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Topsoil carbon stocks in urban greenspaces of the Hague, the Netherlands

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Abstract
Urbanization influences soil carbon (C) stocks and flows, which, in turn, affect soil-derived ecosystem services. This paper explores soil C storage in urban greenspaces in the Dutch city of The Hague along a transect from the suburban seaside towards the city centre, reflecting a toposequence from dune to peaty inland soils. C storage and C mineralisation potential were evaluated in relation to soil type and greenspace categories. Several soil-quality characteristics were measured, including dissolved organic C, pH, electrical conductivity, nitrogen, phosphorus, sulphur, calcium carbonate, and the water-holding capacity of the soil to evaluate what drives soil C storage in the urban context. The total SOC storage of the upper 30 cm of the greenspaces in The Hague (20.8 km^2 with 37% greenspace) was estimated at 78.4 kt, which was significantly higher than assumed given their soil types. Degradability of soil organic matter in laboratory batch tests varied between 0.2 and 3 mg C g_{SOC}^{-1} day^{-1}. Degradability was highest in the seaside dune soils; however, extrapolated to the topsoil using the bulk density, topsoil C mineralization was higher in the urban forest. Soils beneath shrubs appeared to be hotspots for C storage, accounting for only 13% of the aerial cover but reflecting 24% of the total C storage. Land ownership, land use, greenspaces size, litter management and soil type did not result in significantly different C stocks, suggesting that processes driving urban soil C storage are controlled by different factors, namely land cover and the urbanization extent.

Keywords Urban soils · Carbon storage · Soil organic carbon · Soil respiration · Urban greenspace · The Hague

Introduction

Population growth and urbanization are changing land use at an unpresented rate (Seto et al. 2012). However, the impact on global carbon (C) storage and soil organic matter (SOM) is not fully understood (Herrmann et al. 2020; Peng et al. 2017). For comparison purposes, the growth of urban ecosystem land use is outpacing that of agriculture, the latter of which has already produced a substantial historical decline of approximately 133 Pg soil organic C (SOC; Sanderman et al. 2017). SOM plays a key role in the delivery of soil-derived ecosystem services, such as primary production support, climate and biodiversity regulation, erosion protection and water quality control (Rawlins et al. 2015; Hoffland et al. 2020). In the case of urban soils this role is widely neglected (Morel et al. 2015; O'Riordan et al. 2021). However, with growing urbanization, the services that society receives from urban ecosystems are becoming increasingly important (Lorenz 2016; Tan et al. 2020).

Urban areas are hotspots of anthropogenic C emissions but have the potential to act as significant C sinks (Churkina et al. 2010; Edmondson et al. 2012; Lorenz and Lal 2015; Pouyat et al. 2002; Vasenev and Kuzyakov 2018). Soil C sequestration is a process supporting the ecosystem service of climate regulation (Minasny et al. 2017). Considering the long residence time of C in soils, enhancing soil C stocks (negative C emissions) is a suitable method to offset more of the anthropogenic C emissions (Lorenz and Lal 2015; Hansen et al. 2013). Accurate assessment of soil C stocks is therefore crucial to understand anthropogenic changes in urban soils in relation to the global C cycle and the potential of urban greenspaces to trap atmospheric C (Edmondson et al. 2012).

Changes in soil C stocks are the result of fresh C inputs to (e.g. litterfall, root exudations, by-products of microbial activity) and C output from (soil respiration) the soil (Lehman and

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Kleber 2015; Schmidt et al. 2011). The formation of SOM, or humification, is the process of organic fragments which are continuously transformed by microorganisms toward smaller molecular sizes (Lehman and Kleber 2015). Soil C, therefore, is always in a transient condition, and not a stagnant pool, which necessitates studying C fluxes in addition to C stocks for comprehensive soil C balancing (Janzen 2015; Lehmann and Kleber 2015; Schmidt et al. 2011). The breakdown of SOM increases the amount of polar and ionizable groups, which increases SOM's water solubility. Simultaneously, protection possibilities arise through greater reactivity toward mineral surfaces and incorporation into aggregates. Adsorption can then lead to desorption, exchange reactions with competing SOM, or biotic or abiotic degradation, potentially resulting in complete mineralization (Lehman and Kleber 2015). Biological activity may also lead to the direct depositions of microbial by-products and root exudates on mineral surfaces (Miltner et al. 2012). SOC storage thereby relies on balancing access of decomposing organisms to SOM and protection of SOC by organo-mineral associations (Lehmann and Kleber 2015). Variation in SOM degradability arises through variations in the presence or absence of microorganisms and enzymes and the nutrients they require, the properties and abundance of reactive mineral surfaces, the availability of water, soil acidity, and soil redox state (Lehman and Kleber 2015; Schmidt et al. 2011). In urban soils, the equilibrium between C mineralization on the one hand and transformation and protection, on the other hand, may be disturbed due to anthropogenic activities such as greenspace management (e.g., removal of biomass), atmospheric deposition of pollutants, and urban run-off, leading to accumulation or reduction of SOM stocks (Pavao-Zuckerman and Coleman 2005; Saviozzi et al. 2014; Vauramo and Setälä 2011).

A meta-analysis by Vasenev and Kuzyakov (2018) on C storage in the upper 1 m of global urban soil revealed that the C concentrations in urban soils can be 1.5 to 3 times higher and C accumulation can be deeper compared to non-urban soils, resulting in 3 to 5 times larger urban soil C stocks. Yet, a meta-analysis by Chien and Krumins (2022) focussing on the upper 30 cm of soil exposed that soil C stocks in natural habitats can be significantly larger than those in urban environments, which they explained by the fact that urban soil is often highly disturbed, compacted, contaminated, or otherwise influenced by human activity. However, they also acknowledged several studies that detected higher SOC levels in urban soils than in adjacent non-urban soils highlighting that urban SOC levels are very often context dependent (Chien and Krumins 2022).

Urban soil C has been studied in, amongst others, the cities of Milan (Italy, Canedoli et al. 2020), Leicester (United Kingdom, Edmondson et al. 2012), Berlin (Germany, Richter et al. 2020), and Moscow (Russia, Shchepeleva et al. 2017). These studies reported a high variability in soil C storage in urban greenspaces. Canedoli et al. (2020) detected that urban parks held greater SOC concentrations than non-parks, while SOC stocks did not significantly differ for wooded and grassland land covers, suggesting that a functional classification of urban greenspaces is more appropriate to distinguish soils with similar SOC storage potential. Edmondson et al. (2014) analysed land-cover effects and observed greater SOC concentrations in domestic gardens than in non-domestic greenspaces, which they attributed to the occurrence of woody trees and shrubs as well as management practices that include the addition of organic materials. Richter et al. (2020) found that when evaluating the distribution of soil C storage, the soil C densities increased towards the city's boundaries, which they also explained as a management effect of the large domestic gardens that are typical for the suburbs of Berlin. Lastly, Shchepelva et al. (2017) found intensive soil respiration likely for urban turf grasses, making C sequestration in soils beneath turf grasses uncertain, specifically in the early years after construction.

The aim of this study was to quantify the amount of C stored in the topsoil (0–30 cm) of urban greenspaces and to assess the spatial and categorical variability and drivers of different soil C pools in the city of The Hague, the Netherlands. For the Netherlands, data on urban soil C stocks and flows are lacking, despite the interest in the spatial assessment of urban ecosystem services (Derkzen et al. 2015; van Oorschot et al. 2021). Here, urban greenspaces are defined as natural and semi-natural green areas within the built environment, thereby include relatively undisturbed vegetated patches such as urban forests and heavily managed urban greenspaces such as roadside greenery and residential gardens. The assessment of urban C storage and C mineralisation potential enabled the evaluation of whether urban soil displays similar mechanistic links between C, vegetation, soil, and management as typically observed in non-urban soils. In addition to absolute C losses by measurements of C mineralization, this paper reports on the SOM degradability which is defined as the C mineralization rate normalized to the SOC content of the soil. SOM degradability thereby indicates the degree of stability, or protection, of the soil C stocks. By establishing a basis of soil C stocks in The Hague, the capacity of its soils to capture and store C can be acknowledged and incorporated in the policy framework for sustainable and resilient cities, safeguarding and enhancing the benefits that society receives from urban soils (Rawlins et al. 2015). Finally, this study represents the starting point to fill the lack of data on C storage in Dutch urban areas.

Study location

The Hague is part of the most densely populated area in northwestern Europe, home to the megalopolis formed by the country's main cities including Amsterdam and Rotterdam and is located on the North Sea coast, hence the temperate oceanic climate (Cfb, according to the Köppen-Gauger climate classification). The Hague provided the unique possibility to study urban soil C stocks along a toposequence from less developed sandy dune soils (suburban seaside) to more developed peaty inland soils (city centre). Urban soil was studied along a transect that crossed this toposequence (Supplementary Information: Fig. S1). The sample transect extended over the districts of the city centre and the adjacent suburban district which cover a combined area of ~ 20.8 km^2 (~ 25% of the entire municipality of The Hague), housing 10,253 and 7,978 persons per km^2 in 2021, respectively (CBS 2022). The Hague experienced a population growth of nearly 1% per year (CBS 2022). By assessment of a transect along a toposequence and the urban-suburban gradient, effects of soil type, urbanization extent (i.e. the population density in the city's districts), and type of greenspace on C storage could be evaluated.

Based on the green surface area maps of the Netherlands (green surface coverage by vegetation height on a 10 × 10 m grid, based on infrared satellite imagery; RIVM 2017), 7.6 km^2 (~ 37%) of these districts were covered by greenery, of which 52% was covered by herbaceous vegetation, 13% by shrubs and 35% by trees (Supplementary Information: Fig. S5). The remainder of the city (~ 63%) consists of sealed surfaces, of which the majority has a residential purpose (CBS 2015). The urban soils beneath the greenspaces of The Hague were further classified according to the framework discussed in Section "Classification of soils and greenspaces".

Methods

Selection of sampling plots and sampling method

Soil maps, geomorphological maps and green surface area maps of the Dutch National Institute for Public Health and the Environment (RIVM 2017) were used to pre-select 25 geo-referenced sample plots in urban greenspaces along the transect (Fig. 1). After assessment of the sample feasibility and accessibility, two sample plots were changed to plots nearby. After selection, the urban greenspaces were categorized (Section "Classification of soils and greenspaces"). Within each sample plot, three representative sampling sites were selected (A, B, C) and at each sample site, five subsamples were taken within a radius of 2 m, yielding a total of 25*3*5 = 375 samples. Subsamples were taken with a gouge auger (3 cm diameter) at a depth of 0–30 cm excluding the litter layer (hereinafter referred to as the topsoil). In cases where rocks, roots or other impenetrable material was hit before reaching the 30 cm depth, subsamples were retaken. Subsamples were mixed and homogenized. The bulk density of each sampling site was determined using the cylindrical core method at the middle point of the subsamples with five bulk density rings of 250 ml (Fig. 1).

The homogenized soil samples were analysed for soil C concentration, bulk density, clay content, pH, electrical conductivity (EC), nutrient level (N, P, and S), calcium carbonate ($CaCO_3$) levels, the water-holding capacity of the soil, and the mineralization of SOC (Table 1). These standard soil properties are indicative of soil quality and are also important for soil C storage. As $CaCO_3$ is likely not the only SIC species present in urban soils, SIC was calculated as the difference between total C and SOC.

To assess the potential SOM mineralization rates and infer the degradability of SOM, an aerobic incubation procedure was used. Approximately 50 g of field-fresh mixed soil sample prepared from all three sites per plot was moistened to 60% of the water-holding capacity in 1000 ml glass bottles sealed with butyl rubber stoppers and incubated under aerobic conditions at 20 °C in the dark. Soil respiration was monitored weekly for six weeks by measuring the evolution of CO_2 in the bottle headspace over time using a gas chromatograph (490 Micro-GC, Da Vinci Laboratory Solutions). All values are reported as the sum of CO_2-C measured in the gas phase and the share of CO_2-C dissolved in the aqueous phase (also referred to as dissolved inorganic carbon, DIC). The latter was calculated using the CO_2 concentration, the pH of the sample, the volume of water, the pressure in the bottle headspace and the temperature-corrected solubility of CO_2 in water as given by Henry's constant (given in Sander 2015).

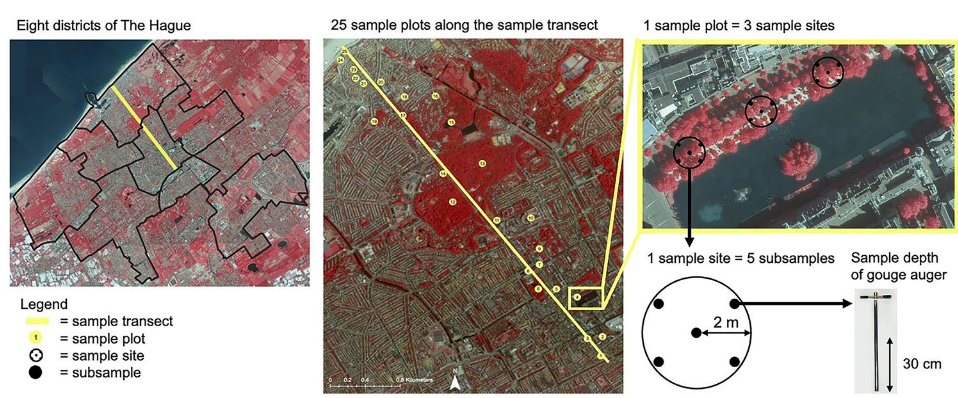

Fig. 1 Schematic presentation of the sampling method. The sample transect of ~ 5 km runs from North-West (North Sea) to South-East (city centre) (projected on infrared map (25 cm resolution) from Data. overheid (2019). Each sample plot consists of 3 sample sites. Each sample site consists of 5 subsamples that were mixed. Each subsample was taken with a gouge auger to a depth of 30 cm

Table 1 Parameters and procedures of standard solid analysis. NEN = Royal Netherlands Standardization Institute. ISO = International Organization for Standardization. BS = British Standard Institution

Parameter	Method
Bulk density	Cylindrical core method, five 250 ml rings, in the middle of the 0–30 cm soil profile, BS 1337–2 (1990)
Water content	Gravimetrically after drying at 105 °C (ISO 11465 (1993))
Texture	By hand, according to KA5 by Ad-hoc AG Boden (2005) and translated to the FAO/USDA system
Water-holding capacity (WHC)	20 g of field fresh soil in filtered funnels with 100 ml of H_2O for at least 24 h, gravimetrically after drying at 105 °C
pH value	In water extractions (40 ml: 10 g) (ISO 10390 (2005))
Electrical conductivity (EC)	In water extractions (40 ml: 10 g) ISO 11265 (1994)
Soil organic matter (SOM)	Loss-on-ignition (LOI) at 550 °C for 3 h (ISO 10694 1995)
SOC mineralization	Measurement of CO_2 produced upon aerobic incubation in the laboratory at 60% WHC and 20 °C (details below this table)
Dissolved organic carbon (DOC)	In water extractions (40 ml: 10 g) and ultraviolet–visible absorption spectrum at 254 nm
Total C (TC)	NEN-ISO 10694 (2008)
Soil organic carbon (SOC)	NEN-ISO 10694 (2008)
Nitrogen (N), Phosphorus (P)	NEN 6966 (2005)
Sulphur (S)	NEN 6966 (2005)
Calcium carbonate ($CaCO_3$)	NEN-EN-ISO 10693 (2014)
Soil inorganic carbon (SIC)	TC—SOC

C mineralization data were used to compute three parameters: (1) C mineralization rates, (2) Topsoil C mineralization rates, and (3) the degradability of SOM.

(1) C mineralization rates (mg C g_{DW}^{-1}) were calculated from the slope of the CO_2 concentration over time (Eq. 1):

$$SOC_{min} = \frac{dC}{dt} \times V \times \frac{P \times M}{P_S \times V_m \times m} + DIC \quad (1)$$

where dC/dt denotes the slope of the CO_2 concentration (fraction) over time, V the headspace volume (l),, P the pressure of the sample (hPa), M the molar mass of C (12.0 g mol^{-1}), P_S the standard pressure (1013.25 hPa), V_m the molar gas volume at 20 °C (24.1 l mol^{-1}), m the dry soil weight (mg), and DIC the dissolved inorganic C dissolved in the pore water (mg C g_{DW}^{-1}).

(2) Topsoil C mineralization rates (SOC_{min_TOP}, mg C m^{-2} day^{-1}) were calculated from the C mineralization rates (SOC_{min}) and bulk density BD (g cm^{-3}), normalised to a topsoil thickness of 0.3 m (Eq. 2):

$$SOC_{min\_TOP} = \frac{SOC_{min}}{BD} \times 0.3 \quad (2)$$

(3) Degradability of SOM (SOC_{deg}, mg C g_{SOC}^{-1} day^{-1}) was calculated by normalizing C mineralization rates (SOC_{min}) to SOC (Eq. 3):

$$SOC_{deg} = \frac{SOC_{min}}{SOC} \quad (3)$$

All three parameters reflect the potential C release as measured under laboratory conditions.

C densities and C stocks

SOC concentrations were transformed to SOC densities of the topsoil (kg C m^{-2}) according to Eq. 4, where SOC is the SOC concentration (%), BD the bulk density (g cm^{-3}), and D the sampling depth (m) (Weissert et al. 2016).

$$SOC\ density = SOC \times BD \times D \quad (4)$$

Using the C densities, the green surface area maps of The Hague (Supplementary Information, Fig. S5) were transformed into a soil C density map based on the method of Richter et al. (2020). The total C stock was computed by multiplying the surface areas of each vegetation type with the corresponding soil C density, according to Eq. 5 where v refers to the land cover being either trees, shrubs or herbaceous vegetation.

$$Total\ C\ stock = \sum mean\ TC\ density_v * surface\ area_v \quad (5)$$

Classification of soils and greenspaces

National soil maps often exclude soil types of urban areas (Rawlins et al. 2008), which is also the case for The Hague (Supplementary Information: Fig. S1). Nonetheless, a pattern can be derived with sandier dune soils along the coastline towards peaty and/or clayey soils more inland. It was hypothesised that this spatial pattern is continuous and therefore could be extrapolated over the urban area. Following the definitions by De Bakker et al. (1989), four soil types were derived using the Dutch soil map (Table 2; Supplementary

Table 2 Greenspace categorization of each sample plot

Sample plot	Soil type	Land cover	Land ownership	Urbanization	Green-space size	Land use	Litter management class	Management class
1	Meerveen	Grass	Private	City centre	Medium	Park	Litter removed	Private
2	Meerveen	Trees	Public	City centre	Small	Street tree	Litter removed	Trees on sealed surface
3	Meerveen	Grass	Public	City centre	Medium	Park	Litter removed	Fertilized grass
4	Beekeerd	Trees	Public	City centre	Medium	Park	Litter removed	Trees on unsealed surface
5	Beekeerd	Trees	Public	City centre	Small	Street tree	Litter removed	Trees on sealed surface
6	Beekeerd	Shrubbery	Public	City centre	Large	Park	Litter undisturbed	Shrubs
7	Vlakvaag	Shrubbery	Public	City centre	Small	Non-park	Litter undisturbed	Shrubs
8	Vlakvaag	Trees	Public	City centre	Small	Street tree	Litter removed	Trees on sealed surface
9	Vlakvaag	Trees	Public	City centre	Small	Street tree	Litter removed	Trees on sealed surface
10	Vlakvaag	Grass	Public	City centre	Medium	Park	Litter removed	Unfertilized grass
11	Vlakvaag	Trees	Private	City centre	Large	Park	Litter undisturbed	Private
12	Vlakvaag	Trees	Private	Suburban	Large	Urban forest	Litter undisturbed	Private
13	Vlakvaag	Trees	Public	Suburban	Large	Urban forest	Litter undisturbed	Natural forest
14	Vlakvaag	Trees	Public	Suburban	Large	Urban forest	Litter undisturbed	Natural forest
15	Vlakvaag	Trees	Public	Suburban	Large	Urban forest	Litter undisturbed	Natural forest
16	Duinvaag	Grass	Public	Suburban	Small	Street tree	Litter removed	Fertilized grass
17	Duinvaag	Trees	Public	Suburban	Medium	Street tree	Litter removed	Trees on unsealed surface
18	Duinvaag	Shrubbery	Private	Suburban	Small	Non-park	Litter removed	Private
19	Duinvaag	Grass	Public	Suburban	Large	Park	Litter undisturbed	Unfertilized grass
20	Duinvaag	Shrubbery	Public	Suburban	Medium	Non-park	Litter undisturbed	Shrubs
21	Duinvaag	Shrubbery	Private	Suburban	Medium	Non-park	Litter undisturbed	Private
22	Duinvaag	Trees	Public	Suburban	Medium	Park	Litter undisturbed	Trees on unsealed surface
23	Duinvaag	Grass	Public	Suburban	Medium	Park	Litter removed	Fertilized grass
24	Duinvaag	Grass	Public	Suburban	Medium	Park	Litter undisturbed	Dune thickets
25	Duinvaag	Grass	Public	Suburban	Medium	Non-park	Litter undisturbed	Dune thickets

Information: Fig. S1): (1) 'Duinvaag' soils: poorly developed sandy soils of which the sand particles are coated with iron (World Reference Base for Soil Resources (WRB 2015): Arenosol), (2) 'Vlakvaag' soils: lightly coloured, humus-poor, poorly developed sandy soils (WRB: Arenosol), (3) Beekeerd' soils: nutrient-rich humus layer on top of a nutrient-poor sandy layer. The soil is dominated by oxidation processes (WRB: Cambisol), and (4) 'Meerveen' soils: mineral topsoil on top of a eutrophic peat layer (WRB: Histosol).

The soils were further classified using eight criteria according to the urban soil framework of Ajmone Marsan et al. (2016): (1) physical and chemical properties, (2) pollution, (3) landscape metrics, (4) ownership, (5) aesthetical value, (6) specific ecological function, (7) social and (8) historical value. These criteria were assessed based on a combination of field-based measurements and observations, and municipal data.

Greenspaces were categorized according to their land cover, land ownership, urbanization extent, land use, greenspace size, litter management and management class as follows (Table 2):

The land cover was divided into three vegetation classes: trees (> 2.5 m), shrubs (> 1 m), and herbaceous vegetation (< 1 m). This categorization was consistent with the green surface maps of The Netherlands (RIVM 2017). The category land ownership was based on who managed the greenspace. Publicly owned greenspaces included greenspaces that were under municipal management and privately owned greenspaces were not. The urbanization extent was based upon whether the sample plot was located in the district of the severely urbanized city centre or in the more suburban district of Scheveningen. The category land use entailed urban forests, street trees, parks and non-parks. The category 'park' included playgrounds, cemeteries and plots of herbaceous vegetation used for recreational purposes. The category 'non-park' included plots of shrubbery on pavements that could not be considered a park because of their small size and/or inaccessibility for recreation inside the greenspace. The size of the greenspaces was assessed using the most recently available satellite imagery (April 2020). The litter management entailed plots that were naturally augmented with plant litter and plots where litter was regularly removed. For the publicly owned greenspaces, all types of greenspace management were provided by the municipality (van Droesbergen 2017) and for the privately owned greenspaces, litter management was provided by the respective private owners. All greenspaces received management practices that involved the removal of plant litter from the system in varying intensities. The management packages that allowed for the development of thick litter layers with plant litter in varying stages of degradation were placed in the category of 'litter undisturbed'. The remaining greenspaces were assigned to the 'litter removed' category.

Statistical analyses

To determine whether the soil C storage differed for the soil and greenspace categories of Section "Classification of soils and greenspaces", the soil parameters SOC density, LOI, DOC, pH and WHC were analysed using the non-parametric Kruskal–Wallis and Wilcoxon rank sum test as the assumptions of the ANOVA and Student's *t*-test were not met. Mean values are reported with their corresponding standard error (mean value ± standard error). To find out what drives the variation in urban soil C stocks, correlations between all measured soil parameters were tested with Spearman's rank correlation coefficients, as the data were non-normally distributed. Relevant coefficients of the linear regression analysis of the LOI and SOC concentration of the soil are reported with their confidence interval (CI).

Results

Applying the urban soil framework of Ajmone Marsan et al. (2016) the Hague urban soils were of a sandy texture with a wide range of pH values ranging from extremely acidic to moderately alkaline (facet: physical and chemical properties). For the sample plots where data was available, slight to moderate levels of contamination by heavy metals were reported (facet: pollution). Greenspaces were of varying shapes and sizes ranging from several square meters to over 100 hectares. Greenspaces were either surrounded by buildings or main roads, where most public greenspaces were adjacent to roads and most private greenspaces were surrounded by buildings or closed fences (facets: landscape metrics and ownership). All greenspaces were valuable, either from the ecological, social, historical, or aesthetical perspective (facets: specific ecological function and values, see Supplementary Information: Table S1 for the classification of each sample plot).

Urban soil properties and their interrelationships

Soil properties

The mean SOC concentration was 2.92 ± 0.36% with large variations along the transect (Fig. 2A). High SOC levels were detected in the mid-section of the city centre (sample plots 5, 6, and 7), but the suburban area also contained greenspaces with high SOC levels. Relatively low SOC levels were measured in the urban forest (sample plots 10–15). The lowest SOC levels were detected in the dunes (sample plots 24 and 25). Linear correlation analysis revealed a strong relationship between loss-on-ignition and SOC ($R^2 = 0.77$, Supplementary Information: Fig. S2). The slope of the regression function equalled 0.49, with the 95% CI ranging from 0.37 to 0.60, meaning that approximately 49% of the SOM consisted of SOC.

The mean dry bulk density of the soil was 1.24 ± 0.04 g cm^{-3}. Bulk density was especially low in the urban forest (sample plots 11–15). In the dunes, the highest bulk densities were measured (sample plots 24 and 25). The bulk density also varied locally, i.e. within a sample plot, revealing the heterogeneity of urban soils even within the same greenspace (Fig. 2B).

The mean clay content of all sample plots was approximated at 8.3 ± 1.68%. The clay content was especially high in greenspaces that consisted of allochthonous topsoil (sample plots 9 and 23: silty clay loam). The remaining samples contained low levels of clay and ranged from sand to loamy sand, to sandy loam (FAO/USDA system, Fig. 2D).

DOC concentrations varied significantly along the transect and intra-plot variability was also observed (Fig. 2E). Low DOC levels were observed in the dunes (sample plots 24 and 25), and higher DOC values were detected in the urban forest (sample plots 11–15), but also in the mid-section of the city centre (sample plots 6 and 7).

The water-holding capacity of the soil ranged between 19 and 37% related to the dry weight of the soil. The water-holding capacity was consistently high in the forested area in the middle of the transect and significantly lower in the dune area. Lower and higher water-holding capacities were also detected in several other greenspaces (Fig. 2F).

The pH value differed significantly along the transect ranging from 3.83 to 8.58 with a mean pH of 6.39 ± 0.10 (Fig. 2G). The city centre soils had pH values ranging from moderately acidic to neutral, but once the forested area in the middle of the transect was reached, the pH strongly dropped to very to extremely acidic. In the suburban area, the pH

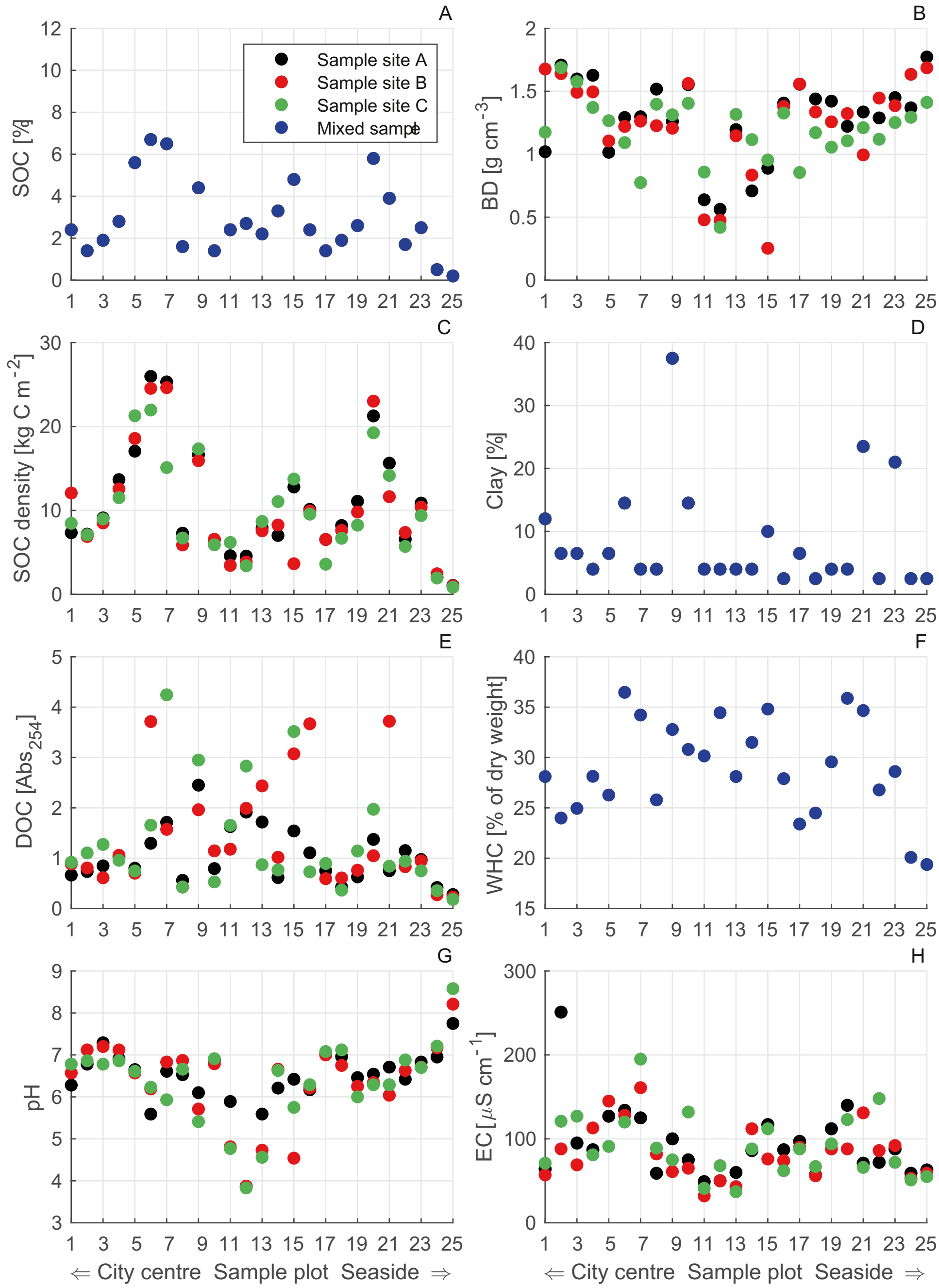

Fig. 2 SOC concentration (**A**), bulk density (**B**), SOC density (**C**), clay content (**D**), DOC concentration (**E**), water-holding capacity (**F**), pH (**G**) and electrical conductivity (**H**) in the upper 30 cm of soil along the transect. Each dot represents a measurement of the mixed sample (blue) or a measurement of each sample site (green, red, and black)

increased again and finally became slightly to moderately alkaline in the dune area.

The mean value of electrical conductivity (EC), representative of the number of charged solutes in the pore water, was 89 ± 4.30 µS cm^{-1}. The spatial trend along the transect roughly followed the spatial pattern of SOC (Fig. 2H).

Total nutrient levels (N, P, S) were determined to be $0.20 \pm 0.03\%$ N, $0.06 \pm 0.01\%$ P, and $0.06 \pm 0.01\%$ S on average. Nutrient percentages translated to mean C:N, C:P and C:S ratios of 14.6, 54.8 and 55.4, respectively (Supplementary Information: Fig. S3). In general, the C:N ratio of the soil was similar or higher than the C:N ratio of SOM (C:N ratio of SOM: 12). A large variation in C:P ratios was detected along the transect (C:P ratio of SOM: 50). The C:P ratio was consistently high in the urban forest (sample plots 11–15), however, some greenspaces in the city centre also contained high C:P ratios (sample plots 5 and 7). Finally, the C:S ratio was lower than that of SOM in almost all sample plots (C:S ratio of SOM: 70).

Interrelationships between soil properties

Several relevant interrelationships were detected between the measured soil properties (Table 3). Strong correlations between SOC and N and S were detected. The association of SOC with P was only moderate but strong with the C:P ratio. N levels strongly correlated with S levels, but not with P. A strong association between N and the water-holding capacity of the soil was observed, which was not present for the other nutrients. Additionally, a strong association between SOC and the water-holding capacity of the soil was detected.

Loss-on-ignition values strongly correlated with DOC concentrations. The remaining strong correlations of LOI and the other soil properties were similar to that of SOC. DOC values strongly correlated with the water-holding capacity of the soil. The pH strongly correlated with the $CaCO_3$ levels of the soil. No significantly strong correlations between EC and other soil properties were detected.

The topsoil mineralization rates positively correlated with the SOC concentration, the pH, the C:P ratio and the water-holding capacity of the soil. The correlation of the topsoil mineralization rates with C:N and C:S ratios was only weak. Additionally, a moderate association between topsoil mineralization rates and DOC levels was observed. Naturally, the correlations between the C mineralization normalized to SOC were inversely proportional to the C and nutrient concentration of the soil (Table 2).

C densities, C storage and degradability of urban SOM

The mean total C density was 110 ± 0.70 t ha^{-1} of which $89 \pm 0.36\%$ was contributed by SOC and $11 \pm 0.05\%$ by SIC, all related to the upper 30 cm of soil. SIC contribution to total C was especially large in the dune areas, whilst SOC dominated the total C values for the remainder of the samples. A large variation in SOC densities was observed in the 25 sample plots (Fig. 2C). The highest SOC densities were found in sample plots 5, 6, 7, and 20. Plots 7 and 20 consisted of patches of shrubs on the pavement, plot 5 of an old chestnut tree and plot 6 of shrubbery in the palace gardens. The lowest C densities were found in plots 24 and 25: the dunes. No clear spatial trend was observed along the transect.

Densities of soil organic C, N, P and S in different types of greenspaces

Categorizing the sample plots by land cover revealed that soils beneath shrubbery contained significantly higher SOC densities (17.7 ± 6.85 kg C m^{-2}) than those beneath trees (9.12 ± 4.86 kg C m^{-2}) or herbaceous vegetation (7.18 ± 3.63 kg C m^{-2}; Kruskal–Wallis test, $p < 0.01$; Fig. 3A). DOC values were significantly higher in soils beneath shrubbery (1.69 ± 1.24 Abs_{254}) and trees (1.35 ± 0.82 Abs_{254}) than herbaceous vegetation (0.84 ± 0.68 Abs_{254}; Kruskal–Wallis test, $p < 0.01$). Moreover, N, P and S densities were higher in soils beneath shrubs (1.26 ± 0.50 kg N m^{-2}; 0.29 ± 0.14 kg P m^{-2}; 0.34 ± 0.12 kg S m^{-2}) than beneath trees (0.58 ± 0.30 kg N m^{-2}; 0.17 ± 0.09 kg P m^{-2}; 0.16 ± 0.09 kg S m^{-2}) and herbaceous vegetation (0.60 ± 0.29 kg N m^{-2}; 0.23 ± 0.11 kg P m^{-2}; 0.17 ± 0.06 kg S m^{-2}; Kruskal–Wallis Test, $p < 0.01$). However, P densities in soils beneath shrubs were only significantly higher than soils beneath trees, and similar to P densities of soils beneath grasses.

For the soil types, soils classified as 'Beekeerd' soils held significantly higher SOC densities (18.6 ± 5.26 kg C m^{-2}) than those classified as 'Meerveen' (8.40 ± 1.62 kg C m^{-2}), 'Vlakvaag' (9.65 ± 6.09 kg C m^{-2}) and 'Duinvaag' (8.75 ± 5.71 kg C m^{-2}; Kruskal–Wallis test, $p < 0.01$). SOC densities in 'Meerveen', 'Vlakvaag' and 'Duinvaag' soils did not significantly differ from each other (Fig. 3B). Additionally, 'Beekeerd' soils exposed higher EC values (114 ± 22.7 μS cm^{-1}) than 'Meerveen' (105 ± 60.1 μS cm^{-1}), 'Vlakvaag' (83.0 ± 39.1 μS cm^{-1}) and 'Duinvaag' soils (83.0 ± 26.2 μS cm^{-1}).

Soils in greenspaces that were publicly owned showed similar SOC densities to the greenspaces that were privately owned (Wilcoxon rank sum test > 0.01; Fig. 3C). Although SIC densities were significantly higher in publicly (1.01 ± 0.94 kg C m^{-2}) than in privately owned greenspaces (0.20 ± 0.19 kg C m^{-2}; Wilcoxon rank sum test, $p < 0.05$). Greenspaces located in the city centre contained higher SOC densities (12.5 ± 6.84 kg C m^{-2}) than those located in the suburban area (8.45 ± 5.15 kg C m^{-2}; Wilcoxon rank sum test, $p < 0.05$; Fig. 3D). Moreover, the more urbanized city centre contained higher EC values (100 ± 45.5 μS cm^{-1}) than the suburban district (80.7 ± 26.7 μS cm^{-1}; Wilcoxon rank sum test, $p < 0.01$). Categorizing greenspaces by land ownership, land use, greenspace size or litter management did not result in significantly different SOC densities (Kruskal–Wallis test or Wilcoxon rank sum test, $p > 0.05$; Fig. 3).

Although no significant difference in C densities was detected between greenspaces with and without litter management, litter management did have a pronounced effect on other soil properties. Higher DOC (1.48 ± 1.05 Abs_{254} vs. 1.01 ± 0.70 Abs_{254}) and loss-on-ignition levels ($6.76 \pm 4.65\%$ vs. $4.66 \pm 1.97\%$) were detected in greenspaces where plant litter was relatively undisturbed (Wilcoxon rank sum test, $p < 0.05$). Additionally, pH values were lower in greenspaces where plant litter was left relatively undisturbed (6.1 ± 1.1) compared to greenspaces where plant litter was regularly removed (6.7 ± 0.4, Wilcoxon rank sum test, $p < 0.01$). Lastly, soils with undisturbed plant litter contained a higher water-holding capacity ($30 \pm 0.06\%$) than those where litter was removed ($27 \pm 0.03\%$; Wilcoxon rank sum test, $p < 0.05$; Supplementary Information, Fig. S4).

Table 3 Spearman correlation *r* of the soil C fractions, soil-quality characteristics and mineralization rates

	TC	TC_d	SOC	SIC	LOI	DOC	pH	EC	N	P	S	C:N	C:P	C:S	Ca-CO_3	WHC	BD	Clay	C_{deg}	C_{min_TOP}
TC	-		***0.99***	-0.30	***0.83***	*0.63*	*-0.54*	*-0.47*	***0.83***	*0.56*	***0.82***	0.34	*0.67*	0.42	-0.48	***0.78***	*-0.52*	0.38	***-0.80***	***0.74***
TC_d		-	***0.79***	-0.18	***0.65***	*0.37*	-0.17	*0.62*	***0.72***	*0.68*	***0.77***	0.22	*0.34*	0.26	-0.22	*0.48*	0.01	*0.42*	***-0.71***	*0.42*
SOC			-	-0.41	***0.81***	*0.61*	*-0.58*	*0.42*	***0.83***	*0.52*	***0.81***	0.37	***0.72***	0.44	*-0.54*	***0.81***	*-0.55*	0.36	***-0.81***	**0.76**
SIC				-	-0.21	-0.26	*0.58*	0.13	-0.38	-0.10	-0.20	-0.14	-0.41	-0.24	***0.80***	-0.46	*0.33*	-0.30	0.32	-0.32
LOI					-	***0.74***	*-0.57*	*0.46*	***0.81***	*0.67*	***0.74***	0.07	*0.46*	0.21	*-0.44*	***0.78***	*-0.45*	*0.35*	***-0.73***	*0.62*
DOC						-	0.11	0.23	*0.61*	*0.40*	*0.40*	0.10	*0.51*	*0.35*	*-0.61*	***0.70***	*-0.48*	*0.31*	*-0.50*	*0.64*
pH							-	0.10	*-0.57*	-0.17	-0.29	-0.17	*-0.64*	*-0.45*	***0.84***	*-0.68*	*-0.65*	-0.18	*0.49*	**-0.72**
EC								-	*0.44*	*0.48*	*0.49*	0.06	0.10	0.12	0.05	0.26	0.00	*0.32*	*-0.42*	-0.19
N									-	*0.60*	***0.82***	-0.12	*0.55*	0.17	*-0.56*	***0.77***	*-0.41*	0.44	***-0.72***	*0.58*
P										-	*0.63*	-0.07	-0.11	-0.00	-0.21	0.47	0.06	0.40	*-0.53*	-0.15
S											-	0.03	0.45	-0.09	-0.24	*0.63*	*-0.30*	0.42	*-0.64*	*-0.49*
C:N												-	0.45	***0.74***	-0.10	0.11	*0.31*	-0.04	-0.31	0.38
C:P													-	*0.52*	*-0.53*	*0.64*	***-0.71***	0.19	*-0.50*	**0.83**
C:S														-	-0.43	0.28	*-0.38*	0.04	*-0.40*	*0.49*
Ca-CO_3															-	*-0.64*	*-0.50*	-0.07	*0.46*	*-0.53*
WHC																-	*-0.56*	0.43	***-0.77***	**0.72**
BD																	-	-0.07	*0.60*	**0.72**
Clay																		-	-0.16	0.38
SOC_{deg}																			-	*-0.38*
SOC_{min_TOP}																				-

Italic numbers indicate significant correlations ($p < 0.01$), and bold numbers indicate strong correlations ($rs \geq 0.70$). *TC* total soil carbon, *SOC* soil organic carbon, *SIC* soil inorganic carbon, *LOI* loss-on-ignition pH, *EC* electrical conductivity, *N* nitrogen, *P* phosphorus, *S* sulphur, C:N ratio, C:P ratio, C:S ratio, *WHC* water-holding capacity, *BD* bulk density, Clay clay fraction, SOC_{deg} [mg C g_{SOC}^{-1} day^{-1}], SOC_{min_TOP} [mg C m^{-2} day^{-1}]

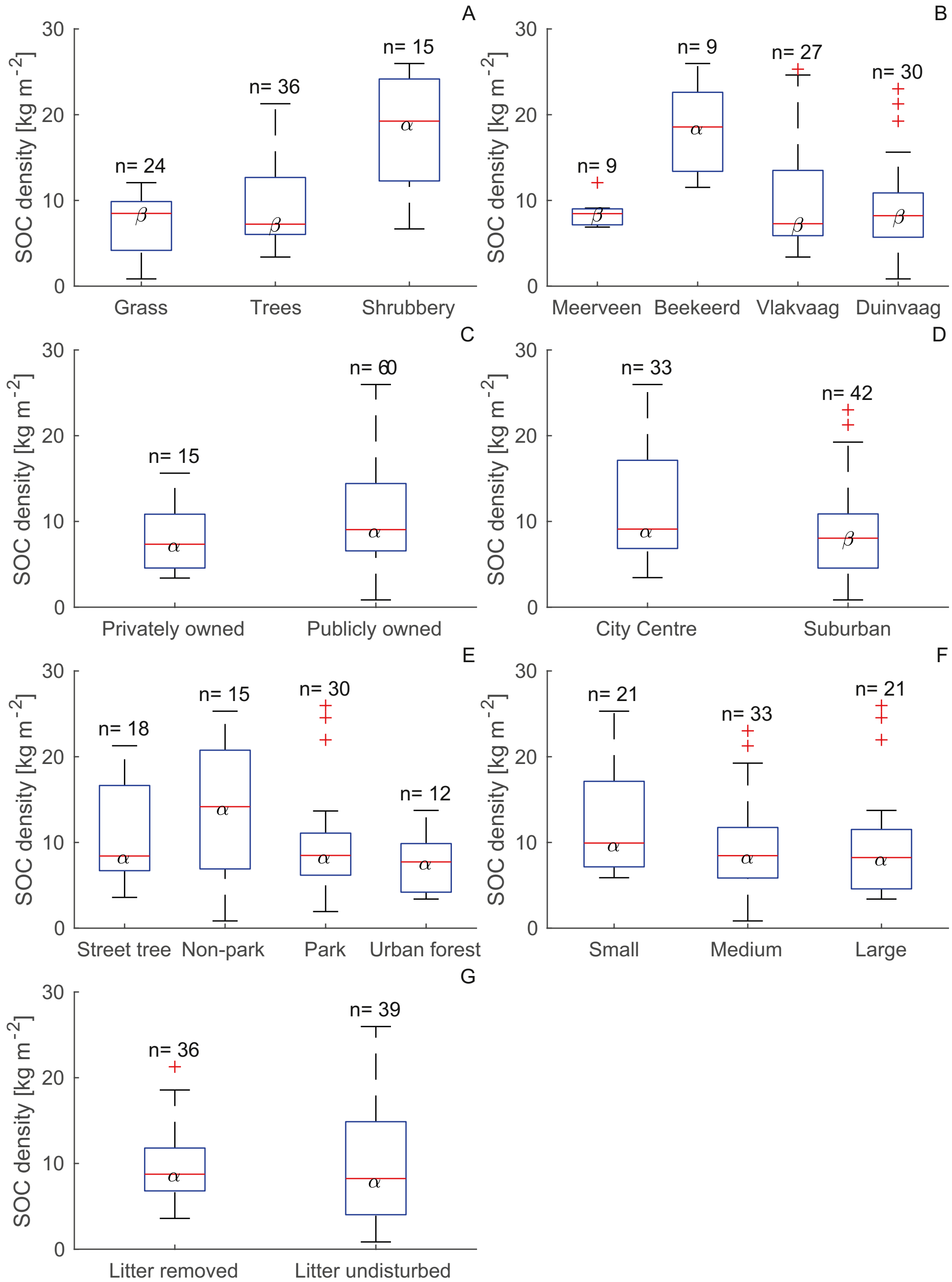

Fig. 3 Variability of SOC densities in the upper 30 cm of soil of different soil and greenspace types. Line = median, box = first and third quartiles, whiskers = highest and lowest data points excluding outliers (plus signs). α, β = indication of statistical similarity or difference

Soil carbon storage

Based on the land cover, it was estimated that the upper 30 cm of soil in The Hague that is covered by greenspaces has the potential to store 78.4 kt of C (± 2.64 for grasses, ± 1.87 for shrubs, ± 2.12 kt C for trees). The soil beneath shrubs proved to be hotspots for C accumulation as shrubs were responsible for more C storage (~ 24%) than what corresponded to their surface area (~ 13%). This relation was reversed for soils beneath herbaceous vegetation (53% of aerial cover, ~ 42% of total C storage) and the tree cover and C stored beneath trees was consistent (~ 35% of aerial coverage, ~ 35% of total C storage).

Degradability of urban soil organic matter

The mean cumulative topsoil mineralization was 115 ± 9.58 mg C m^{-2} and the mean cumulate mineralization normalized to SOC, i.e. the degradability of SOM, was 20.6 ± 2.75 mg C g_{SOC}^{-1} within a 6-week incubation period, both for a temperature of 20 °C. Mineralization over time was linear for all soil samples (R^2 = 0.98–1.00) and could therefore be translated to potential mineralization rates (Fig. 4). The mean topsoil mineralization rate was 2.75 ± 0.23 mg C m^{-2} day^{-1} and the mean SOM degradability was 0.49 ± 0.07 mg C g_{SOC}^{-1} day^{-1}.

The SOM degradability was 4.5–4.9 times higher in the dune soils than in the city centre, urban forest and suburban area (Fig. 4; Kruskal–Wallis test, $p < 0.01$). However, the topsoil mineralization rates were 4.6 times higher in the urban forest than in the dunes. No significant difference was detected between the topsoil mineralization rates of greenspaces located in the city centre and suburban area (Fig. 4; Kruskal–Wallis test, $p < 0.01$).

SOM degradability was independent of soil and greenspace categories (Kruskal–Wallis test or Wilcoxon rank sum test, $p > 0.05$). The topsoil mineralization rates were higher in soils beneath shrubs and threes than in soils beneath herbaceous vegetation (Table 4). Also, the topsoil mineralization rates were higher in privately managed greenspaces than in publicly managed greenspaces. Lastly, the topsoil mineralization rates were higher in the greenspaces where plant litter was undisturbed than in greenspaces where plant litter was regularly removed (Table 4).

Discussion

This study aimed to assess the spatial and categorical variance of different soil C pools in the city of the Hague to better understand the C storage potential of The Hague urban soils and to initially fill the data gap on C storage in Dutch urban areas. In addition, the total soil C storage of the greenspaces of The Hague was estimated. Using the mean C densities of the land cover classes led to a total soil C storage of 78.4 kt in the upper 30 cm of soil in the greenspaces of the studied districts of which 32.8 kt was stored beneath herbaceous vegetation, 27.0 kt was stored beneath trees, and 18.6 kt was stored beneath shrubs. The use of high spatial resolution GIS data at the scale of 10×10 m enabled the inclusion of small patches of green in the total soil C storage of The Hague. This inclusion proved to be significant as large greenspaces only comprised 26% of the greenspaces in the studied districts and the measured C densities in the medium and smaller greenspaces were comparable to those in larger greenspaces. Based on the patterns commonly observed in non-urban soils, it was hypothesized that soil type and land use would be appropriate predictors for urban soil C storage. However, the hypothesized links with land use and soil type were not as apparent in this case study, suggesting that processes driving soil C storage are controlled by different factors.

The total soil C storage of 78.4 kt in greenspaces of the studied districts is comparable to the 106 kt stored up to the same depth in the greenspaces of the city of Seoul, South Korea (Yoon et al. 2016), a city 29 times the size of the districts of the Hague. On average, a soil C density of 11.0 kg C m^{-2}, of which 10.2 kg C m^{-2} was SOC, was measured in the topsoil of the 25 sampled greenspaces in The Hague. For the urban C storage studies performed in the same Köppen-Gauger climate classification, namely Cfb, Beesley (2012) detected a total C density between 1.0 and 5.0 kg C m^{-2} in the upper 15 cm of soil in Liverpool, U.K., Edmondson et al. (2012) reported a SOC density of 14.4 kg C m^{-2} in the upper 21 cm of soil in Leicester, U.K., Cambou et al. (2018) estimated a SOC density of 9.9 kg C m^{-2} in the upper 30 cm of soil in Paris, France, and Weissert et al. (2016) reported a SOC density between 9.3 and 16.4 kg C m^{-2} in the upper 30 cm of soil in Auckland, New Zealand. Data were in the same order of magnitude, however, sample depths between

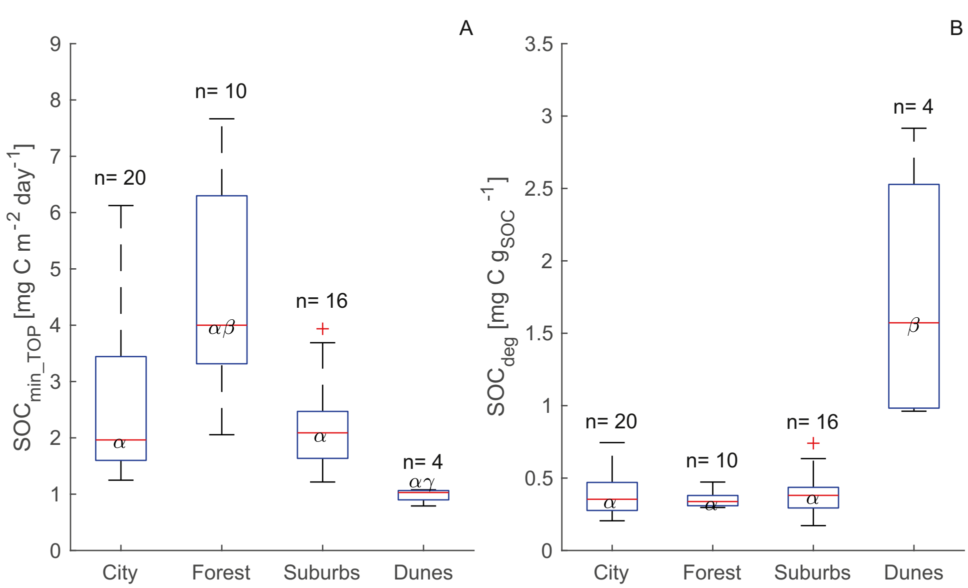

Fig. 4 Potential topsoil mineralization rates of the upper 30 cm of soil (**A**) and potential SOM degradability (**B**) under laboratory conditions (60% water holding capacity, 20 °C) in 25 urban soil samples along the transect running from the city centre to the urban forest to the suburbs to the dune area

Table 4 Topsoil mineralization rates for the greenspace categories. α, β = indication of statistical similarity or difference

Categorization	Categories	Mean topsoil mineralization rates [C m^{-2} day^{-1}]
Land cover (Kruskal–Wallis test, $p < 0.01$)	Trees	3.17 ± 0.35 (α)
	Shrubs	3.61 ± 0.47 (α)
	Herbaceous vegetation	1.57 ± 0.14 (β)
Land ownership (Wilcoxon rank sum test, $p < 0.05$)	Privately owned	3.66 ± 0.57 (α)
	Publicly owned	2.52 ± 0.24 (β)
Litter management (Wilcoxon rank sum test, $p < 0.05$)	Litter undisturbed	3.40 ± 0.38 (α)
	Litter removed	2.04 ± 0.14 (β)

the studies differed significantly, making them not directly comparable as C concentrations vary non-linearly with depth (Renforth et al. 2011).

On average, the soil C density measured in the topsoil of the greenspaces in The Hague was significantly higher than the global average soil C storage in temperate oceanic cities reported in relatively undisturbed greenspaces (~ 5 kg m^{-2} in the upper 30 cm) and slightly higher than in highly managed greenspaces (~ 9 kg m^{-2} in the upper 30 cm) (Chien and Krumins 2022). This suggests that appropriate greenspace management may lead to higher SOC levels.

It also appears that the urban soils of The Hague are not SOC impoverished due to anthropogenic influences, but instead are similar to or even exceed the C levels of non-urban soils. For Dutch urban soils, Lof et al. (2017) assumed a soil C stock of factor 0.9 of the SOC stock of the respective soil type, which was based on the widely held assumption that urban soils are SOC impoverished due to anthropogenic influences. For urban greenspaces in The Hague, the data suggests that this assumption would result in an underestimation of the current soil C stock. Most of the urban soils of greenspaces in The Hague were relatively undisturbed, i.e. vegetation was predominantly permanent, SOC stocks were sufficient for soil functioning (> 1.5%, Lal 2016), and soil compaction was limited and thereby did not restrict root growth. This led to urban SOC stock in the Hague that were 1.4 to 2.5 times larger than assumed by Lof et al. (2017) for the 'Bekeerd', 'Vlakvaag' and 'Duinvaag' soils (comparison made on highest level of classification). 'Meerveen' soils on the other hand contained lower C stocks than assumed. 'Meerveen' soils are not typical peat soils as they consist of a mineral topsoil on top of a peaty layer and may therefore not be represented well by the peaty head class. Moreover, the urban SOC concentration exceeded the C concentrations in Dutch grasslands, croplands, and nature for all soil types except for the peaty 'Meerveen' soils (Conijn and Lesschen 2015; Lesschen et al. 2012), which is in line with Lindén et al. (2020), Edmondson et al. (2014) and Cambou et al. (2018) who reported higher SOC levels in urban ecosystems than in adjacent agricultural grasslands, cropland or upland forest soils. The lower SOC concentrations in agricultural soil may reflect the long-term effect of agricultural practices, such as ploughing, application of chemical fertilizers and crop removal, on SOC content and soil quality (Edmondson et al. 2014; Lal 2009).

Soil C densities in the different types of greenspaces

SOC densities of the topsoil differed significantly in the following urban greenspace categories: land cover, soil type and urbanization extent. No differences in C densities were detected for the categories land ownership, land use, litter management, and size of the greenspace (Fig. 3).

Concerning the land cover, shrubs outperformed trees and herbaceous vegetation considering SOC accumulation (Fig. 3A). However, assuming that the quality of shrub litter and level of root functioning are promoting SOM formation is too simplistic. Rather, it is assumed that SOM accumulation is the result of favourable environmental, abiotic and biotic factors of the soil and vegetation combined (Lehmann and Kleber 2015). One important abiotic factor may be the increased nutrient concentrations of soils beneath shrubs (N, P, S). The higher topsoil mineralization rates of soil beneath shrubs and trees than in the soils beneath herbaceous vegetation also demonstrates that these soil C pools are in flux. Whether this leads to an accumulation or reduction of SOC stocks over time requires further research on potential C sequestration rates to get a full overview of the soil's C balance.

Greater C accumulation beneath shrubs is not in line with the pattern commonly observed in non-urban ecosystems where usually the SOC densities are highest in woodlands (Bell et al. 2011). However, this pattern is consistent with Lindén et al. (2020) who also reported higher C densities beneath shrubbery than beneath trees and herbaceous vegetation in the urban soils of Helsinki, Finland. Although, Lindén et al. (2020) could not distinguish whether the different SOC stocks were the result of management or the vegetation itself. Edmondson et al. (2014) on the other hand hypothesized higher SOC densities beneath trees than grassland but detected no difference in the urban soils of Leicester, the U.K. These findings contrast the widespread idea of tree planting to increase the provision of urban ecosystem services, although increasing tree cover may have a positive effect on aboveground C storage (Davies et al. 2011; Edmondson et al. 2014; Hutyra et al. 2011).

Regarding the soil types, this study did not classify the substrate types, as only the upper 30 cm of soil were sampled. Instead, this study hypothesized that the national soil map of the Netherlands could be extrapolated over the urban area of The Hague. It was presumed that the more developed 'Meerveen' (mineral topsoil on top of eutrophic peat layer) and 'Beekeerd' (nutrient-rich humus layer on top of nutrient-poor sandy layer, dominated by oxidation processes) soils would contain higher SOC densities than the less developed 'Vlakvaag' and 'Duinvaag' soils (poorly developed sandy soils). However, in this sample campaign, only the 'Beekeerd' soil contained significantly higher SOC densities than the other soil types (Fig. 3B).

The rejected hypothesis of using soil maps to estimate SOC storage may result from the fact that urban soils are often constructed. Especially in greenspaces that were used as playgrounds, it was clearly visible that the upper 20 cm consisted of allochthonous soil with a different texture and colour. Over time, mixing may occur due to the burrowing activity of soil fauna, especially by soil invertebrates, such as earthworms, which was observed at some of the older sites. However, at some of the younger or recently redecorated greenspaces, the external top layer did not visibly mix yet, which implied minimal influence of the original substrate. These findings have implications for soil C modelling as the extrapolation of soil maps over urban areas may not be the most appropriate approach to estimate soil C stocks. It also has significance for soil C stock estimations as the buried horizon, i.e. the former topsoil, or peaty subsoils in the case of the 'Meerveen' soils may contain significant amounts of C. Additionally, where peaty soils were expected, the applied sampling method could not confirm whether this was the case. 'Meerveen' soils consist of mineral topsoil on top of a nutrient-rich peat layer (~ 60 cm deep). As only the upper 30 cm of soil was sampled, the soil type could not be confirmed. Plausibly, the SOC concentration in the upper 30 cm of the mineral layer of 'Meerveen' soil did not accurately reflect the C concentrations of the entire soil profile.

Considering land ownership, this study detected higher soil total C densities in the soils of publicly owned greenspaces than in those of privately owned greenspaces. However, when looking at SOC only, land ownership did not result in significantly different SOC densities (Fig. 3C). SIC densities on the other hand were significantly higher in publicly owned greenspaces than privately owned greenspaces, which is likely due to their closer location to roads, and thus higher susceptibility to dust inputs. Moreover, topsoil mineralization rates were higher in privately owned greenspaces than in publicly owned greenspaces which may explain the observed differences in C stocks and may reflect the different greenspace management practices of the landowners.

In contrast with Edmondson et al. (2014), Rawlins et al. (2008) and Pouyat et al. (2009), this study did not find that privately owned greenspaces contained higher SOC contents than publicly owned greenspaces, which is likely because this study did not include private domestic gardens. Rather, the distinction between private and public was made based on whether greenspaces were managed by the municipality. Private greenspaces in this study entailed communally owned greenspaces such as communal gardens and cemeteries.

Regarding the urbanization extent, larger C densities were detected in the city centre than in the suburbs of The Hague (Fig. 3D). This pattern differed for several urban soil studies; for example in Berlin the suburbs contained higher C densities than the city centre (Richter et al. 2020), but in Paris, the city centre contained higher SOC densities than the suburbs (Cambou et al. 2018). For Berlin, higher SOC densities in the suburbs likely resulted from management effects in the large domestic gardens that are typical for the suburbs of Berlin (Richter et al. 2020). In Paris, the higher SOC densities are explained by the substrate origin; city centre greenspaces were constructed with soil rich in SOM and suburban greenspaces were constructed with soils poorer in SOM (Cambou et al. 2018). This historic origin was also likely the case for The Hague in combination with the fact that the original substrate was also C poorer in the suburbs.

Concerning greenspace management, this study detected no pronounced differences in SOC storage under different urban greenspace management practices (data not presented; Table 2). Investigating the impact of urban greenspace management on SOC storage was complicated because urban greenspace management packages may have reverse effects on SOC stocks, which is for example observed in the maintenance of lawns that incorporate fertilization, but also the removal of grass clippings. These reverse responses make it difficult to predict its effects on SOC (Lindén et al. 2020). However, when simplifying urban greenspace management to litter management only, a pronounced effect was found in several soil parameters (LOI, DOC, pH, water-holding capacity; Supplementary Information: Fig. S4) and topsoil mineralization rates). However, these conditions did not lead to significantly different SOC densities in this study.

Degradability of urban soil organic matter

Investigating why some SOM persists for a long time and other SOM degrades readily is a prerequisite to predicting SOM stock's response to climate change (Schmidt et al. 2011; Wan et al. 2020). In this study, mineralization normalized to SOC was used to assess the potential degradability of urban SOM. For SOM to contribute to long-term C storage, the formation of stable (recalcitrant) SOM is not the only mechanism. The dynamic soil C stock can be enlarged by either increasing the C inflows or by decreasing the C outflows (Janzen 2015), with this study presenting first data to assess the latter, the mineralization of SOC. Since the standardised conditions during laboratory measurements may be more favourable than those in the field, for example with respect to oxygen supply, temperature and availability of water, the measured respiration rates likely exceed in situ C fluxes. Although mineralization rates may be overestimated, the experimental design allowed for comparison between the samples collected from the different plots along the transect.

Potential mineralization rates of SOM differed along the transect (Fig. 4), which suggests different C availability for decomposer organisms during the incubation period (Lehman and Kleber 2015; Saviozzi et al. 2014). The highest mineralization rates normalized to SOC were found in the sandy dunes, suggesting a high SOM degradability. However, when translated to potential topsoil mineralization rates the dunes emitted the lowest amount of C, as the initial SOC concentrations of the dunes were very low (Fig. 2A). The mean SOC concentration of the dune samples (0.35%) was comparable to the values reported in similar dune vegetation in the region (0.44%, de Vries 1993). The dunes are a relatively young ecosystem consisting of soil with a coarse texture and a low water-holding capacity, which makes the chemical and physical protection of SOM from decomposition minimal (Barré et al. 2014; Zacháry et al. 2018). Dune systems are thus of low relevance for soil C sequestration.

The highest topsoil mineralization rates were found in the urban forest, despite their lower C densities compared to the city centre and suburbs (Figs. 2C and 4). Soil properties that could explain the relatively high topsoil mineralization rates in the urban forest are the presence of thick litter layers in combination with the relatively high water-holding capacity, the low pH, the high C:P ratio of the urban forest soil and potentially the susceptibility to urban pollution of the urban forest soil.

The effect of litter management on the topsoil mineralization rates was pronounced even though litter management did not result in significant different SOC densities (Fig. 2G). After 6 weeks of incubation, the analysis quantified a topsoil C mineralization of 86 ± 5.9 mg C m^{-2} for the soils depleted from plant litter and 143 ± 16 mg C m^{-2} for the soils naturally augmented with plant litter, revealing that urban soils that received plant litter possessed a higher SOC turnover rate. Thick litter layers are also associated with higher water-holding capacities of the soil (Supplementary Information: Fig. S4), which is an important resource for the decomposer community (Miltner et al. 2012, r_s between topsoil mineralization rate and water-holding capacity = 0.72).

The urban forest soil was strongly acidified with a mean pH of 5.2, with locally extremely acidic conditions of 3.8 (Fig. 2G). The pH inversely correlated with the topsoil mineralization rates, which is in line with Saviozzi et al. (2014) who incubated Italian urban soil. Soil pH strongly affects C and nutrient availability and the solubility of metals (Rousk et al. 2009). Moreover, when soil acidifies, the soil microbial community shifts from a balance between soil bacteria and fungi to a fungal-dominated soil, which changes the way organic matter is decomposed (Francini et al. 2018; Rousk et al. 2009). A fungal-dominated soil is characterized by slow nutrient cycling and a high capacity to retain nutrients (de Vries et al. 2006).

Additionally, the C:P ratio was significantly higher in the urban forest than in the remaining greenspaces (81:1 vs. 55:1, Supplementary Information, Fig. S3) and the C:P ratio strongly correlated with the topsoil mineralization rates ($r_s = 0.83$). The influence of P on C mineralization in urban forests was also investigated by Chen et al. (2014) who observed higher C mineralization under P enrichment in organic matter in urban sites. What caused the relatively high P levels in the urban forest of The Hague is unclear as the forest is not managed with fertilizers. Whether it is the litter layer, pet waste pollution, plant-symbiotic fungi that thrive in acidic soils, or input of P via atmospheric deposition as dust that is captured by the forest's canopy (Sohrt et al. 2017; Theobald et al. 2016) requires further investigation.

Relatively high C mineralization rates in urban forests are in line with Kim and Yoo (2020) who measured higher soil respiration rates in urban forest compared to roadside tree systems, although they measured respiration in the field using the chamber method, making the results not directly comparable. Kim and Yoo (2020) added that it may be more difficult for soil microorganisms to mineralize organic materials in roadside soils than in urban forests because roadside soils may be more susceptible to urban pollutants which are likely inhibiting microbial activity (Lovett et al. 2000; Trammell et al. 2017; Zhao et al. 2013).

Nutrient availability in the urban context

Nutrient availability is critical for soil C sequestration (Kirkby et al. 2013). SOM in the soil fraction smaller than 4 mm is believed to have a nearly constant C:N, C:P and C:S ratio of 12, 50 and 70 respectively, which suggests that at these nutrient proportions, humification occurs most effectively (Kirkby et al. 2011). This optimum implies that for each tonne of sequestered soil C, the soil approximately co-sequesters 80, 20 and 14 kg of N, P, and S, respectively (Kirkby et al. 2011). Moreover, a higher C:N:P:S ratio than the humification optimum may result in C and nutrient losses to the atmosphere after organic amendments aimed at increasing SOM stocks. This loss is due to the positive priming effect, which is caused by the response of soil microorganisms to the fresh organic inputs (e.g. co-metabolism, microbial mining, Kirkby et al.

2014). The urban soils of the Hague had mean C:N, C:P and C:S ratios of 15, 55 and 55 respectively, which meant that N and P may be limiting factors in C sequestration.

Moreover, the degradability of SOM negatively correlated with the C, N, P and S concentration of the soil (Table 3), which suggests that SOC accumulates where the mineralization conditions are less favourable for mineralization. The significant correlations between C mineralization and SOC and N were confirmed by Ahn et al. (2009) and Zacháry et al. (2018). Conversely, the C:N ratio of the soil only weakly correlated with the C mineralization (Table 3), which also is in line with Zacháry et al. (2018) who state that these ratios are likely a less good indicator for the 'protected' C pools. Finally, the strong correlation between C and N and S, and moderate correlation between C and P further confirmed the dependency between C and those nutrients in the urban context (Table 3). The C to nutrient ratio of the urban soils of The Hague implies that opportunities exist to improve C sequestration rates (Kirkby et al. 2013, 2016).

Conclusion

This study reports the first urban soil C balance in the Netherlands. The use of high spatial resolution GIS data enabled the inclusion of small patches in the total soil C storage of The Hague, which turned out to be significant as the smaller urban greenspaces, which are typical for dense urban centres, contained similar soil C density as the larger urban greenspaces. The greenspaces located in the city centre and suburban district of The Hague (total area of 20.8 km^2 of which 7.6 km^2 is greenspace) stored 78.4 kt C in the upper 30 cm of soil, which is significantly larger than estimated by current modelling studies (weighted average of Lof et al. (2017) based on soil type coverage of 7.3 kt SOC km^{-2} * 7.6 km^2 = 55.5 kt C). This shows that despite their presence in a severely urbanized environment, the urban soil and their greenspace management function to retain these relatively high soil C levels. Soils beneath shrubs proved to be significant hotspots for C storage as they only accounted for 13% of the land cover but were responsible for almost a quarter of the total C storage. Based on the patterns commonly observed in non-urban soils, it was hypothesized that soil type and land use would be appropriate predictors for urban soil C storage. However, these hypothesized links were not apparent in this case study, revealing the significance of persistent anthropogenic influences of the urban area on soil C stocks.

Although urban soil can be highly disturbed or altered by anthropogenic activities, the relatively high C densities in The Hague suggest that its potential to store C appears unaffected. Urban soils are therefore believed to be capable of playing an important role in soil C sequestration and thereby support the ecosystem service of climate regulation. Further research on C dynamics is required to quantify urban soil C sequestration rates. As the measured mineralization rate in this study presents potentials under laboratory conditions, it is recommended to further study the mineralization rates of urban soils over a longer time frame to incorporate the governing factors, such as temperature and water availability. Additionally, to contribute to natural soil C inventories, it is recommended to expand the sampling campaign in sample size and sample depth. Subsoils may contain more C because of their greater depths, but also because former topsoil may have been buried in subsoils during the construction of greenspaces (Bae and Ryu 2015; Cambou et al. 2018; Vasenev and Kuzyakov 2018; Zhu et al. 2017). Expanding the sampling campaign will also contribute to the development of a reliable set of parameters to estimate urban soil C stocks in temperate climates.

The positive association between SOC levels and other soil-derived ecosystem services and the high SOC levels in urban ecosystems suggests that the urban ecosystem services have potentially been undervalued. It is therefore advised to acknowledge urban soils and their soil C stocks as a valuable resource in urban greenspace management as this recognition may lead to more resilient urban ecosystems.

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References

Ahn MY, Zimmerman AR, Comerford NB, Sickman JO, Grunwald S (2009) Carbon mineralization and labile organic carbon pools in the sandy soils of a North Florida watershed. Ecosystems 12(4):672–685. https://doi.org/10.1007/s10021-009-9250-8

Ajmone Marsan F, Certini G, Scalenghe R (2016) Describing urban soils through a faceted system ensures more informed decision-making. Land Use Policy 51:109–119. https://doi.org/10.1016/j.landusepol.2015.10.025

Bae J, Ryu Y (2015) Land use and land cover changes explain spatial and temporal variations of the soil organic carbon stocks in a constructed urban park. Landsc Urban Plan 136:57–67. https://doi.org/10.1016/j.landurbplan.2014.11.015

Barré P, Fernandez-Ugalde O, Virto I, Velde B, Chenu C (2014) Impact of phyllosilicate mineralogy on organic carbon stabilization in soils: incomplete knowledge and exciting prospects. Geoderma 235:382–395. https://doi.org/10.1016/j.geoderma.2014.07.029

Beesley L (2012) Carbon storage and fluxes in existing and newly created urban soils. J Environ Manage 104:158–165. https://doi.org/10.1016/j.jenvman.2012.03.024

Bell MJ, Worrall F, Smith P, Bhogal A, Black H, Lilly A, Barraclough D, Merrington G (2011) UK land-use change and its impact on SOC: 1925–2007. Global Biogeochem Cycles 25:4. https://doi.org/10.1029/2010GB003881

Boden A-H (2005) Bodenkundliche Kartieranleitung, 5th edn. Bundesanstalt für Geowissenschaften und Rohstoffe und Geologische Landesämter der Bundesrepublik Deutschland, Schweitzerbartsche Verlagsbuchhandlung, Stuttgart

BS 1377–2:1990 (1990) Methods of test for soils for civil engineering purposes — Part 2: classification tests

CBS (2015) Centraal Bureau voor Statistiek. https://nationaalgeoregister.nl/geonetwork/srv/dut/catalog.search#/metadata/2d3dd6d2-2d2b-4b5f-9e30-86e19ed77a56?tab=general

CBS (2022) Centraal Bureau voor Statistiek. https://www.cbs.nl/nl-nl/onze-diensten/methoden/onderzoeksomschrijvingen/korte-onderzoeksbeschrijvingen/kerncijfers-wijken-en-buurten

Cambou A, Shaw RK, Huot H, Vidal-Beaudet L, Hunault G, Cannavo P, Nold F, Schwartz C (2018) Estimation of soil organic carbon stocks of two cities, New York City and Paris. Sci Total Environ 644:452–464. https://doi.org/10.1016/j.scitotenv.2018.06.322

Canedoli C, Ferre C, El Khair DA, Padoa-Schioppa E, Comolli R (2020) Soil organic carbon stock in different urban land uses: high stock evidence in urban parks. Urban Ecosystems 23(1):159–171. https://doi.org/10.1007/s11252-019-00901-6

Chen FS, Yavitt J, Hu XF (2014) Phosphorus enrichment helps increase soil carbon mineralization in vegetation along an urban-to-rural gradient, Nanchang, China. Appl Soil Ecol 75:181–188. https://doi.org/10.1016/j.apsoil.2013.11.011

Chien SC, Krumins JA (2022) Natural versus urban global soil organic carbon stocks: A meta-analysis. Sci Total Environ 807:150999. https://doi.org/10.1016/j.scitotenv.2021.150999

Churkina G, Brown DG, Keoleian G (2010) Carbon stored in human settlements: the conterminous United States. Glob Change Biol 16(1):135–143. https://doi.org/10.1111/j.1365-2486.2009.02002.x

Conijn JG, Lesschen JP (2015) Soil organic matter in the Netherlands: Quantification of stocks and flows in the top soil. Plant Research International, Business Unit Agrosystems Research 619

Data.overheid (2019) Luchtfoto 2019 25cm CIR open data. Open data van de Overheid. https://data.overheid.nl/dataset/efdd769e-a76e-4994-b5f2-7bb19a8d6b5d

Davies ZG, Edmondson JL, Heinemeyer A, Leake JR, Gaston KJ (2011) Mapping an urban ecosystem service: quantifying above-ground carbon storage at a city-wide scale. J Appl Ecol 48(5):1125–1134. https://doi.org/10.1111/j.1365-2664.2011.02021.x

de Bakker H, Schelling J, Brus DJ, van Wallenburg C (1989) Systeem voor bodemclassificatie voor Nederland; de hogere niveaus. Wageningen, Winand Staring Centre

Derkzen ML, van Teeffelen AJ, Verburg PH (2015) Quantifying urban ecosystem services based on high-resolution data of urban green space: an assessment for Rotterdam, the Netherlands. J Appl Ecol 52(4):1020–1032. https://doi.org/10.1111/1365-2664.12469

de Vries W (1993) De chemische samenstelling van bodem en bodemvocht van duingronden in de provincie Zuid-Holland. DLO-Staring Centrum 280

de Vries FT, Hoffland E, van Eekeren N, Brussaard L, Bloem J (2006) Fungal/bacterial ratios in grasslands with contrasting nitrogen management. Soil Biol Biochem 38(8):2092–2103. https://doi.org/10.1016/j.soilbio.2006.01.008

Edmondson JL, Davies ZG, McHugh N, Gaston KJ, Leake JR (2012) Organic carbon hidden in urban ecosystems. Sci Rep 2:963. https://doi.org/10.1038/srep00963

Edmondson JL, Davies ZG, McCormack SA, Gaston KJ, Leake JR (2014) Land-cover effects on soil organic carbon stocks in a European city. Sci Total Environ 472:444–453. https://doi.org/10.1016/j.scitotenv.2013.11.025

Francini G, Hui N, Jumpponen A, Kotze DJ, Romantschuk M, Allen JA, Setälä H (2018) Soil biota in boreal urban greenspace: Responses to plant type and age. Soil Biol Biochem 118:145–155. https://doi.org/10.1016/j.soilbio.2017.11.019

Hansen J, Kharecha P, Sato M, Masson-Delmotte V, Ackerman F, Beerling DJ et al (2013) Assessing "dangerous climate change": Required reduction of carbon emissions to protect young people, future generations and nature. PLoS ONE 8:12. https://doi.org/10.1371/journal.pone.0081648

Herrmann, DL, Schifman LA, Shuster WD (2020) Urbanization drives convergence in soil profile texture and carbon content. Environ Res Let 15.11:114001. https://doi.org/10.1088/1748-9326/abbb00

Hoffland E, Kuyper TW, Comans RN, Creamer RE (2020) Eco-functionality of organic matter in soils. Plant Soil 455:1–22. https://doi.org/10.1007/s11104-020-04651-9

Hutyra LR, Yoon B, Alberti M (2011) Terrestrial carbon stocks across a gradient of urbanization: a study of the Seattle. WA Region Global Change Biology 17(2):783–797. https://doi.org/10.1111/j.1365-2486.2010.02238.x

ISO 11465 (1993) Soil quality — Determination of dry matter and water content on a mass basis — Gravimetric method

ISO 11265 (1994) Soil quality — Determination of the specific electrical conductivity

ISO 10694 (1996) Soil quality — Determination of organic and total carbon after dry combustion (elementary analysis)

ISO 10390 (2005) Soil quality — Determination of pH

Janzen HH (2015) Beyond carbon sequestration: soil as conduit of solar energy. Eur J Soil Sci 66(1):19–32. https://doi.org/10.1111/ejss.12194

Kim YJ, Yoo G (2020) Suggested key variables for assessment of soil quality in urban roadside tree systems. J Soils Sediments 21(5):2130–2140. https://doi.org/10.1007/s11368-020-02827-5

Kirkby CA, Kirkegaard JA, Richardson AE, Wade LJ, Blanchard C, Batten G (2011) Stable soil organic matter: a comparison of C: N: P: S ratios in Australian and other world soils. Geoderma 163(3–4):197–208. https://doi.org/10.1016/j.geoderma.2011.04.010

Kirkby CA, Richardson AE, Wade LJ, Batten GD, Blanchard C, Kirkegaard JA (2013) Carbon-nutrient stoichiometry to increase soil carbon sequestration. Soil Biol Biochem 60:77–86. https://doi.org/10.1016/j.soilbio.2013.01.011

Kirkby CA, Richardson AE, Wade LJ, Passioura JB, Batten GD, Blanchard C, Kirkegaard JA (2014) Nutrient availability limits carbon sequestration in arable soils. Soil Biol Biochem 68:402–409. https://doi.org/10.1016/j.soilbio.2013.09.032

Kirkby CA, Richardson AE, Wade LJ, Batten GD, Blanchard CL, Mclaren G, Zwart AB, Kirkegaard JA (2016) Accurate

measurement of resistant soil organic matter and its stoichiometry. Eur J Soil Sci 67(5):695–705. https://doi.org/10.1111/ejss.12378
Lal R (2009) Challenges and opportunities in soil organic matter research. Eur J Soil Sci 60(2):158–169. https://doi.org/10.1111/j.1365-2389.2008.01114.x
Lal R (2016) Resilience: Quality and. In: Lal R (ed) Encyclopedia of soil science, 3rd edn. CRC Press, pp 1918–1923
Lehmann J, Kleber M (2015) The contentious nature of soil organic matter. Nature 528(7580):60–68. https://doi.org/10.1038/nature16069
Lesschen JP, Heesmans HIM, Mol-Dijkstra JP, van Doorn AM, Verkaik E, van den Wyngaert IJJ, Kuikman PJ (2012) Mogelijkheden voor koolstofvastlegging in de Nederlandse landbouw en natuur. Alterra 2396
Lindén L, Riikonen A, Setälä H, Yli-Pelkonen V (2020) Quantifying carbon stocks in urban parks under cold climate conditions. Urban Forestry & Urban Greening 49:126633. https://doi.org/10.1016/j.ufug.2020.126633
Lof M, Schenau S, de Jong R, Remme R, Graveland C, Hein L (2017) The SEEA EEA carbon account for the Netherlands. Statistics Netherlands
Lorenz K, Lal R (2015) Managing soil carbon stocks to enhance the resilience of urban ecosystems. Carbon Management 6(1–2):35–50. https://doi.org/10.1080/17583004.2015.1071182
Lorenz K (2016) Urban Lands: Management. In: Lal R (ed) Encyclopedia of soil science, 3rd edn. CRC Press, pp 2400–2406
Lovett GM, Traynor MM, Pouyat RV, Carreiro MM, Zhu WX, Baxter JW (2000) Atmospheric deposition to oak forests along an urban−rural gradient. Environ Sci Technol 34(20):4294–4300. https://doi.org/10.1021/es001077q
Miltner A, Bombach P, Schmidt-Brücken B, Kästner M (2012) SOM genesis: microbial biomass as a significant source. Biogeochemistry 111(1):41–55. https://doi.org/10.1007/s10533-011-9658-z
Minasny B, Malone BP, McBratney AB, Angers DA, Arrouays D, Chambers A et al (2017) Soil carbon 4 per mille. Geoderma 292:59–86. https://doi.org/10.1016/j.geoderma.2017.01.002
Morel JL, Chenu C, Lorenz K (2015) Ecosystem services provided by soils of urban, industrial, traffic, mining, and military areas (SUITMAs). J Soils Sediments 15(8):1659–1666. https://doi.org/10.1007/s11368-014-0926-0
NEN-EN-ISO 10693 (2014) Bodem – Bepaling van het gehalte aan carbonaten – Volumetrische methode
NEN-ISO 10694 (2008) Bodem – Bepaling van organisch en totaal koolstofgehalte na droge verassing (elementaire analyse)
NEN 6966 (2005) Milieu – Analyse van geselecteerde elementen in water, eluaten en destruaten – Atomaire emissiepectrometrie met inductief gekoppeld plasma
O'Riordan R, Davies J, Stevens C, Quinton JN, Boyko C (2021) The ecosystem services of urban soils: A review. Geoderma 395:115076. https://doi.org/10.1016/j.geoderma.2021.115076
Pavao-Zuckerman MA, Coleman DC (2005) Decomposition of chestnut oak (Quercus prinus) leaves and nitrogen mineralization in an urban environment. Biol Fertil Soils 41(5):343–349. https://doi.org/10.1007/s00374-005-0841-z
Peng J, Tian L, Liu Y, Zhao M, Wu J (2017) Ecosystem services response to urbanization in metropolitan areas: Thresholds identification. Sci Total Environ 607:706–714. https://doi.org/10.1016/j.scitotenv.2017.06.218
Pouyat R, Groffman P, Yesilonis I, Hernandez L (2002) Soil carbon pools and fluxes in urban ecosystems. Environ Pollut 116:S107–S118. https://doi.org/10.1016/S0269-7491(01)00263-9
Pouyat RV, Yesilonis ID, Golubiewski NE (2009) A comparison of soil organic carbon stocks between residential turf grass and native soil. Urban Ecosystems 12(1):45–62. https://doi.org/10.1007/s11252-008-0059-6
Rawlins BG, Vane CH, Kim AW, Tye AM, Kemp SJ, Bellamy PH (2008) Methods for estimating types of soil organic carbon and their application to surveys of UK urban areas. Soil Use Manag 24(1):47–59. https://doi.org/10.1111/j.1475-2743.2007.00132.x
Rawlins BG, Harris J, Price S, Bartlett M (2015) A review of climate change impacts on urban soil functions with examples and policy insights from England, UK. Soil Use Manag 31:46–61. https://doi.org/10.1111/sum.12079
Renforth P, Leake JR, Edmondson J, Manning DA, Gaston KJ (2011) Designing a carbon capture function into urban soils. Proceedings of the ICE-Urban Design and Planning 164(2):121–128. https://doi.org/10.1680/udap.2011.164.2.121
Richter S, Haase D, Thestorf K, Makki M (2020) Carbon Pools of Berlin, Germany: Organic Carbon in Soils and Aboveground in Trees. Urban Forestry & Urban Greening 54:126777. https://doi.org/10.1016/j.ufug.2020.126777
RIVM (2017) Groenkaart van Nederland. Atlas Leefomgeving. https://www.atlasleefomgeving.nl//kaarten?config=3ef897de-127f-471a-959b-93b7597de188&gm-x=150000&gm-y=455000&gm-z=3&gm-b=1544180834512,true,1%3B1544725575974,true,0.8%3B&activateOnStart=layermanager,info
Rousk J, Brookes PC, Baath E (2009) Contrasting soil pH effects on fungal and bacterial growth suggest functional redundancy in carbon mineralization. Appl Environ Microbiol 75(6):1589–1596. https://doi.org/10.1128/AEM.02775-08
Sanderman J, Hengl T, Fiske GJ (2017) Soil carbon debt of 12,000 years of human land use. Proc Natl Acad Sci 114(36):9575–9580. https://doi.org/10.1073/pnas.1706103114
Sander R (2015) Compilation of Henry's law constants (version 4.0) for water as solvent. Atmos Chem Phys 15(8):4399-4981. https://doi.org/10.5194/acp-15-4399-2015
Saviozzi A, Vanni G, Cardelli R (2014) Carbon mineralization kinetics in soils under urban environment. Appl Soil Ecol 73:64–69. https://doi.org/10.1016/j.apsoil.2013.08.007
Schmidt MW, Torn MS, Abiven S, Dittmar T, Guggenberger G, Janssens IA et al (2011) Persistence of soil organic matter as an ecosystem property. Nature 478(7367):49. https://doi.org/10.1038/nature10386
Seto KC, Güneralp B, Hutyra LR (2012) Global forecasts of urban expansion to 2030 and direct impacts on biodiversity and carbon pools. Proc Natl Acad Sci 109(40):16083–16088. https://doi.org/10.1073/pnas.1211658109
Shchepeleva AS, Vasenev VI, Mazirov IM, Vasenev II, Prokhorov IS, Gosse DD (2017) Changes of soil organic carbon stocks and CO2 emissions at the early stages of urban turf grasses' development. Urban Ecosystems 20(2):309–321. https://doi.org/10.1007/s11252-016-0594-5
Sohrt J, Lang F, Weiler M (2017) Quantifying components of the phosphorus cycle in temperate forests. Wiley Interdiscip Rev Water 4(6):e1243. https://doi.org/10.1002/wat2.1243
Tan PY, Zhang J, Masoudi M, Alemu JB, Edwards PJ, Grêt-Regamey A, Richards DR, Saunders J, Song XP, Wong LW (2020) A conceptual framework to untangle the concept of urban ecosystem services. Landsc Urban Plan 200:103837. https://doi.org/10.1016/j.landurbplan.2020.103837
Theobald TF, Schipper M, Kern J (2016) Phosphorus flows in Berlin-Brandenburg, a regional flow analysis. Resour Conserv Recycl 112:1–14. https://doi.org/10.1016/j.resconrec.2016.04.008
Trammell TL, Day S, Pouyat, RV, Rosier C, Scharenbroch B, Yesilonis I (2017) Drivers of urban soil carbon dynamics. In: Lal R, Steward BA (eds) Urban soils. Adv Soil Sci 1st edn. Taylor & Francis Group, Boca Raton, FL 93–120
van Droesberg C (2017) Groenbeheersysteem Den Haag. Beheerdershandleiding. Beheersysteem GISIB
van Oorschot J, Sprecher B, van't Zelfde M, van Bodegom PM, van Oudenhoven AP, (2021) Assessing urban ecosystem services in support of spatial planning in the Hague, the Netherlands. Landsc Urban Plan 214:104195. https://doi.org/10.1016/j.landurbplan.2021.104195

Vauramo S, Setälä H (2011) Decomposition of labile and recalcitrant litter types under different plant communities in urban soils. Urban Ecosys 14(1):59–70. https://doi.org/10.1007/s11252-010-0140-9

Vasenev V, Kuzyakov Y (2018) Urban soils as hot spots of anthropogenic carbon accumulation: Review of stocks, mechanisms and driving factors. Land Degrad Dev 29(6):1607–1622. https://doi.org/10.1002/ldr.2944

Wan SZ, Chen FS, Hu XF, Zhang Y, Fang XM (2020) Urbanization aggravates imbalances in the active C, N and P pools of terrestrial ecosystems. Global Ecology and Conservation 21:e00831. https://doi.org/10.1016/j.gecco.2019.e00831

Weissert LF, Salmond JA, Schwendenmann L (2016) Variability of soil organic carbon stocks and soil CO2 efflux across urban land use and soil cover types. Geoderma 271:80–90. https://doi.org/10.1016/j.geoderma.2016.02.014

WRB (2015) International soil classification system for naming soils and creating legends for soil maps. World Soil Resources Report 106, FAO, Rome

Yoon TK, Seo KW, Park GS, Son YM, Son Y (2016) Surface soil carbon storage in urban green spaces in three major South Korean cities. Forests 7(6):115. https://doi.org/10.3390/f7060115

Zacháry D, Filep T, Jakab G, Varga G, Ringer M, Szalai Z (2018) Kinetic parameters of soil organic matter decomposition in soils under forest in Hungary. Geoderma Reg 14:e00187. https://doi.org/10.1016/j.geodrs.2018.e00187

Zhao D, Li F, Yang Q, Wang R, Song Y, Tao Y (2013) The influence of different types of urban land use on soil microbial biomass and functional diversity in Beijing. China Soil Use and Management 29(2):230–239. https://doi.org/10.1111/sum.12034

Zhu W, Hulisz P, Egitto BA, Yesilonis ID, Pouyat RV, Lal R, Stewart BA (2017) Soil carbon and nitrogen cycling and ecosystem service in cities. In: Lal R, Steward BA (eds) Urban soils. Advances in soil science. 1st edn. Taylor & Francis Group, Boca Raton, FL 121–136