



Soil organic carbon stock in different urban land uses: high stock evidence in urban parks

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Abstract

Urban areas are major producers of atmospheric CO₂ emissions, but at the same time they can offset some of the associated C losses by retaining stable organic carbon in their soils (SOC). Despite the importance of urban SOC there are still many uncertainties associated, resulting in low accuracy of SOC stock estimations or in its neglectation in the regional and national carbon budgets. We focused on one of the most extended urban areas of Italy (Milan) estimating SOC stock in the topsoil and comparing it for different urban land uses (park/non-park) and covers (woodland/grassland). We also compared urban SOC stocks with other typologies of natural and semi-natural land use in the region. We found generally high variability in urban soil properties. Average SOC of urban parks was higher ($7.9 \pm 2.4 \text{ kg m}^{-2}$) than urban non-parks ($5.3 \pm 2.5 \text{ kg m}^{-2}$); while SOC stock did not significantly differ for urban land cover types. Urban parks had SOC stock values comparable with forest, pasture and grasslands of the same region and higher than SOC of croplands. At the same time, urban non-parks had the lower SOC stocks values found in the region. The results highlighted the significant contribution of urban SOC stock to carbon budget estimations and differences based on urban land use types. Furthermore, we described the main pedological characteristics of investigated urban soils and we identified the factors that contribute to limiting our capacity to model urban SOC stock that can be partially overcome by the addressed strategies.

Keywords Urban region · Regulatory ecosystem services · Climate regulation · Urban forest · Urban soil · Milan

Introduction

Natural landscapes have been globally altered under the pressure of land use conversion, in particular due to the intensification of agriculture and urbanization (Hester and Harrison

2010). The conversion of natural to managed ecosystems makes lands more susceptible to soil degradative processes, with a consequent depletion of soil carbon stock and release of CO₂ and other greenhouse gases (GHGs) into the atmosphere (Lal 2008). Because current policies indicate that there is an urgent need to identify strategies for stabilizing atmospheric concentrations of CO₂ (WMO 2006), carbon stock provided by soil represents a key contributor to regulating climate change (Morel et al. 2015). Moreover, compared to other compartments (e.g., vegetation), carbon stock stored in soils has a long residence time, making it suitable for effectively offsetting atmospheric emissions in the long term (Lorenz and Lal 2015; Hutyra et al. 2014; Hansen et al. 2013).

The paradox faced by urban soils is that there is a great deal of interest in them regarding property and building issues, but they are almost totally ignored regarding their ecological function and role in the management of urban ecosystems (Morel et al. 2015). In fact, despite their importance in regulating ecosystem service (MEA 2005), soil organic carbon (SOC) stocks in urban areas have rarely been quantified (Edmondson et al. 2012; Raciti et al. 2012; Jo and McPherson 1995); instead, most soil carbon studies have focused on agricultural and natural ecosystems (Li et al. 2012;

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Don et al. 2011; Guo and Gifford 2002). To date, there are still significant research gaps that constrain our general understanding of the effects of urbanization on soils (Elmqvist et al. 2008), and this in turn strongly limits our capacity to model and predict their chemical-physical properties, with consequences for proper land use management and planning. There are several reasons why sufficient consideration has not been given to urban soils. On the one hand, urban soils suffer the misconception that their quality is so compromised that their contribution to carbon storage could be considered negligible (Tao et al. 2015). On the other hand, the majority of studies on carbon sequestration in urban areas have primarily focused on net primary production by urban vegetation (Morel et al. 2015; Lorenz and Lal 2009; Pouyat et al. 2006), ignoring the contribution of the soil compartment.

Compared to natural soils, urban soils are strongly affected by human activities (such as sealing, mixing, and incorporation of novel anthropogenic materials, and vegetation management practices) that alter their properties and functionality (such as compaction; perturbations in structure, texture, distribution of rock fragments, and C and N content; reduction of organic matter input) (Lehmann and Stahr 2007; de Kimpe and Morel 2000; IUSS Working Group WRB 2015). Soils whose properties and functions are profoundly modified and dominated by long-term technical human activity are classified as *Technosols* (IUSS Working Group WRB 2015). These soils may contain significant amounts of artefacts or be sealed by technic hard rock and include soil from waste, pavement with its underlying unconsolidated materials, soil with geomembranes, and soil constructed from human-made materials. Urbanization that influences the natural soil template through disturbance results in urban ecosystems with characteristics that are fundamentally distinct from their nonurban counterparts (Pavao-Zuckerman 2008; Kaye et al. 2005). As a consequence, the function of urban soils and their ability to provide ecosystem services are different from those of natural soils and often impaired (Morel et al. 2015).

In cities, urban parks are important natural assets (Bae and Ryu 2015) and are major components of carbon sequestration strategies (Strohbach et al. 2012). However, little effort has been made to quantify SOC stocks in urban parks (Bae and Ryu 2015) and determine whether they stock significantly different amounts of carbon compared to other urban land uses.

Kyoto Protocol signatories are required to provide accurate inventories in national estimates of carbon storage (Edmondson et al. 2012). From this perspective, carbon storage by soils can be directly measured or predicted with deterministic models (Morel et al. 2015). A common approach to estimating terrestrial carbon stocks is to use models based on the assumption that the only changes in carbon stocks are due to changes in land cover (e.g., the InVEST Carbon Storage and Sequestration model developed by the Natural Capital Project team) (Tallis et al. 2013; Daily et al. 2009). For example, these models

allowed production of maps of carbon stocks and comparisons of different areas and historical land cover maps (Tao et al. 2015). The required input data are estimates of the amount of carbon stored in a given carbon pool (e.g., soil organic matter) for each land cover type (Chan et al. 2006). This implies that the results of the model strongly depend on the accuracy of land use and land cover maps and above all on the reliability of OC stock estimations for each land use and land cover type (Muñoz-Rojas et al. 2011). However, due to the unique features of urban soils and their high heterogeneity, accurate estimates of OC stock for different urban land use and land cover types are not available or are associated with high uncertainty. Studies revealed that assumptions underpinning current regional or national estimates of ecosystem OC stocks are not robust and are likely to have seriously underestimated the contributions of urban areas (Edmondson et al. 2012).

Current studies intending to model carbon stocks in urban areas often use as input data information available from the literature (Tao et al. 2015; IPCC 2006), based on the assumption that the value reported for soils in a certain city may be considered representative of another, as well as for all the urban land use and land cover typologies. However, to date, unfortunately there are still no positive answers to questions like: i) are there significant differences in SOC stock for different typologies of urban land use and land cover (e.g., parks, tree lines, vacant land)? ii) is it possible to predict and model differences in urban stocks based on urban environmental parameters? Based on these uncertainties, it is reasonable to ask what current evaluations based on literature data (sometimes taken from different geographical regions that can be minimally reliable for local context) actually estimate.

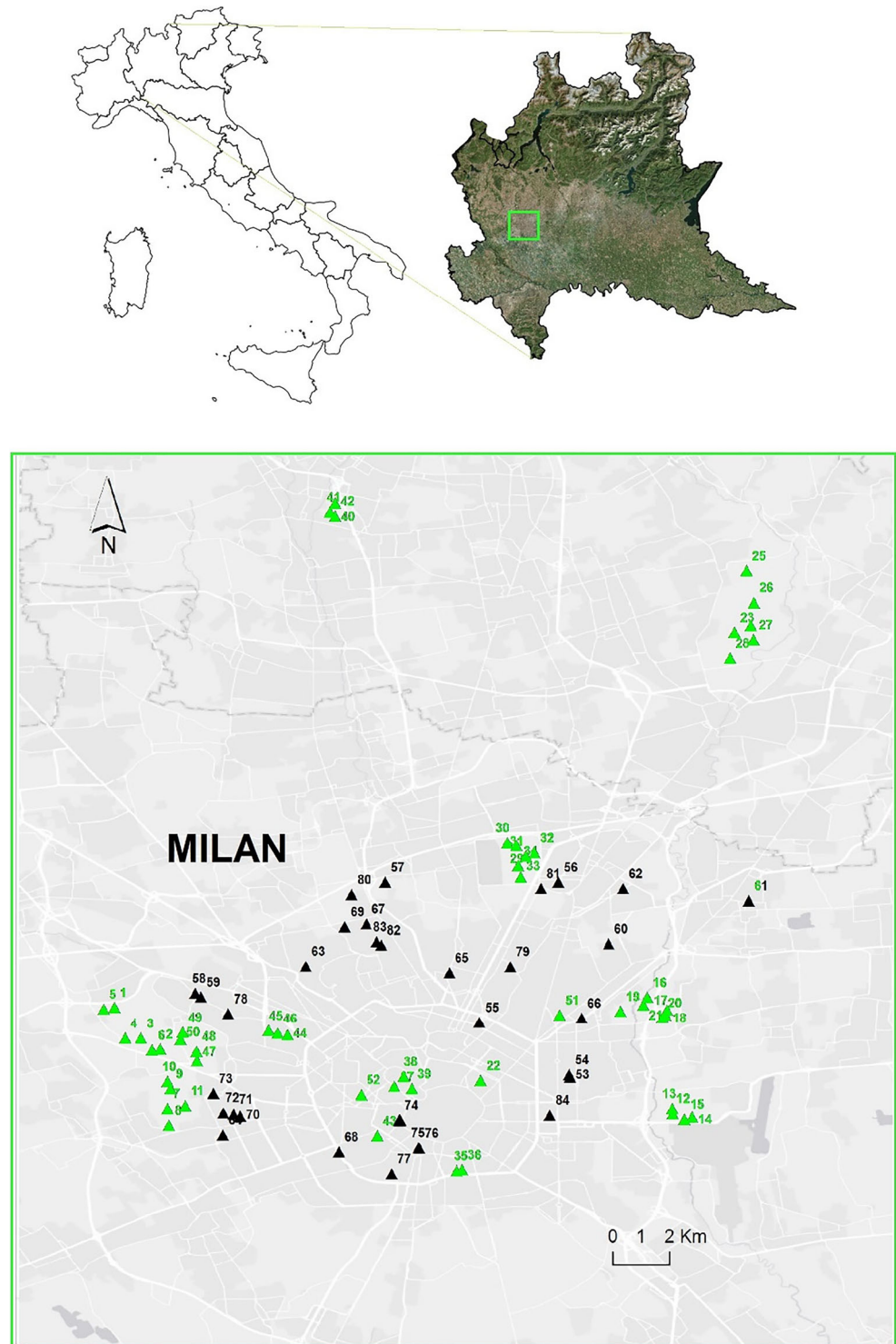
Although we acknowledge that the need to compile evaluations for large areas allows for approximation of some fine-scale parameters, we point out the necessity to reduce the degree of uncertainty associated with carbon stock estimations in urban soils. For these reasons, in this study we: (1) described the main chemical-physical properties of urban soils in one of the most extended urban area of Italy (Milan); (2) estimated SOC stock in the topsoil and comparing it for different urban land uses (parks and non-parks) and land covers (wooded and grassland areas) and with other typologies of land use surrounding the city (cropland and permanent grassland); 3) discussed the main constraints that limit our capacity to model urban soil properties and which issues should be addressed to enhance it.

Materials and methods

Study area

In Italy, the carbon stock of the Lombardy region (Fig. 1) was estimated in a regional monitoring project on climate change and greenhouse gases by the Regional Agency for Agriculture

Fig. 1 Study area and location of sampled sites. Triangles' colour indicates urban parks (green) and non-parks (black)



and Forests of Lombardy (ERSAF). As has happened with other inventories, urban areas were excluded from the regional carbon stock estimation mainly because of a lack of reliable data (Edmondson et al. 2012). This led the study area (but it is common to other realities, see Don et al. 2011; Guo and Gifford 2002) to the presence of empty spaces in regional soil monitoring systems and a consequent deficit in the total

regional carbon budget estimation. Such errors became particularly relevant if we considered that urban land cover may represent a significant percentage of the territory. In Lombardy, urban areas cover approximately 14% of the territory. On the average, urban land (soil surface covered with an impermeable material) is estimated to cover 9% of the European area (Scalenghe and Marsan 2009).

Milan (45°27' N, 9°11' E; Fig. 1) and the surrounding metropolitan area is one of the most extended and densely populated urban areas of Italy (2.053 inhab/km²). The city of Milan is situated in the central-western part of the alluvial Po Valley. The topography of the territory is nearly flat, with an average altitude of 100 m. Green spaces within the city cover an area of 25 km², representing almost 14% of the city. Milan has a typical continental climate, with yearly average rainfall of 920 mm and average annual minimum and maximum daily temperatures of 8 °C and 17 °C, respectively. The metropolitan area experiences a 2–3 °C higher rise in temperature than would be normal due to the urban heat island phenomenon. Soil types within the region are dominated by Luvisols (mainly Dystric and Gleyic) and Cambisols (mainly Skeletic and Gleyic).

Data collection

Urban soil sampling

Soil sampling was performed during 2014–2015, and samples were collected using a gouge auger (Eijkelpamp; 2.5 cm diameter) in 84 georeferenced plots (Fig. 1) randomly selected from the land cover map of the study area using the land cover category “Artificial, non-agricultural vegetated areas” (which comprised unpaved green areas such as parks, vacant sites, vegetated urban squares, private gardens, tree alleys, etc.). Within each plot, sampling sites were selected (from 1 to 6, depending on the size of the area). In each site, a squared area of 30 × 30 m was defined and soil was taken in 5 subsamples at the center and corners of the area (see Fig. 2 for sampling design). The soil was sampled at different depths (0–10 cm, 10–20 cm, and 20–40 cm, labeled layer I, II, and III, respectively). Many studies differentiate between topsoil and subsoil by depth (De Vos et al. 2005), here we take the subsamples by predefined layers, meaning that there was a uniform sampling depth between sites and the number of samples per layer taken at a given location was always 3 (except for locations with a soil depth <40 cm). For each site, we recorded soil sampling depth when it was lower than 40 cm.

Soil bulk density (BD) was determined for the first layer with the cylindrical core method (core diameter 5.4 cm) on undisturbed core samples, considering the volume of stones, when present. In order to estimate bulk density for layers II and III, we used a pedotransfer function for compacted subsoils (Hollis et al. 2012; Eq. 1) and a pedotransfer function for horizon A (Hallett et al. 1998) for layer I (Eq. 2) when field data were not available.

$$\begin{aligned} \text{BD} = & 1.1257 - (0.1140245 \times \text{Log}_e(\text{SOC})) \\ & + (0.0555 \times \text{Log}_e(\text{Horizon midpoint})) \\ & + (0.002248 \times \text{Sand}) \end{aligned} \quad (1)$$

Where BD is expressed in g cm⁻³, SOC is the percentage of organic carbon, horizon midpoint is the mean depth of the layer (in cm), and Sand is the percentage of sand.

$$\begin{aligned} \text{BD} = & 0.870 + (0.0710 \times \text{Log}_e(\text{Clay})) \\ & + (0.0930 \times \text{Log}_e(\text{Sand})) - (0.254 \times \text{Log}_e(\text{SOC})) \end{aligned} \quad (2)$$

where Clay is the percentage of clay, Sand is the percentage of sand, and SOC is the percentage of organic carbon.

Through BD and stone volume, SOC stock on an area basis (kg m⁻²) was computed for each mineral layer by Eq. (3):

$$\text{SOC stock} = (m \times C) / 100 \quad (3)$$

where m is the fine earth (kg), C is the carbon content (in %). M was calculated as:

$$m = \text{BD} \times (V_{\text{tot}} - V_{\text{sk}}) \quad (4)$$

Where BD is the bulk density, V_{tot} is the total volume (dm³) sampled and V_{sk} the volume of rock fragments (dm³).

SOC stock of other land uses

In order to evaluate OC stock in urban soils, we compared it with soils of surrounding suburban areas that belong to similar soilscapes. We selected suburban areas within a buffer of 20 km around the city of Milan. Data were taken from the available literature (technical reports; D'Alessio and Febelli 1999; D'Alessio et al. 1993). The reports analyzed carbon stock for the topsoil of the main land use typology around Milan. We selected the data related to croplands and permanent grasslands.

Soil physical and chemical analyses and environmental variables

The collected soil samples were air-dried and sieved (2 mm mesh). We measured SOC concentration by elemental analyzer (Flash EA 1112 NCSOil, Thermo Fisher Scientific CN elemental analyzer, Pittsburgh, PA, USA) after carbonates removal by treatment with HCl. Black soil organic matter (black SOM) was analyzed by thermogravimetric analysis (TGA) using 375 °C and 550 °C as thermal thresholds to separate SOM, black SOM, and carbonates (Plante et al. 2009).

Statistical analyses

For each variable, a preliminary exploratory data analysis was performed. A principal component analysis (PCA) was carried out to investigate variable patterns. Soil and environmental variables used in the PCA were OC stock, C/N, pH in water, BD, soil thickness, available P, texture parameter of cSa, fSa, Silt and Clay, heavy metals, and black SOM.

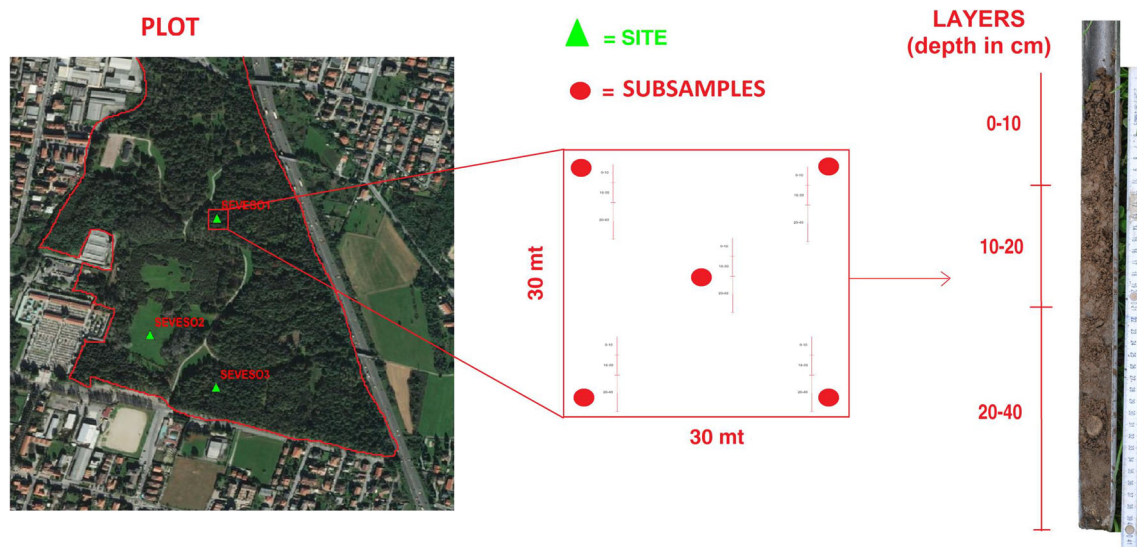


Fig. 2 Sampling design: in each 30×30 m site, five subsamples of three different layers were collected

To detect differences in urban SOC stocks, three comparisons were made. First, we considered two urban land use categories: urban park and urban non-park. The former was defined by its land use, the latter designated all the green investigated areas that were not parks and were vacant sites, green squares, private gardens, tree lines, or street greens. Secondly, we compared wooded and grassland land covers within urban parks considering as woodlands the sites, covered more than 50% by trees (in a $50 \text{ m} \times 50 \text{ m}$ area around the center of the sampling site); otherwise the investigated areas were considered grassland. A third comparison was made between urban soils and soils of suburban areas. Here, soil land use categories were croplands and permanent grasslands. In order to test the effect of different land use and land cover on SOC stocks, linear mixed models (LMMs) were performed using the PROC MIXED procedure in SAS (release 9.4; SAS Institute, Cary, NC, USA) (Littell et al. 2006). The evaluation of the response variable in relation to land use or land cover was carried out considering land use and land cover as fixed effect and territorial ambits as random effect in the linear mixed model; territorial ambits were defined based on the area they belonged to, in the case of urban soils (e.g., park, street green), or on the soilscape unit for croplands and permanent grasslands. In modelling the relationships between SOC stock and land use in the comparison between urban and nonurban soils, residual spatial correlation was found and thus the spatial model was used, consisting in adding a stochastic element, described by a parametric covariance function of residuals, to the deterministic component. The simultaneous estimates of covariance parameters and fixed effect coefficients were obtained by restricted maximum likelihood (REML) estimation (Littell et al. 2006), to consider the loss of degrees of freedom due to the estimation of the fixed effects coefficients (Cressie 1993); for variance components (fixed effects) the null hypothesis of no effect

was tested using the fixed effects F-test (Morgan and Gumpertz 1996) with the corresponding residual variance (spatial or non-spatial model). In more detail, the spatial covariance function of residuals was determined iteratively by estimating the partial sill, range, and nugget effect parameters (using the statement REPEATED). The differences between mean values were obtained through contrast analysis.

Statistical analyses were performed with Statistica (StatSoft version 10) and SAS (release 9.4, SAS Institute).

Results

Urban soil properties and determinants of C stock

The characterization of the investigated urban soils of Milan is presented in Table 1 (for a detailed description see Supplementary Materials). As described in other studies (Morel et al. 2015), urban soils are characterized by high heterogeneity in their properties, and they ranged from slightly modified to very intensively managed and disturbed. In general, Milan soils showed pronounced complexity (intended as the amounts of different matter within a given volume) and heterogeneity (differences in composition and arrangement) (Byrne 2007) among the investigated areas and different sites in the same area (i.e., adjacent sites in the same park). According to natural trend, the SOC stock generally decreased with soil depth increasing. Maximum thickness of sampling was set at 40 cm, but in some sites soil thickness was lower, reaching the minimum thickness of 10 cm. The amount of black OM in Milan urban soils decreased from topsoil to subsoil.

The first two components of the PCA explained 50% of the total variance (Fig. 3). The first component, which explained

Table 1 Descriptive statistics of soil data. Layer = depth in cm. SOC stock: organic carbon stock; pH: pH in water; SOC: soil organic carbon; SOM: soil organic matter; Black SOM: black soil organic matter

	Layer	Mean	s.d.	Min	Max
Depth		35	8.7	10	40
SOC stock (kg m ⁻²)	0–10	3.11	1.16	0.75	6.48
	10–20	1.92	0.59	0.7	3.36
	20–40	2.56	0.87	0.81	5.07
	0–40	6.93	2.8	1.32	12.88
SOC (g kg ⁻¹)	0–10	29.23	8.68	8.12	58.93
	10–20	16.43	5.21	6.34	32.24
	20–40	11.7	4.39	5.48	25.54
	0–40	19.6	9.88	5.84	58.93
SOM (g/kg ss)	0–10	72.27	18.14	28.40	130.13
	10–20	49.26	11.49	21.97	82.08
	20–40	41.43	8.73	26.09	64.83
	0–40	55.28	18.98	21.97	130.13
Black SOM (g/kg ss)	0–10	16.49	3.15	10.45	23.58
	10–20	14.83	3.00	9.35	23.14
	20–40	14.04	2.68	9.59	21.41
	0–40	15.20	3.12	9.35	23.58
Black SOM/SOM	0–10	0.23	0.04	0.15	0.37
	10–20	0.31	0.04	0.20	0.43
	20–40	0.34	0.04	0.28	0.45
	0–40	0.29	0.06	0.15	0.45

31% of the variance, was mainly represented by SOC, N content, percentage of silt in the texture and thickness of soil sampled (positively correlated each other and negatively correlated with the content of sand and pH). The second component explained 19.19% of the variance and was mainly represented by heavy metals that were positively correlated with the content of black SOM. Soils with higher SOC content showed also higher N content, a soil texture characterized by high silt content, and higher soil thickness. Soils with lower SOC stock had a high sand texture content, high C/N and high black SOM.

Comparison of SOC stock for different land uses and land covers

SOC stock in urban soils was highly variable among investigated sites and areas. Results of the mixed models showed significant differences in SOC stock among different urban land uses: urban parks were found to store more SOC than non-urban parks. Land cover (wooded areas or grasslands) did not determine differences in SOC content (Tables 2 and 3).

By comparing urban SOC stock with other land uses of adjacent areas, some significant differences emerged (Table 4): SOC stock in urban parks was comparable to that of croplands and permanent grasslands. A decrease in the SOC stock was observed ranging from soils of permanent

grasslands to urban parks to croplands and finally to urban non-parks, which own the lowest content of SOC.

Discussion

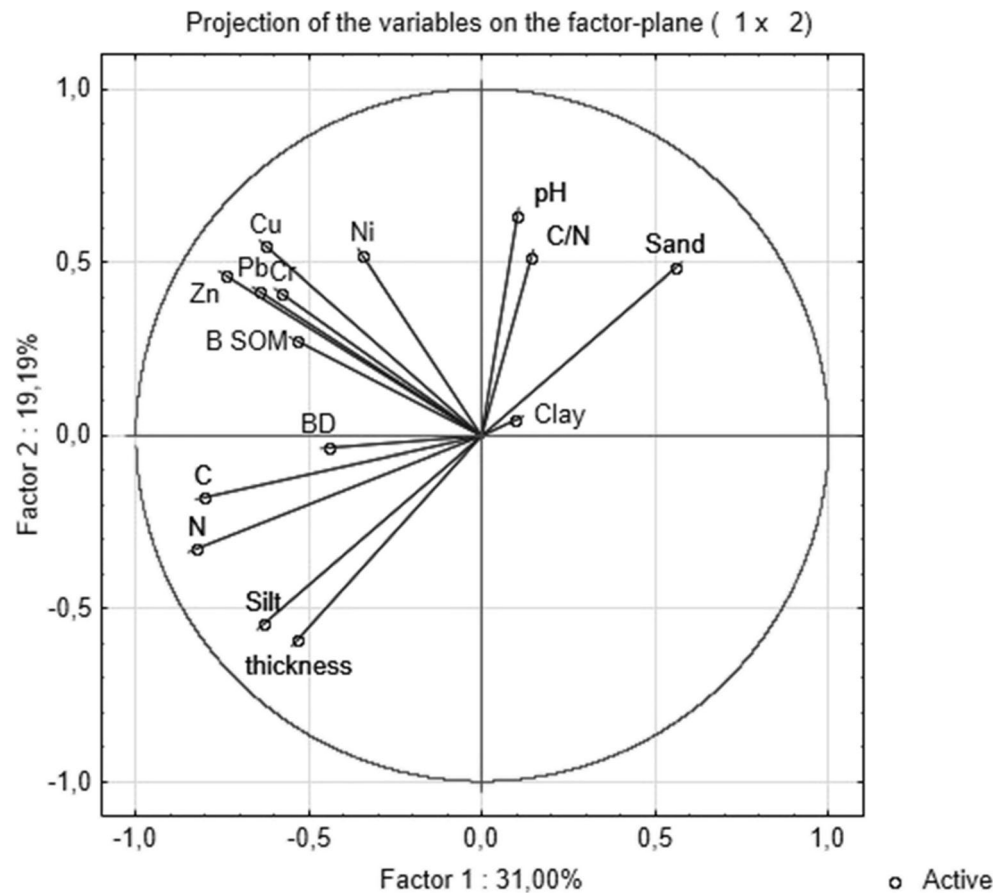
Characteristics of urban soils

Urban soils analyzed in this study were frequently characterized by incorporation of anthropogenic coarse material like bricks, mortar, concrete, slag, or various types of garbage; we found other uncharacterized materials that may be related to industrial waste, processed oil products, ash or sludge, and sewage (Rossiter 2007) (Fig. 4). Materials imported while urban soils were under construction (including pollutants, garbage, fertilizers) may affect the biogeochemical cycling of C and N in various ways (Lorenz and Lal 2009). However, any generalization of observed effects of the urban environment on biogeochemical C and N cycles remains highly uncertain (Pickett et al. 2001) because of the scant ecological information about urban areas (Byrne 2007).

Soil depth is a remarkable characteristic of urban soils. In this study, we found some soils with a depth of less than 20 cm or less than 10 cm. In these cases, the soil layer was often abruptly interrupted by a thick layer composed of anthropogenic material (artifacts, bricks, and concrete). Shallow soils were more frequently found in urban non-park sites, but cases of urban park soils, thinner than 40 cm, were also observed. Depth directly determines the amount of matter in a certain space, calculated as the surface multiplied by the thickness (i.e., kg of SOC per m⁻² for a standard depth). Many estimations of soil properties (e.g., SOC stock) are calculated for standard depths (the most common are 30, 40, and 60 cm). However, urban soils have often thickness smaller than these (still thin) layers, and shallow soil depths are commonly reported in other urban areas (Yan et al. 2015). This has to be taken in consideration when estimate OC stock in urban areas for standardized depth (i.e. for 40 cm), because the presence of thin layers can add bias, for example by overestimating the total volume of soil at city scale. Moreover, soil depth is an important parameter for defining other soil properties such as soil fertility or other soil functions (root expansion, water retention capacity, contaminant filtering).

The presence of considerable amounts of coarse materials that we found resulted in a reduction of fine earth volume. Bulk density of soils may be considered low for values lower than 1.0 g cm⁻³ and high when it exceeds 1.6–1.7 g cm⁻³, for example (Hazelton and Murphy 2007). Despite soil compaction being a common problem in urban areas making it reasonable to expect high values of soil density, we found very low bulk density in many sites (e.g., 0.61 and 0.83 g cm⁻³ in an urban park, 0.62 g cm⁻³ and 0.74 g cm⁻³ in a nonurban park) and although a certain variability was observed, the mean value was of 1.29 g cm⁻³ (s.d. 0.12). Soils with low

Fig. 3 Loading plot of the PCA. C = OC content; N = N content; C/N = C N ratio; Thickness = thickness of soil sampled; Sand = total sand content; Silt = total silt content; Clay = total clay content, B SOM = black organic matter content; Zn, Pb, Cu, Cr and Ni = heavy metals



bulk density values were generally characterized by limited soil depth (less than 40 cm) and heavy inclusion of coarse anthropogenic materials, probably related to recent mixing perturbation. Similar studies on urban soils reported mean values of 1.39 and 1.73 g cm⁻³ (Scharenbroch et al. 2005) or substantially lower (0.97 g cm⁻³) (Edmondson et al. 2011).

Organic carbon stock

Previously, a lack of data necessitated the assumption that soils in urban areas are so functionally compromised that they are unable to store carbon (Edmondson et al. 2012). However, as we found in this study and as the literature highlights (Lorenz

Table 2 Mixed model results: estimates and their standard error of the fixed effects. The response variable is the SOC stock and the fixed effect is land use (urban park: URB-P; urban not-park: URB-NP; woodland urban park: WOOD; grassland urban park: GRASS; agricultural arable

lands: ARABL; permanent grassland: PERM GRASS). Spatial/Spher: the spatial covariance function of residuals was the Spherical model; non-spatial: the model residuals were spatially uncorrelated. Significant regressors (*P* < 0.05) are in bold. * define gaussian variable

SOC stock		Model type	Parameter	Statistics		
				Estimate	SE	Pr > t
Urban soils	Non-spatial	Interc	7.4944	0.4870	<.0001	
		URB-NP	-2.0100	0.6974	0.0049	
		URB-P	0	.	.	
Urban park soils	Non-spatial	Interc	7.5372	0.5386	<.0001	
		GRASS	-0.1221	0.4544	0.7889	
		WOOD	0	.	.	
Urban and not-urban soils *	Spatial/Spher (range: 3.7 km)	URB-P	0.4558	0.2235	0.0462	
		URB-NP	-0.2613	0.1724	0.1453	
		ARABL	-0.6587	0.2252	0.0043	
		PERM GRASS	0.09871	0.1991	0.6218	

Table 3 Mixed model results: estimates and their standard error of the covariance parameters

SOC stock	Statistics				95% Conf. Int.	
	Parameter	Estimate	SE	Pr > t	Lower bound	Upper Bound
Urban soils	Terr. Amb.	3.4805	1.1653	0.0014	1.9830	7.6432
	Residual	3.1136	0.4518	<.0001	2.3882	4.2301
Urban park soils	Terr. Amb.	3.8775	1.6911	0.0109	1.9208	11.5261
	Residual	3.3914	0.5238	<.0001	2.5602	4.7072
Urban and non-urban soils *	Terr. Amb.	0.1444	0.08333	0.0415	0.06001	0.6992
	Partial sill	0.5319	0.1214	<.0001	0.3560	0.8807
	Residual	0.3617	0.06461	<.0001	0.2622	0.5311

and Lal 2015; Tao et al. 2015; Edmondson et al. 2012; Pouyat et al. 2006; Pavao-Zuckerman 2008) urban soils may (or may not) store significant amounts of organic carbon.

Generally, the concentration of SOC in urban soils is reported to be highly variable (Lorenz and Lal 2009; Pouyat et al. 2006; Scharenbroch et al. 2005). In Milan urban soils, OC concentrations at a depth of 0–40 cm varied between 3.0 and 24.8 g C kg⁻¹ (Table 1). SOC stock, expressed as kg of organic carbon stored in 1 m² ranged from 1.3 to 12.9 in Milan (Table 1). Other studies reported varying but comparable values (Table 5). A great limitation in comparing SOC stock of different urban soils from other studies is the difference in sampling methods applied, in particular the soil depth reference used to express OC. For this issue, a standardize methods will help in make more reliable comparison among different geographical areas of the world.

By comparing different land uses (park and non-park), we found that urban parks stored significantly higher SOC stock (Fig. 5). However, vegetation cover was not found to influence carbon stocks, in contrast to what often happens for natural soils (Don et al. 2011; Yimer et al. 2006). Apparently, a functional classification of urban spaces (based on land use type) can be more useful to distinguish soils with similar characteristics (at least regarding carbon stock) than a structural classification (based on land cover types or vegetation characteristics). A recent study (Bae and Ryu 2015) explained

differences in urban SOC stocks based on different land cover types (forests, wetlands, lawns, etc.); however, comparisons were made within a single park, representing a slightly different comparison than ours (intra-park vs. inter-park). Considering that several factors work synergistically to influence SOC stock in urban areas and that most of these may be attributed to human activities, it is expected that we will find patterns that are altered compared to natural systems, included a low explanatory power of vegetation cover. Among these, we can include the fact that the urban soils that we have studied may have experienced for example mechanical disturbance in recent years (few years or decades) and the effect of vegetation on SOC may not yet be appreciable.

In general, the urban SOC stock of Milan (mean 6.9 kg m⁻²) was comparable with stocks found in croplands in Lombardy (5.7 kg m⁻² in the first 30 cm) (D'Alessio and Febelli 1999; D'Alessio et al. 1993) and a bit lower than those found in natural lands (forest, pasture, and grasslands: 8.0 kg m⁻² in the first 30 cm). Average SOC stock for the Lombardy region was reported to be 6.9 kg m⁻² in 0–30 cm depth, and it was 5.8 kg m⁻² for the Po plain, where the city of Milan is located. Thus, urban soils of Milan showed a slightly higher average OC stock than that found in Lombardy region. The analysis comparing urban soils with adjacent agricultural soils and permanent grasslands showed that SOC stock of urban parks was comparable to that of

Table 4 Contrast analyses

SOC stock	Land use	Statistics			
		Estimate	SE	Pr > t	
Urban soils	URB-NP	URB-P	-2.0100	0.6974	0.0049
Urban park soils	WOOD	GRASS	-0.1221	0.4544	0.7889
Urban and non-urban soils *	PERM GRASS	ARABL	0.7172	0.1964	0.0003
	PERM GRASS	URB-NP	1.1146	0.3157	0.0007
	PERM GRASS	URB-P	0.3571	0.2982	0.2358
	ARABL	URB-NP	0.3974	0.2802	0.1615
	ARABL	URB-P	-0.3600	0.2567	0.1693
	URB-NP	URB-P	-0.7574	0.2298	0.0016

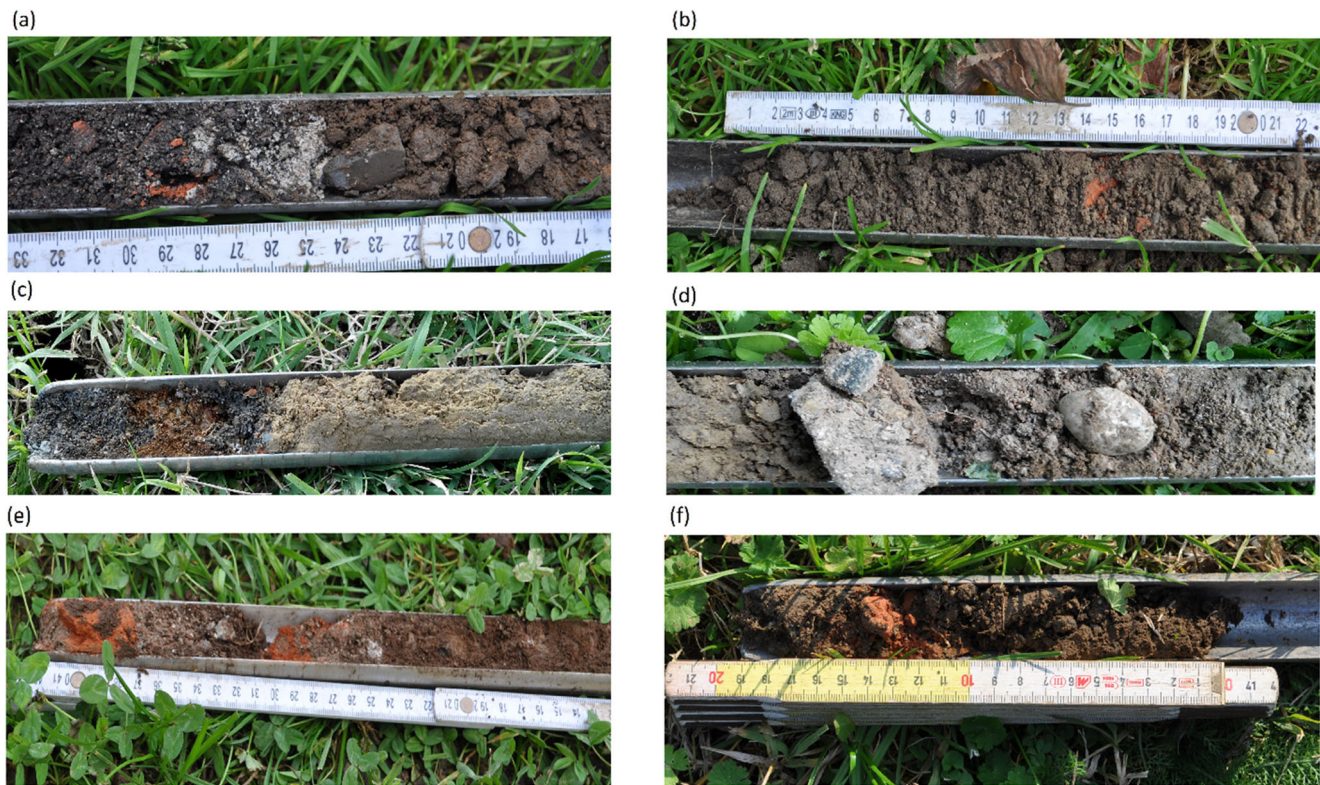


Fig. 4 Examples of soil samples belonging to urban parks (a–d) and urban non-parks (e, f)

forest, pasture and grassland, while urban non-park soils showed the lowest values. These results confirm the nonnegligible contribution of urban soils to carbon storage.

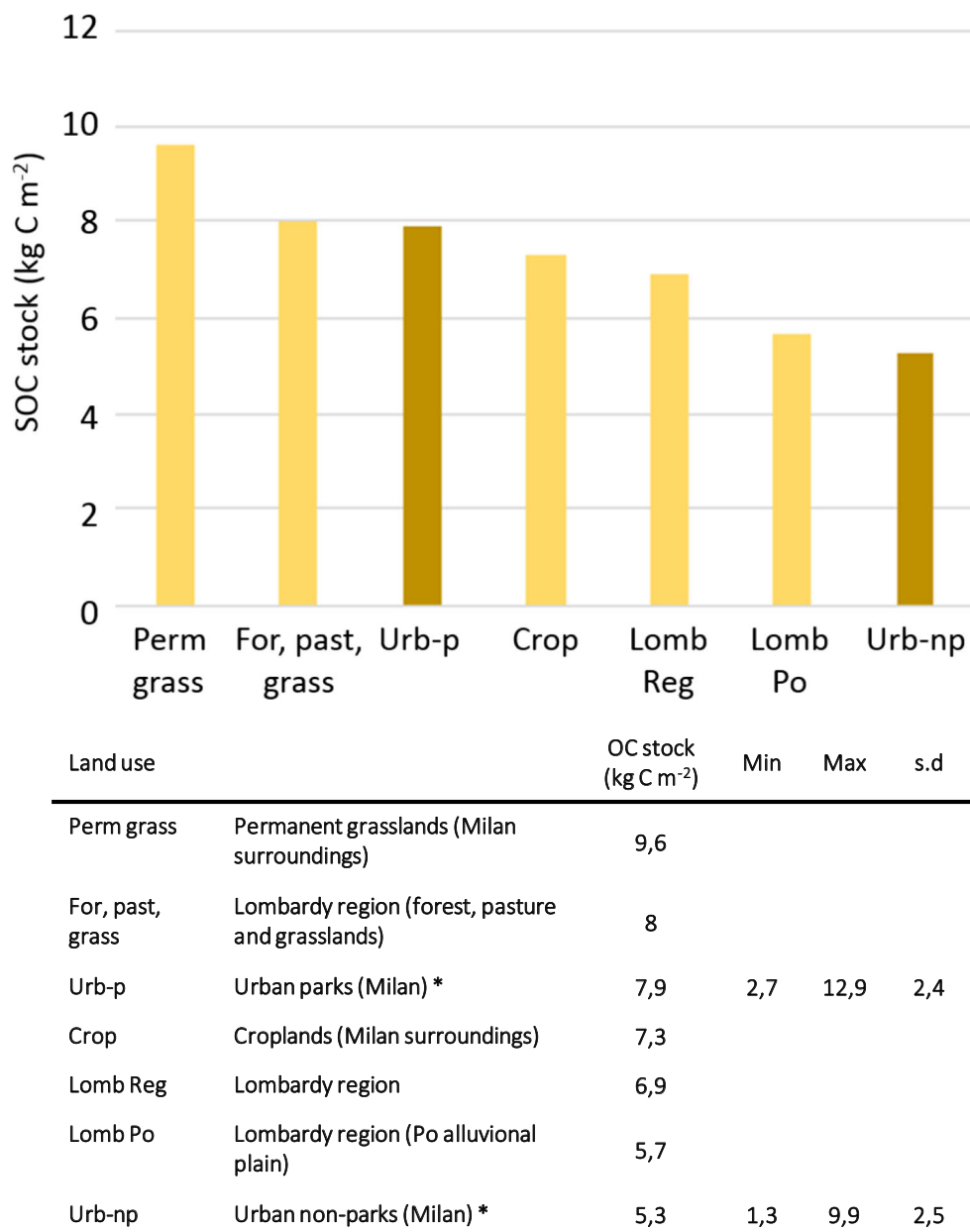
As we highlighted in the introduction, SOC stock estimations strongly depend on the reliability of the data on the amount of carbon stored for each land use or land cover type. However, we know that “urban soils” make up a heterogeneous group of human-altered soils, with properties that are rarely identical, sometimes not even in the same urban area. A first result of this study helped us to enhance the accuracy of our data on Milan urban SOC

stock estimates. Results of this study suggested that for Milan urban topsoil at 0–40 cm, 7.9 (s.d. 2.42) and 5.3 (2.53) kg C m⁻² can be used as values of typical SOC stock in urban parks and urban non-parks, respectively. Although apparently a simplistic classification, we believe that “urban parks” and “urban non-parks” represents a first distinction that we can address toward a better modelling of urban soil properties. Beyond this first distinction, it would be useful to uncover other differences in urban land use categories (see Rall et al. 2015 for an example of classification of different urban green spaces).

Table 5 SOC stock in urban areas

Area	Depth	OC stock (kg C m ⁻²)	Reference
Milan (Italy)	0–40	6.9	this study
Alabama, Georgia (USA) (beneath homes)	0–20	2.0	Majidzadeh et al. 2017
Alabama, Georgia (USA) (urban lawns)	0–20	5.3	Majidzadeh et al. 2018
Hamburg (Germany)	0–30	2.9	Dorendorf et al. 2015
Ruhr area (Germany)	0–30	11.1–16.4	Burghardt and Schneider 2018
Urumqi, Xinjiang (China)	0–80	8.1	Yan et al. 2015
Singapore	0–100	1.1–42.5	Ghosh et al. 2016
Ohio (USA)	0–100	16.3–21.1	Lal et al. 2012
Baltimore and New York (USA)	0–100	8.2	Pouyat et al. 2002
USA cities (Atlanta, Baltimore, Boston, Chicago, Oakland, Syracuse)	0–100	6.3	Pouyat et al. 2006
Central Chemozemic (Russia)	0–150	20–50	Sarzhhanov et al. 2017

Fig. 5 Comparison of soil organic carbon (SOC) stocks in different land uses, darkest column are soil sampled within this study, the others are data extracted from literature



Given the necessity to produce reliable estimates, we will discuss some of the constraints we have encountered when dealing with urban soils that limit our capacity (or the possibility) to model SOC stock for different land uses and land covers and what efforts should be made in order to enhance our knowledge about urban soils.

Fraction of black soil organic matter

Black soil organic matter (black SOM) refers to a wide array of thermally altered products derived from the incomplete combustion of fossil fuels or wood (Edmondson et al. 2015). It is generally considered inert or resistant to decomposition in terrestrial environments (Edmondson et al. 2015) and may

have a significant residence time in soils and sediments, e.g. 2000 years (Kuzyakov et al. 2009), 1300–2600 years (Lehmann et al. 2008) or 10,000 years (Swift 2001).

Black SOM in Milan urban soils decreased by increasing depth and this trend is reasonable to expect, as the primary source in cities is fossil fuel combustion, especially from traffic emissions (Liu et al. 2011), and thus we can hypothesize that it reached soil with atmospheric fallout, as happens with heavy metals. However, as black SOM has been found also in deeper layers, we can argue that disturbance (i.e. handling or mixing caused by human activities) may have move black SOM to subsurface layers. (In contrast, the fraction of black SOM on total SOM (Table 1) increased with increased soil depth, a trend observed in other studies on urban soils

(Edmondson et al. 2015). This may be due to the different residence time of black SOM compared to total SOM. With increased depth, total SOM is mineralized at a higher rate than black SOM, leading to high black SOM/SOM ratios.

Song et al. (2002) indicated that the fraction of black SOM in urban soils ranged from 18.3% to 41.0%. In Milan, we found comparable results, with black SOM representing 20% to 40% of the total organic matter. Concentrations of black SOM in urban soils of Milan can be considered high, suggesting the urban environment is an essential source and sink of black carbon (Liu et al. 2011). In fact, today global warming is likely to increase the decomposition of organic carbon in soil, and thus the release of carbon dioxide from soil, that may create positive feedback (Lehmann et al. 2008; Heimann and Reichstein 2008). Studies demonstrate that highly stable fractions of organic carbon in soil (such as black SOM) represent an important soil carbon pool (in terms of volume and residence time), and this is important to consider when dealing with the regulative role of soil in climate change (Czimeczik and Masiello 2007). Moreover, black carbon particulate in the atmosphere is dangerous for people's health and its incorporation in soil contributes to other processes such as disease regulation (MEA 2005).

Constraints in estimating OC stock in urban areas

Urban soils are a complex system. Several factors determine this complexity and different processes, sometimes not easy to detect and account for, may lead to an increase or depletion of organic matter in urban soils. For example, an increased input of water, nutrients and CO₂ increases the above-ground net primary productivity, soil respiration, and total below-ground C allocation, suggesting enhanced C cycling rates by urbanization (Kaye et al. 2005; Lorenz and Lal 2009). On the other hand, vegetation management practices are responsible for the export of material from vegetated soil (like removal of grass clippings, tree leaves, and other organic debris), and mechanical soil removal (usually restricted to topsoil, the layer richer in roots and humus) and the loss of highly active soil biota (Lorenz and Lal 2009; Craul 1999) are responsible for lowering SOC storage. However, many of the constraints we encounter in model urban soil properties are related to the uncertainty of environmental parameters recorded.

Concerning soil origin, urban and suburban areas often expand into former agricultural, industrial, extraction, or mining areas, which leads to residual soil effects from those former activities (Pavao-Zuckerman 2008). Previous studies have shown that historical land-use changes affect current SOC stocks in constructed urban parks (Takahashi et al. 2008), and past agriculture activity can double the soil C compared to urban soils with no legacy of agriculture (Lewis et al. 2006). However, this parameter is difficult to account for quantitatively. Besides the soil's history, the effect of the soil's age on SOC stock is known to be relevant (Scharenbroch et al. 2005), but accurate estimation

of the age of urban soils is highly uncertain. In our analysis, we excluded age as an environmental variable because its estimation was too ambiguous for many plots within the study area.

Mediation of vegetation management practices alters vegetation-soil interactions and thus input of organic matter in soil. Different vegetation management practices, such as the type of additives used (fertilizers, pesticides, herbicides), removal of litter and other organic debris, and the frequency of these events, are not easy to track. Moreover, vegetation management practices vary greatly within the same category of land cover or land use.

Soil thickness determines the quantity of SOC stocks and is directly related to human-made origins. Predicting soil depth in an urban environment is a hard task, because it is not dependent on soil-forming factors but most of the time requires timely field investigations.

Another limitation encountered is related to the land use and land cover classification system adopted. In fact, a standardized classification for urban areas does not exist. From this study we observed that land use explains variations in SOC better than land cover does. It is thus possible that a landscape effect is predominant over a local effect because of human-mediated activities. However, it is more plausible that both land use and land cover affect SOC in different ways, but more studies are necessary to infer the relative contribution of each factor.

The complexity and heterogeneity of urban soils together with uncertainty regarding their origin, their history, and past management may summarize the reasons why our capacity to model their properties is limited. We might even say that what is missing today in the case of urban soils (and may be missed in the future) is the soil-landscape relationship that is present in natural or seminatural soils in agriculture and forests, and that is exceptional support in the preparation of soil maps. In other words, the soil-landscape paradigm (Hudson 1992), on which soil mapping is explicitly or implicitly based, is much less relevant in the case of urban soils, for which it is not possible to rationalize the factors and processes of soil formation due to their high variability and unpredictability in an urban environment.

Some initiatives may be implemented to enhance our knowledge of urban soil properties. Among these, one of the most relevant is the intensification of soil sampling, stratified among different urban land cover and land use typologies and among different urban areas. This would allow urban soil property patterns to emerge. From this perspective, legislative incentives such as the institution of soil monitoring programs and protocols to quantify SOC stocks or update soil information systems could support more research.

Conclusions

Even though quantification of urban SOC stock and other soil properties remains difficult, a better understanding of

fundamental ecosystem functions is crucial to advance toward sustainability, because the contribution of urban ecosystems to national SOC inventories may have implications for land-use changes and planning policy, particularly in densely urbanized regions (Tao et al. 2015). Thus, accurate assessments of ecosystem carbon stocks are crucial to guide effective management of urban soils (Guo and Gifford 2002), and studies aiming to enhance the accuracy of carbon stock estimations are urgently required.

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