Topsoil Carbon Stocks in Urban Greenspaces of The Hague, the Netherlands

Anniek J Kortleve (a.j.kortleve@cml.leidenuniv.nl)
Universiteit Leiden Faculteit der Wiskunde en Natuurwetenschappen https://orcid.org/0000-0003-4617-2281
José M Mogollón
Universiteit Leiden Faculteit der Wiskunde en Natuurwetenschappen
Timo J Heimovaara
Delft University of Technology: Technische Universiteit Delft
Julia Gebert
Delft University of Technology: Technische Universiteit Delft

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Abstract

This research studied 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 sandy dune soils to peaty inland soils. C storage and C mineralisation potential were evaluated in relation to soil type and properties, land ownership, type of vegetation, litter management and eco-zone status. 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.

The total soil C storage of the upper 30 cm of the greenspaces in The Hague was estimated at 23.5 kt. Degradability of soil organic matter in laboratory batch tests varied between 0.2 and 2 mg C g_{SOC}^{-1} d^{-1}. Degradability was highest in the seaside dune soils; however, extrapolated to the top 30 cm using C density, potential 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 almost a quarter of the total C storage. Along the transect, a mean soil C density of 110 t ha^{-1} was measured, of which 98 t ha^{-1} was organic C. Land ownership, use, 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 the type of vegetation and the urbanization extent.

1 Introduction

Soil organic matter (SOM) plays a key role in the delivery of soil-derived ecosystem services, such as biodiversity, primary production, climate regulation, erosion protection and water quality (Rawlins et al. 2015; Hoffland et al., 2020). In the case of urban soils this role is widely neglected (Morel et al., 2015). However, with growing urbanization, the services that society extracts from urban ecosystems are becoming increasingly important (Lorenz 2016). Urban areas are hotspots of anthropogenic carbon (C) emissions (Pouyat et al. 2002) but have the potential to act as significant C sinks (Churkina et al. 2010; Edmondson et al. 2012; Lorenz & Lal 2015; Vasenev & Kuzyakov 2018). Soil C sequestration is a process supporting the ecosystem service of climate regulation (Minasny et al. 2017; Vasenev & Kuzyakov 2018). 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 & Lal 2015; Hansen et al. 2013). Soil C storage can be managed for optimizing ecosystem services (Lorenz 2016).

There has been a substantial historical decline in soil C of approximately 133 Pg C driven by land-use change from natural to agricultural use (Sanderman et al. 2017). With shifts in land use to urban ecosystems outpacing agricultural transformation (Seto et al. 2012), there is a great urgency to understand how SOM pools are altered by urbanization (Hermann et al. 2020; Peng et al. 2017). Accurate assessment of soil C stocks is therefore crucial to understand anthropogenic changes in urban soils in relation to the global C cycle (Edmondson et al. 2012) and the potential of urban greenspaces to trap atmospheric C. Here, urban greenspaces are defined as areas with unsealed soils that encompass all green surfaces areas detected by infrared satellite images.

C storage in urban soils was studied for, 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 several Korean cities (Yoon et al.,2016). These studies reported a high variability in soil C storage and higher or similar urban soil C densities compared to regional, non-urban soils. Canedoli et al. (2020) detected higher C densities in urban parks compared to non-park areas. Soil organic C (SOC) levels in urban parks were comparable with those of regional forests, pastures and grasslands and higher than those in croplands. Edmondson et al. (2012) also observed significant greater C storage in urban soils than regional agricultural land. Yoon et al. (2016) confirm the C stock variability within a country with a total C storage of 106, 44 and 26 kt C in the cities of Seoul, Daegu and Daejeon, respectively. The relatively low C storage potential for those cities was explained by the low soil C concentrations and low share of land area under urban greenspaces. Lastly, Richter et al. (2020) found that when evaluating the distribution of soil C storage across the city of Berlin, the soil C densities increased towards the city's boundaries. For the Netherlands, data on urban soil C storage is lacking, in spite of the interest in the spatial assessment of urban ecosystem services (Derkzen et al. 2015; van Oorschot et al. 2021).

The aim of this study was to quantify the amount of C stored in the topsoil (0–30 cm) of urban spaces and to assess the inter-zone variance of different soil C pools by example of the city of The Hague, the Netherlands. The Hague is part of the most densely populated area in north-western Europe, home to the megalopolis formed by the county's main cities including Amsterdam and Rotterdam and is located on the North Sea coast. The Hague provided the unique opportunity to study urban soil C stocks along a toposequence from sandy dune soils (seaside) to peaty inland soils (city centre). It was hypothesized that urban soil C storage was dependent on the type of soil and the type of greenspace as characterised by size, land use, vegetation type, management, urbanization extent and land ownership. By establishing a basis of soil C stocks in The Hague, the capacity of its soils to capture and store C can be safeguarded and enhanced, raising the benefits that society receives from urban soils (Rawlins et al. 2015).

2 Methods

2.1 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 manually select 25 geo-referenced sample plots. Within each sample plot, three 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. Subsamples were mixed to even out small scale inhomogeneities. The bulk density of each sampling site was determined at the middle point of the subsamples (Fig. 1).

2.2 Standard soil properties

The homogenized soil samples were analysed for standard soil physical and chemical parameters as listed in Table 1.

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	BS 1337-2 (1990)
Water content	ISO 11465 (1993)
Texture	By hand, according to KA5 by Ad-hoc AG Boden (2005) and translated to the FAO (Food and Agriculture Organization)/USDA system.
Water-holding capacity	20 g of field fresh soil in filtered funnels with 100 ml of H_2O for at least 24 hours, gravimetrically after drying at 105°C
pH value	ISO 10390 (2005)
Electrical conductivity (EC)	ISO 11265 (1994)
SOM	Loss-on-ignition (LOI) (ISO 10694, 1995)
Dissolved organic carbon (DOC)	Ultraviolet-visible absorption spectrum at 254 nm
Total C	NEN-ISO 10694 (2008)
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)

2.3 C densities, C storage and degradability of urban SOM

SOC concentrations were transformed to SOC densities of the topsoil (kg C m^{-2}) according to Eq. 1, where SOC is the soil organic C concentration (%), BD is the bulk density (kg m^{-3}) and D the sampling depth (m) (Weissert et al. 2016).

$$Eq.1 : SoilorganicCdensity = SOC * BD * D$$

Using the SOC densities, the green surface area maps of The Hague were transformed into a soil C density map based on the method of Richter et al. (2020). The total soil C stock was computed by multiplying the surface areas of each vegetation type with the corresponding soil C density, according to Eq. 2 where *v* refers to either trees, shrubs or herbaceous vegetation.

$$Eq.2 : TotalsoilorganicCstock = \sum meanSOCdensity_v * surfacearea_v$$

To assess the stability of the soil C stocks, an aerobic incubation procedure was used to determine the potential of urban soils to mineralize SOM. 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). C mineralization rates in relation to the SOC content (mg C g SOC^{-1} day^{-1}) were calculated from the slope of CO_2 concentrations over time using the soil dry mass (g) and the volume of the bottle headspace (ml) according to Eq. 3, where CO_2 and SOC are in percentages and V_m (molar volume) of CO_2 at 20°C = 24.1 l mol^{-1} and *M* (molar mass) of C = 12.0 g mol^{-1}. The C mineralization rate normalized to SOC was used as an indication of the degradability of SOM.

$Eq.3$:C mineralization in relation to SOC = CO_2 / 100 * volume bottle/ 1000 / Vm * *M* * 1000 / soil weight / SOC * 100 / days

The C mineralization rates were also translated to potential C mineralization rates (mg C m^{-2} day^{-1}) of the topsoil (Eq. 4), using the C mineralization in related to SOC (Eq. 3), SOC (%) and the bulk density (BD in g cm^{-3}). The potential C mineralization reflects the potential C release rate of the urban soils under laboratory conditions.

$Eq.4$:Potential C mineralization = C mineralization in relation to SOC * SOC / 100 / BD * 30/100 * 1000

2.4 Classification of soils and greenspaces

Following the definitions by De Bakker et al. (1989), four soil types were derived using the Dutch soil map (Supplementary Information: Fig. S1), assuming that soil types could be extrapolated over urban areas: (1) ‘Duinvaag’ soils: poorly developed sandy soils of which the sand particles are coated with iron. (2) ‘Vlakvaag’ soils: lightly coloured, humus-poor, poorly developed sandy soils. (3) Beekeerd’ soils: nutrient-rich humus layer on top of a nutrient-poor sandy layer. The soil is dominated by oxidation processes. (4) ‘Meerveen’ soils: mineral topsoil on top of a eutrophic peat layer.

Soils and sampling sites were further classified using eight criteria according to the framework of Aimone-Marson et al. (2016): (1) physical and chemical properties, (2) pollution (e.g. heavy metals, polycyclic aromatic hydrocarbons (PAHs), mineral oils, asbestos), (3) landscape metrics, (4) ownership, (5) aesthetical value, (6) specific ecological function, (7) social and (8) historical value.

Greenspaces were categorized according to their land use, vegetation class, litter management, greenspace management, ownership and size of greenspace and status of 'Ecozone' (Supplementary Information: Table S1) as follows:

- 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 vegetation was divided into three classes, namely trees (> 2.5 m), shrubs (> 1 m) and herbaceous vegetation (< 1 m). This division was consistent with the green 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. Privately owned greenspaces were not managed by the municipality. 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, litter management was 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 'no litter management'. The remaining greenspaces were assigned to the 'litter management' category.
- The status of 'Ecozone', green areas that are part of the ecological main structure of the Netherlands, was assessed using municipal maps (Den Haag Dataplatform 2020).

2.5 Statistical analyses

Collected data were analysed using the non-parametric Kruskal-Wallis and Wilcoxon rank sum test as the assumptions of the ANOVA and Student's *t*-test test were not met. Mean values are reported with their corresponding standard error (mean value ± standard error). Correlations were tested with Spearman's rank correlation coefficients, as the data were non-normally distributed. Relevant coefficients of the linear regression analysis are reported with their confidence interval (CI).

3 Results

The overview of each sample plot with their corresponding multi-faceted soil classification (Supplementary Information, Table S2) exposed that, in general, the urban soils were of a sandy texture with a wide range of pH values. For the sample plots where data was available, slight to moderate levels of contamination by heavy metals were reported. 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 located adject to roads and most private greenspaces were surrounded by buildings or closed fences. All greenspaces were valuable, either from the ecological, social, historical, or aesthetical perspective, applying the framework of Aimone-Marson et al. (2016).

3.1 Urban soil properties and their interrelationships

3.1.1 Soil properties

The mean SOC content 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 factor of almost 2 is in line with Pribyl (2010) who challenged the conventional factor of 1.724 and discovered that the estimate that 58% of SOM consists of SOC is too high.

The mean dry bulk density of the soil was 1.24 g cm^{-3} (± 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 large 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 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 μS cm^{-1} (± 4.30 μS cm^{-1}). The spatial trend roughly followed the spatial pattern of SOC (Fig. 2H).

Total nutrient levels (N, P, S) were determined to be 0.20% (± 0.03%) N, 0.06% (± 0.01%) P and 0.06% (± 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). Sample plots 10 and 21 formed clear exceptions with low C:N ratios of 9.33 and 9.07 respectively. 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). Only sample plots 5, 6, 7, 12 and 13 contained higher C:P ratios than SOM.

3.1.2 Interrelationships between soil properties

Several relevant interrelationships were detected between the measured soil properties (Table 2). 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. Moreover, a strong association between SOC and the water-holding capacity of the soil was detected. SOC levels were negatively correlated with the C mineralization expressed as the amount of C mineralized per SOC and positively with the potential C mineralization rate of the topsoil.

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. Additionally, the pH was negatively correlated to potential C mineralization. No significantly strong correlations between EC and other soil properties were detected.

Nitrogen levels strongly correlated with S levels, but not with P. Additionally, a strong association between N and the water-holding capacity of the soil was observed, which was not present for the other nutrients.

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

Table 2

Spearman correlation *r* of the soil C fractions, soil-quality characteristics and mineralization rates. Italic numbers indicate significant correlations ($p < 0.01$) a 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 condu P: phosphorus, S: sulphur, C:N ratio, C:P ratio, C:S ratio, WHC: water-holding capacity, BD: bulk density, Clay: clay fraction, C_{min}/SOC: C mineralization [mg C g C_{min}: potential C mineralization of the topsoil [mg C m^{-2} day^{-1} soil]).

	TC	TC_d	SOC	SIC	LOI	DOC	pH	EC	N	P	S	C:N	C:P	C:S	Ca-CO_3	WHC	BD	Clay
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
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*
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
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
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*
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*
pH							-	0.10	*-0.57*	-0.17	-0.29	-0.17	*-0.64*	*-0.45*	0.84	*-0.68*	*-0.65*	-0.18
EC								-	*0.44*	*0.48*	*0.49*	0.06	0.10	0.12	0.05	0.26	0.00	*0.32*
N									-	*0.60*	0.82	-0.12	*0.55*	0.17	*-0.56*	0.77	*-0.41*	0.44
P										-	*0.63*	-0.07	-0.11	-0.00	-0.21	0.47	0.06	0.40
S											-	0.03	0.45	-0.09	-0.24	*0.63*	*-0.30*	0.42
C:N												-	0.45	0.74	-0.10	0.11	*0.31*	-0.04
C:P													-	*0.52*	*-0.53*	*0.64*	-0.71	0.19
C:S														-	-0.43	0.28	*-0.38*	0.04
Ca-CO_3															-	*-0.64*	*-0.50*	-0.07
WHC																-	*-0.56*	0.43
BD																	-	-0.07
Clay																		-
C_{min}/SOC																		
Pot. C_{min}																		

3.2 C densities, C storage and degradability of urban SOM

The mean total C density was 110 t ha^{-1} (± 0.70 t ha^{-1}) of which 89% (± 0.36%) was contributed by SOC and 11% (± 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 soil C densities was observed in the 25 sample plots (Fig. 2C). The highest soil C 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.

3.2.1 Soil organic C densities in different types of greenspaces

Dividing the sample plots by vegetation class revealed that soils beneath shrubbery contained significantly higher SOC densities than those beneath trees or herbaceous vegetation (Fig. 3A). DOC values were significantly higher in soils beneath shrubbery and trees than herbaceous vegetation. Moreover, N, P and S densities were higher in soils beneath shrubs. However, P densities in soils beneath shrubs were only higher than soils beneath trees, and not than soils beneath grasses. For the soil types, soils classified as ‘Beekeerd’ soils held significantly higher SOC densities than those classified as ‘Meerveen’, ‘Vlakvaag’ and ‘Duinvaag’. SOC densities in ‘Meerveen’, ‘Vlakvaag’ and ‘Duinvaag’ soils did not significantly differ from each other (Fig. 3B). Additionally, ‘Beekeerd’ soils exposed higher EC values than ‘Vlakvaag’ and ‘Duinvaag’ soils.

Soils in greenspaces that were publicly owned showed similar SOC densities to the greenspaces that were privately owned (Fig. 3C). Although SIC densities were significantly higher in publicly than in privately owned greenspaces. Greenspaces located in the city centre contained higher SOC densities than those located in the suburban area (sample plots 1–11 and 12–25, Fig. 3D). Moreover, the more urbanized city centre contained higher EC values than the suburban district. Dividing greenspaces by land ownership, land use, greenspace size, litter management or status of ‘Ecozone’ did not result in significantly different SOC densities (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 and loss-on-ignition levels were detected in greenspaces where plant litter was relatively undisturbed. Additionally, pH values were lower in greenspaces where it was assumed the plant litter was left relatively undisturbed compared to greenspaces where plant

litter was regularly removed. Lastly, soils with undisturbed plant litter contained a higher water-holding capacity than those where litter was removed (Supplementary Information, Fig. S4).

3.2.2 Soil carbon storage

The sample transect crossed 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). Based on the green maps of the Netherlands, 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).

Based on the vegetation classes, it was estimated that the upper 30 cm of soil in The Hague that is covered by greenspaces has the potential to store 23.5 kt of C (± 0.79 kt C for grasses, ± 0.56 kt C for shrubs, ± 0.64 kt C for trees). The soil beneath shrubs proved to be hotspots for C accumulation as shrubs were responsible for more C storage than what corresponded to their surface area. This relation was reversed for soils beneath herbaceous vegetation and the tree cover and C stored beneath trees was approximately conformable (Fig. 4).

3.2.3 Degradability of urban soil organic matter

The mean cumulative mineralization of SOC in relation to SOC content was 17.9 mg C g SOC^{-1} (± 2.16 mg C g SOC^{-1}) and the mean potential C mineralization of the topsoil was 102 mg C m^{-2} (± 10.6 mg C m^{-2}) 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 mineralization rates (Fig. 4). The mean C mineralization rate in relation to SOC content was 0.43 mg C g SOC^{-1} (± 0.05 mg C g SOC^{-1} day^{-1}) and the mean potential C mineralization rate of the topsoil was 2.43 mg C m^{-2} (± 0.20 mg C m^{-2} day^{-1}).

C mineralization normalized to the SOC content of the soil was significantly higher in the dunes (sample points 24 and 25) than in the city centre, urban forest and suburban area. However, the potential C mineralization, expressed as the amount of C released from the upper 30 cm, was lowest in the dunes and highest in the urban forest. No difference was detected between the potential C mineralization of greenspaces located in the city centre or suburban area.

C mineralization normalized to SOC was independent of greenspace management, but the potential C mineralization was higher in soils where plant litter was relatively undisturbed than in soils where plant litter was removed regularly. C mineralization in relation to SOC was lower in soils beneath shrubs than soils beneath grasses, but not than trees. Moreover, the potential C mineralization of the topsoil was lower in soils beneath herbaceous vegetation than soils beneath trees and shrubs.

4 Discussion

This study aimed to quantify the soil C storage in The Hague in dependency of its land use, vegetation and soil type, land ownership, urbanization extent, management practises, greenspace size, and status of ‘Ecozone’ in order to better understand the C storage potential of The Hague urban soils.

Using the mean C densities of the vegetation classes led to a total soil C storage of 23.5 kt in the greenspaces of the studied districts of which 9.83 kt was stored beneath herbaceous vegetation (± 0.79 kt C), 5.58 kt was stored beneath shrubs (± 0.56 kt C) and 8.10 kt was stored beneath trees (± 0.64 kt C). The use of high spatial resolution GIS data at the scale of 10 x 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 greenspace 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 apparent in this case study, suggesting that processes driving soil C storage are controlled by different factors.

The total soil C storage of 23.5 kt in greenspaces of the studied districts is comparable to the 27 kt stored up to the same depth in the greenspaces of the city of Daejeon, South Korea (Yoon et al., 2016), a city 26 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 the studies differed significantly, making them not directly comparable as C concentrations vary non-linearly with depth (Renforth et al. 2011).

The mean SOC content of the sampled greenspaces (2.9%) exceeded the mean C content in Dutch grasslands, croplands, and nature for all soil types except for the peaty ‘Meerveen’ soils (Conijn & 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, croplands or upland forest soils. The lower SOC content 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).

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. However, 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. For urban greenspaces in The Hague, the data suggests that this assumption would result in an underestimation of current C stocks.

4.1 Soil C densities in the different types of greenspaces

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

Concerning the vegetation types, shrubs outperformed trees and herbaceous vegetation considering soil C 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 & Kleber 2015). One important abiotic factor may be the increased nutrient concentrations of soils beneath shrubs (N, P, S). Moreover, it appeared that the SOM in soils beneath shrubs was more stable against mineralization under laboratory conditions than SOM beneath grasses, but not than SOM beneath trees. However, the potential C mineralization of the topsoil beneath shrubs was higher than that of soil beneath grasses, which is related to the higher SOM and DOC content of soils beneath shrubs.

Greater C accumulation beneath shrubs is not in line with the pattern commonly observed in non-urban ecosystems where usually the soil C 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 urban soils of Helsinki, Finland. Although, Lindén et al. (2020) could not distinguish whether the different soil C stocks were the result of management or the vegetation itself. Edmondson et al. (2014) on the other hand hypothesized higher soil C densities beneath trees than grassland but detected no difference in land cover 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).

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 soil C densities than the less developed 'Vlakvaag' and 'Duinvaag' soils (poorly developed sandy soils). The hypothesis that the different soil types would result in different soil C densities proved partially correct, since only the 'Beekeerd' soil contained significantly higher soil C densities than the other soil types (Fig. 3B).

The rejected hypothesis of using soil maps to estimate soil C 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, 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 content in the upper 30 cm of the mineral layer of 'Meerveen' soil did not accurately reflect the C content 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.

Contrary to Edmondson et al. (2014), Rawlins et al. (2009) 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); A mean topsoil SOC density of 12.4 kg C m^{-2} (1.19 kg C m^{-2}) was measured in the city centre and a mean topsoil C density of 8.5 kg C m^{-2} (0.80 kg C m^{-2}) in the suburban area. 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 soil C densities than the suburbs (Cambou et al. 2018). For Berlin, higher soil C 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 soil C 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. Investigating the impact of urban greenspace management on SOC storage was complicated because urban greenspace management packages may have reverse effects on soil C 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 soil C (Lindén et al. 2020).

4.2 C:N:P:S ratios

Nutrient availability is critical for soil C sequestration (Kirkby et al. 2013). The urban soils of the Hague had mean C:N, C:P and C:S ratios of 15, 55 and 55 respecitively, which meant that that N and P may be limiting factors in C sequestration. 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 (Table 2). SOM 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 humification optimum implies that for each tonne of sequestered soil C, the soil approximately co-locks 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 microbes to the fresh organic inputs (e.g. co-metabolism, microbial mining, Kirkby et al. 2014). The C to nutrient ratio of the urban soils of The Hague implies that opportunities exist to improve C sequestration rates through increased input of N and P as fertilizers (Kirkby et al. 2013; Kirkby et al. 2016).

4.3 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 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 conditions are 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.

The amount of SOM results from the balance between the ability of decomposers to access SOM and the protection of SOM from decomposition by stabilization on organo-mineral associations (Lehmann & Kleber 2015). The balancing act between decomposition and protection from decomposition may be disturbed in urban soils by greenspace management, soil hydrophobicity, atmospheric deposition of pollutants and altered soil food web structures soil biota behaviour leading to either an accumulation or reduction of SOM stocks (Pavao-Zuckerman & Coleman 2005; Saviozzi et al. 2014; Vauramo & Setälä, 2011).

The degradability of SOM significantly correlated with the C and N contents of the soil (Table 2). 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 2), which also is in line with Zacháry et al. (2018) who state that the C:N ratio is likely a less good indicator for the recalcitrant C pools.

The effect of litter management on the potential C mineralization 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 potential C mineralization of 72 mg C m^{-2} (± 4.0 mg C m^{-2}) for the soils depleted from plant litter and 130 mg C m^{-2} (± 14 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. The greenspaces where plant litter was naturally augmented contained lower pH values, higher DOC values and higher water-holding capacities, which may all lead to increased SOC turnover (Table 2).

Mineralization rates of SOM differed along the transect (Fig. 5), which suggests different C availability for decomposer organisms during the incubation period (Lehman & 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 C mineralization, expressed as the amount of C released by the topsoil, the dunes emitted the lowest amount of C, as the initial SOC content of the dunes was very low (Fig. 2). The mean soil organic C content of the dune samples (0.35% SOC) 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 potential C mineralization rates were found in the urban forest, despite their lower C densities compared to the city centre and suburbs (Fig. 2C and Fig. 5). Soil properties that could explain the relatively high potential C mineralization in the urban forest are the low bulk density, higher water-holding capacity, lower pH and higher C:P ratio of the urban forest soil.

The bulk density in the forest topsoil was significantly lower than in the remaining samples (Fig. 2). The potential mineralization computation was composed of the bulk density, explaining partly why the potential mineralization would be higher in the urban forest despite a similar SOM degradability. The water-holding capacity of the soil was highest in the urban forest compared to the other greenspaces (Fig. 2) and the water-holding capacity was positively associated with the potential mineralization rate (r_s = 0.75).

Furthermore, the urban forest soil was strongly acidified with a mean pH of 5.2, with locally extremely acidic conditions of 3.8 (Fig. 2). 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 (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). In this study, pH inversely correlated with the potential C mineralization (r_s = -0.76), which is in line with Saviozzi et al. (2014) who incubated Italian urban soil.

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 potential C mineralization (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.

These findings are in line with Kim and Yoo (2020) who measured a lower respiration rate in the roadside tree system than in urban forests, although they measured respiration in the field using the chamber method, making the results not directly comparable. They 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.

5 Conclusion

This study drafted 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 proved 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 city centre and suburban district of The Hague had a soil C budget of 23.5 kt in the upper 30 cm of soil. Soils beneath shrubs proved to be hotspots for C storage as they only accounted for 13% of the land area 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 urban greenspace management 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 climate regulation. Further research on C dynamics is required to quantify urban soil C sequestration rates. 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 the greenspace (Cambou et al. 2018; Vasenev & 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 postulates 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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Conflicts of interest

The authors have no relevant financial or non-financial interest to disclose.

Availability of data and material

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Code availability

Not applicable.

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Figures

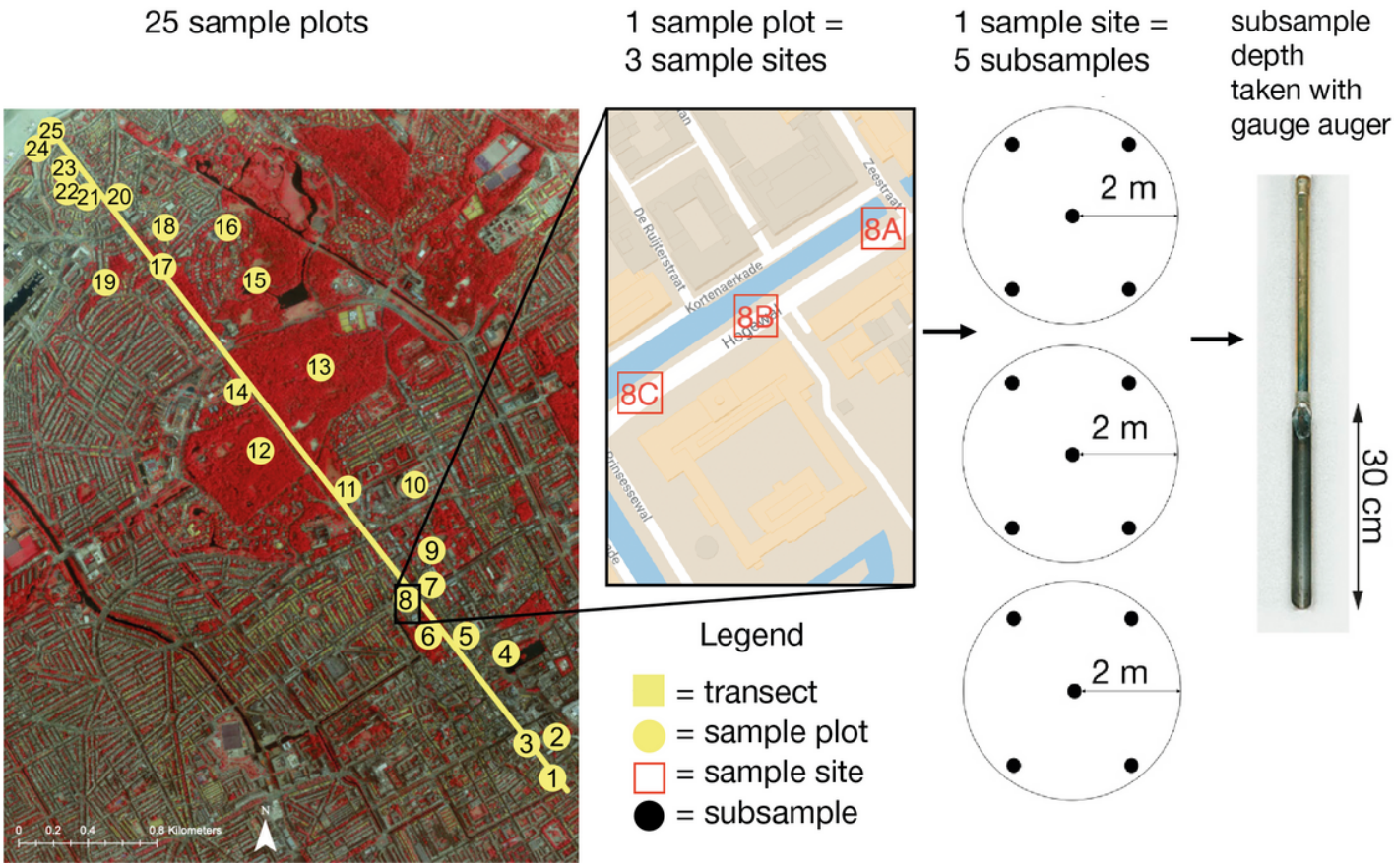

Figure 1

Schematic presentation of the sampling method. 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.

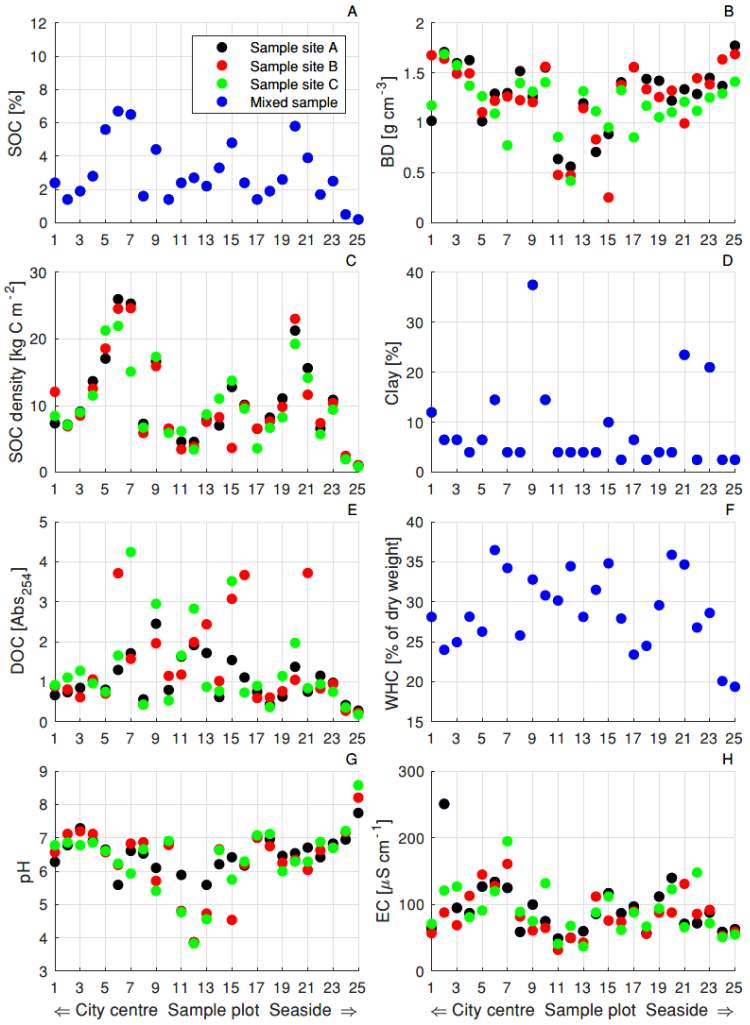

Figure 2

SOC content (A), bulk density (B), SOC density (C), clay content (D), DOC content (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).

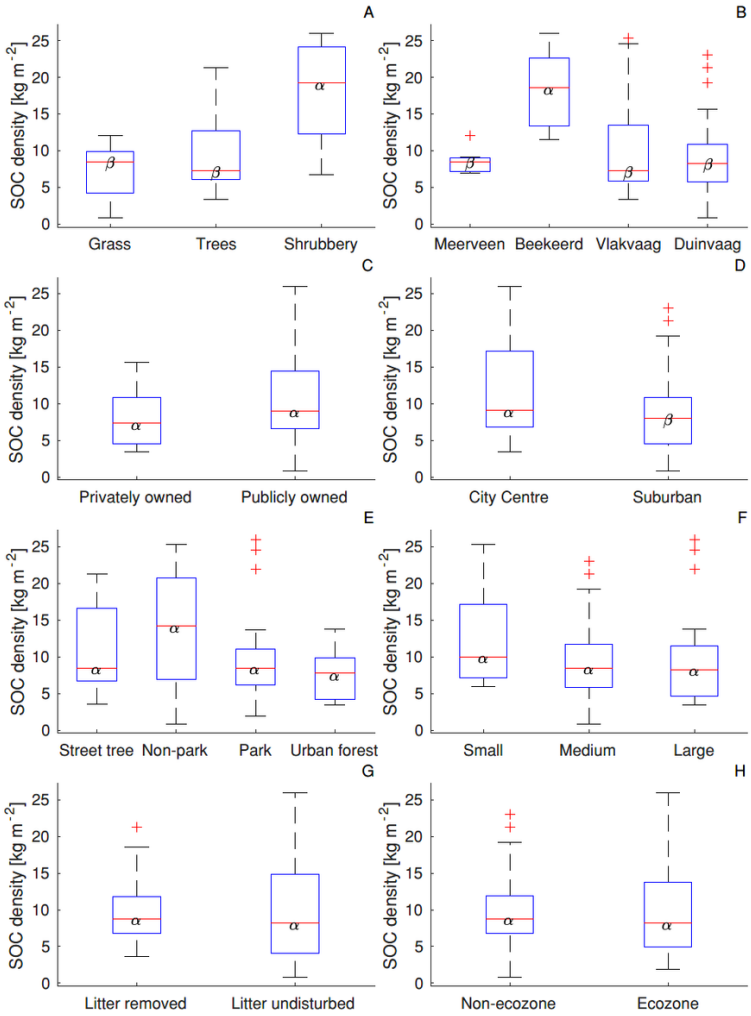

Figure 3

Variability of SOC densities in the upper 30 cm of soil of different 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.

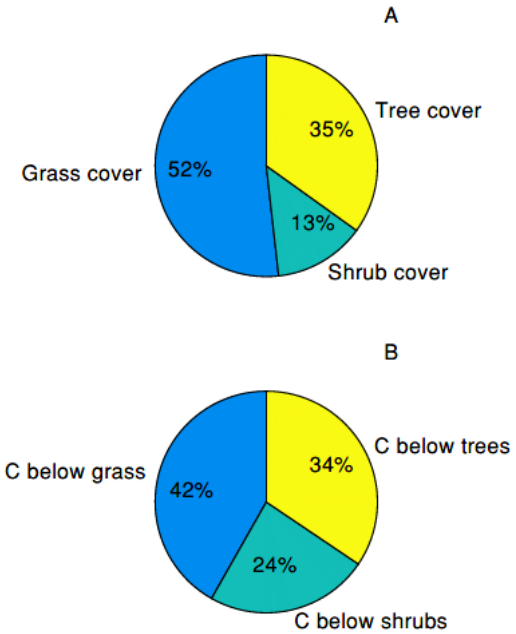

Figure 4

Distribution of vegetation cover (A) and estimated total soil C storage (B) in The Hague for the vegetation categories: grass, shrubs and trees.

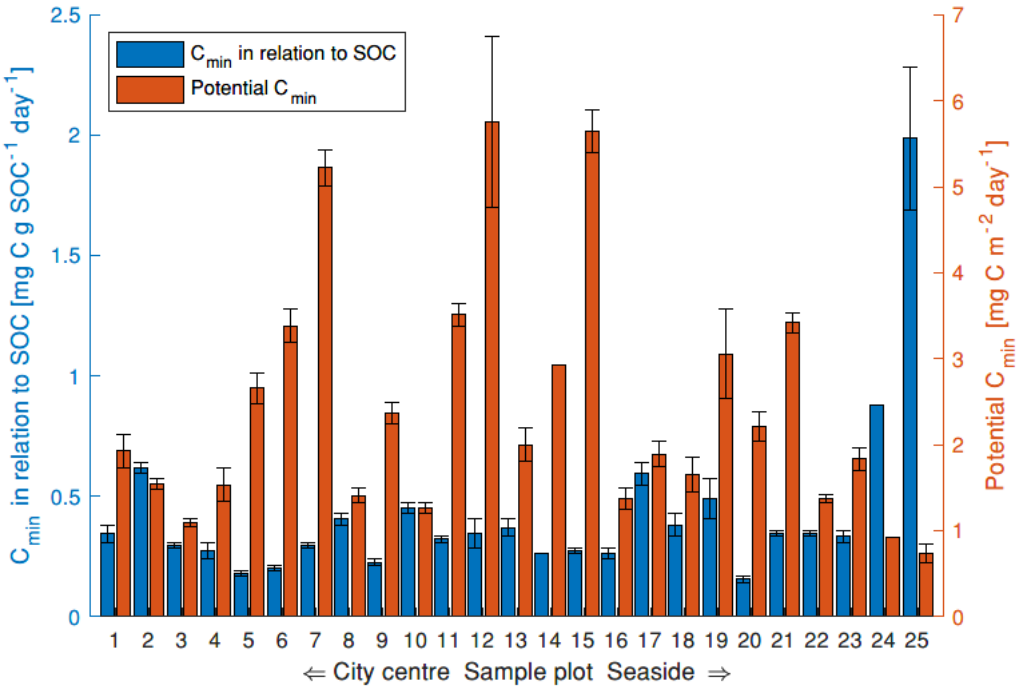

Figure 5

C mineralization rate in relation to SOC (blue) and potential C mineralization rate of the upper 30 cm of soil (red) under laboratory conditions (60% water holding capacity, 20 °C) in 25 urban soil samples.

Supplementary Files

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