetation-driven recovery of surface soils compared to deeper soils (Fig. 3C,D; Golubiewski 2006, Johnston et al. 2016).

The steady increase of C and P with time since development in residential areas, but not in other turfgrass-dominated open spaces or grasslands, may be attributable in part to management history of irrigation, mowing, and fertilization (Groffman et al. 2016, Hobbie et al. 2017). Thus, historical land management is layered upon agricultural legacies. The 2005 P fertilizer ban may slow or halt the increase in urban soil P, but 12 years is likely too short relative to decades of P inputs to detect the benefit of this policy change (Bennett et al. 1999, Motew et al. 2017). Concomitant increases in C and P also suggests a stronger trade-off between C storage (high C) and water quality regulation (low P) in older neighborhoods than in newer developments. Relationships among ES are not necessarily static (Tomscha and Gergel 2016) and accounting for history can improve understanding of contemporary and future ES patterns.

In non-residential green spaces, past land use/land cover seemed to better predict soil properties than time since development; i.e., knowing what pre-dated the green space mattered more than knowing when it was created. City parks established prior to agricultural conversion or on already developed land (e.g., donation of an estate; Mollenhoff 2004), had lower P, but parks that were farmed previously had high soil P, consistent with agricultural legacies (Bennett et al. 1999, Sharpley et al. 2013, Motew et al. 2017). Grassland P was likely also agriculturally driven. The two oldest grasslands in our data set were prairies restored before industrial agriculture, thus avoiding high P inputs. Explanations for increased K_s with time since development in open spaces (Fig. 3J) are less clear. Many older city parks have sandy soils, as they were often developed along lakeshores. Soils in more recently developed open spaces, which are often used for organized sports, may also have been more compacted (Gregory et al. 2006). A lack of detailed data regarding management history precludes a stronger understanding of the relative roles of agricultural legacies vs. historical land management in explaining the observed trends. Future studies should consider the role of previous land-use compared to management or vegetation-driven trends over time, and also consider deeper soil samples, which may reveal further differences among land covers.

Spatial scales of variation among ES indicators and land-cover classes

Differences among land-cover classes in the spatial scales at which variance in soil properties was most pronounced offered insights into how urban lands are managed. Soil properties in forests, grasslands, and open spaces were most variable among sites, which aligns well with the typical scales at which vegetation and management vary in these urban green spaces. For example, two grassland sites may be composed of a grassy meadow and a restored forb-dominated prairie, or one site may be regularly burned while another is unmanaged (Appendix S1: Fig. S3). Contrastingly, soil properties in low- and medium-density developed areas were most variable within sites, reflecting the fine-scale heterogeneity of the residential urban landscape (Cameron et al. 2012, Polsky et al. 2014, Zhou et al. 2017). For example, a 30 × 30 m site in a residential area may contain land parcels managed by different owners or include front and back yards managed differently (Polsky et al. 2014, Appendix S1: Fig. S4). This high fine-scale variability on private land highlights the potential for individuals to improve urban ES supply (Cerra 2017). Future research should focus on fine-scale drivers of urban ES (Grafius et al. 2016), including the social factors that play important roles (Larson et al. 2015, Conway 2016, Groffman et al. 2016, Aronson et al. 2017).

Relative contributions of urban land cover classes to citywide ES

Developed land covers were of high importance regarding citywide ES, even with relatively conservative assumptions regarding the impact of impervious surfaces. Low- and medium-density developed land accounted for over 50% of C stocks in urban green space, and contained ~60% of the city's available soil P. While the latter may negatively influence potential water quality regulation, the ultimate fate of soil P is influenced by the surrounding landscape. Soil erosion (which carries P-laden soil into waterways) may be decreased in areas dominated by turfgrass and impervious surfaces, in which case soil P in developed land could be sequestered for decades, and thus high soil P could be construed as a benefit. However, this same soil P stock could also contribute to a delayed effect of land use on ES supply (Ziter et al. 2017), whereby heavy rain events or other disturbance will facilitate movement of legacy P into surface waters (Motew et al. 2017). Construction or redevelopment of impervious areas, for example, would render previously protected P susceptible to loss through erosion (Betz et al. 2005). Impervious surfaces may also contribute indirectly to greater P loss by increasing runoff, although quantifying the impact of impervious surfaces on urban hydrology is complex, and the role and importance of permeable areas remains unclear (Booth and Jackson 1997, McGrane 2016). Potential interactions between built infrastructure and multiple ES is beyond the scope of this work, but remains an area ripe for future research (Svendsen and Northridge 2012).

Less-developed green spaces also have an important role in a city and may contribute disproportionately to urban ES. For example, urban forests comprised <9% of the total land cover in Madison, but provided 19% of the citywide infiltration capacity. Consequently, the infiltration capacity of urban landscapes, while limited, is disproportionately attributable to forests (Maes et al. 2009). This reaffirms the importance of a mosaic of urban green space rather than a single solution for maximizing urban ES supply.

Conclusion

Ecosystem services can be substantial in urban landscapes, including on developed land. Considering the full mosaic of urban green space is needed to estimate the kinds and magnitude of ES provided in cities, and to augment regional ES assessments that often ignore or underestimate urban ES supply. And while land cover is important, it may be an unsuitable proxy for urban ES. The importance of past land use is recognized in ecology, but this temporal dimension is only recently considered in studies of ecosystem services and infrequently considered in urban ecosystems. Knowledge of historical land-use can improve estimates of contemporary ES and should be considered, especially in expanding urban areas. Future ES studies in urban ecosystems should also consider the fine-scale drivers of ES that can give landowners and managers further agency over the impact of their decisions.

Urban stakeholders and actors can also take an active role to increase ES provision in cities. For example, planners and developers can encourage a diverse mix of land covers (e.g., forest and prairie in addition to turfgrass) in urban green spaces to capture multiple services, rather maximizing only one service. Private land owners can encourage ES provision on their property by minimizing paved surfaces and using landscaping to increase water infiltration and carbon storage. Land owners can also take precautions to reduce soil erosion during construction, preventing legacy nutrients from being carried into waterways. Reduced fertilizer application and diligent pet waste removal also can reduce watershed nutrient loads over time. While seemingly small at the individual scale, such actions can scale well beyond single properties and contribute to more sustainable cities.

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Supporting Information

Additional supporting information may be found online at: http://onlinelibrary.wiley.com/doi/10.1002/eap.1689/full

Data Availability

Data available from the Dryad Digital Repository: https://doi.org/10.5061/dryad.5pr17.