Urban Forestry & Urban Greening 49 (2020) 126633

Contents lists available at ScienceDirect

Urban Forestry & Urban Greening

journal homepage: www.elsevier.com/locate/ufug

Quantifying carbon stocks in urban parks under cold climate conditions

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ARTICLE INFO

Handling Editor: Wendy Chen

Keywords:
CO_2
Ecosystem services
Hemi-boreal climate
Park management
Urban green space
Urban soils

ABSTRACT

Removing CO_2 from the atmosphere and storing carbon in vegetation and soil are important ecosystem services provided by urban green space. However, knowledge on the capacity of trees and soils to store carbon in urban parks - especially in the northern latitudes - is scarce. We assessed the amount of organic carbon stored in trees and soil of constructed urban parks under cold climatic conditions in Finland. More specifically, we investigated the effects of management, vegetation type and time since construction on the amount of carbon stored in park trees and soil. We conducted two tree surveys and collected soil samples (0–90 cm) in constructed parks managed by the city of Helsinki. The estimated overall carbon density was approximately 130 t per park hectare, when the carbon stock of trees was 22 to 28 t ha^{-1} and that of soil 104 t ha^{-1} at the very least. The soil to tree carbon storage ratio varied from 7.1 to 7.5 for vegetated, pervious grounds and from 3.7 to 5.0 for entire park areas. The effects of park management and vegetation type could not be entirely separated in our data, but time was shown to have a distinct, positive effect on tree and soil carbon stocks. The results indicate that park soils can hold remarkable carbon stocks in a cold climate. It also seems that park soil carbon holding capacity largely exceeds that of forested soils in Finland. Preservation and augmentation of carbon stocks in urban parks implies avoidance of drastic tree and soil renovation measures.

1. Introduction

Urban green space can be treated as part of green infrastructure, defined in the EU policy as “a strategically planned network of natural and semi-natural areas with other environmental features designed and managed to deliver a wide range of ecosystem services” (European Commission, 2013). Protection and enhancement of ecosystem services, such as water purification, noise reduction, habitat provision and recreational benefits, is regarded as particularly important in urban areas, where most people now live (European Commission, 2013). Improving urban green infrastructure can at the same time diminish the ecological footprints of cities and improve the quality of life for the city dwellers (Gómez-Baggethun and Barton, 2013). Accordingly, information based on mapping and assessment of urban ecosystems and their services is essential for planning and decision making in cities.

Carbon (C) storage and sequestration is one obvious benefit linked to green infrastructure. Many urban C stock studies have quantified the amount of aboveground C stored by greenspace in cities of the temperate climate zone (Davies et al., 2011; Hutyra et al., 2011; Liu and Li, 2012; Strohbach and Haase, 2012; Schreyer et al., 2014; Timilsina et al., 2017; Zhang et al., 2015). The results indicate that the two dominant factors governing the amount of aboveground C in urban greenspace are tree density and size (Nowak and Crane, 2002; Davies et al., 2013). Although methodological dissimilarities make direct comparisons difficult, it is apparent that there are remarkable differences in C densities (C t ha^{-1}) between cities, due to the prevailing climate, history and pattern of urbanisation, and the species composition and age structure of urban forests. To our knowledge, no reports on the aboveground C stores held by greenspace in boreal cities are thus far available.

On a global scale, soils are the third largest C pool, after the oceanic and geological pools (Lal, 2008). Soil organic C content depends primarily on the magnitude of organic matter input and heterotrophic respiration, which is determined by soil temperature, moisture, nutrient and oxygen content and factors such as the quality of decomposing litter (Berg, 2000; Williams and Rice, 2007) and the structure of the decomposer community (García-Palacios et al., 2016). In urban environments, anthropogenic factors such as recurrent land-use changes, land development, soil sealing and introducing exotic vegetation modify the stock, input and mineralisation of soil organic matter.

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https://doi.org/10.1016/j.ufug.2020.126633
Received 25 April 2019; Received in revised form 4 February 2020; Accepted 6 February 2020
Available online 07 February 2020

Accordingly, urban soil organic C content is highly variable and can be lower or higher than that of rural or forest soils (Pouyat et al., 2006; Lorenz and Lal, 2009; Edmondson et al., 2012; Vasenev et al., 2013; Liu et al., 2016).

Within the urban fabric, parks and residential soils often display a high soil C content, which is assumed to be due to the prevailing vegetation, frequent management inputs and lack of annual disturbance (Pouyat et al., 2006; Edmondson et al., 2014a). The importance of management practices has not yet been extensively investigated, but inputs of water, nutrients and organic residues are thought to influence urban soil CO_2 flux (Decina et al., 2016; Trammell et al., 2017) and thus may also influence soil C stocks. Campbell et al. (2014) demonstrated a positive correlation between fertilization frequency and topsoil C content in residential lawns, whereas addition or removal of grass clippings had no effect on soil C stock. Another recent study on residential lawns could not prove fertilization, irrigation or mulching effects on soil C accumulation (Huyler et al., 2014). Increasing lawn mowing frequency from one to eight times per season and leaving clippings on the ground can raise soil C storage, due to increased aboveground net primary production (Poeplau et al., 2016). Apparently, predicting soil C response to management practices is complicated by the many interacting variables that may be involved.

Vegetation effect on urban soil C pool is often intertwined with management inputs. Edmondson et al. (2014a) found greater differences in soil C storage between woody and herbaceous vegetation in domestic than in non-domestic greenspace, probably caused by different management practices in the two greenspace types. Park soils under trees may show higher or similar C stocks than soils under urban grassland, because different tree species affect C accumulation and distribution within the soil profile differently (Edmondson et al., 2014b; Bae and Ryu, 2015; Setälä et al., 2016). On the other hand, in Auckland, New Zealand, soil organic C stocks were higher in grass-dominated parkland soils than in urban forest soils, largely due to higher bulk densities in parklands (Weissert et al., 2016).

Besides the effects of organic matter input and decomposition, time plays a definite role in urban soil C dynamics. Urban land-use changes such as park construction often involve topsoil removal, soil relocation, surface grading and compaction. Soil disturbance affects C sequestration and release, and it takes a long time until a new equilibrium C content is eventually gained (Guo and Gifford, 2002). In newly established turfgrass systems, a relatively steady state in soil C pools is reached after about 30–50 years (Qian and Follett, 2002; Bandaranayake et al., 2003; Pouyat et al., 2009). In residential yards, street tree plantings and parks, time was identified as the most significant factor affecting urban soil physical, chemical and biological properties (Scharenbroch et al., 2005). Old sites (mean 64 years since disturbance) displayed distinct reductions in soil bulk density, increases in microbial biomass and activity, and increases in organic matter as compared to newer sites (mean 9 years) (Scharenbroch et al., 2005).

So far, few studies have quantified the net C storage of urban ecosystems, i.e. C fixed both above and below the ground (Edmondson et al., 2012; Dorendorf et al., 2015; Liu et al., 2016), and to our knowledge, no such assessment has been done in Nordic countries. In this paper, we report estimates on the amount of C stored in park trees and soil in the city of Helsinki, Finland.

The study was limited to constructed urban parks owned by the city. The hypotheses were: (1) tree and soil C stock is the higher the more intensively a green area is managed, (2) vegetation type affects soil C concentration, and (3) in old parks both tree and soil C stock is higher than in newly constructed green areas. The results are discussed in relation to the management of urban parks.

2. Materials and methods

The city of Helsinki, with 217 km^2 land area and 643,000 inhabitants (Mäki and Vuori, 2018), is located on the south coast of Finland (60°10′15″N, 24°56′15″E) at the northern limit of the hemiboreal region. Nearly 60 % of the land area in Helsinki is vegetated (including all tree-covered areas, fields, yards and other low-vegetation surfaces), about 20 % is used for buildings and about 20 % for roads and other traffic purposes (Helsingin ympäristön tila, 2017). Actual nature areas (forests and parks) account for 36 % of the land area in Helsinki and there is 120 m^2 of recreation area per inhabitant (Helsingin ympäristön tila, 2017). The average annual temperature and the average annual precipitation is 5.3 °C and 682 mm, respectively (Pirinen et al., 2012).

2.1. Tree surveys

To estimate tree C stocks, measurements were taken in constructed municipal parks of Helsinki in 2011 and in 2013. A stratified random sampling was applied during both sampling times, but with differing stratification criteria. Information on the parks and their locations within the city is provided in the Supplementary Fig. 1 and in the Supplementary data.

In 2011, park size was used as the stratification criterion to avoid the more common small parks being overrepresented. On grounds of a datasheet provided by the Urban Environment Division of the City of Helsinki, the total number of constructed park areas managed by the municipality was 807 in 2011. The parks were divided into three groups: small, medium and large (< 2.3 ha, 2.3 − 8 ha and > 8 ha in total area, respectively). Of the 711 small parks, 36 were randomly chosen with 2 sampling plots each, from the 88 medium-sized parks, 8 were chosen with 5 sampling plots each, and from the 8 large parks, 4 were sampled with 10 plots each. This resulted in 152 sample plots altogether, each with an area of 201 m^2 (radius 8 m). Map coordinates for the centre point of each plot were drawn randomly and marked on the map. In October 2011, the diameter at breast height (DBH) and species (or if unidentified, genus) of all trees within every sample plot were recorded. All in all, 466 individual trees and 33 tree taxa were registered.

The effects of park age and management level on tree C stock were assessed by sampling a second set of trees in 2013. In Finland, most municipal parks are managed following the Green Area Maintenance Classification with three maintenance classes for constructed parks (Nuotio, 2014). In this scheme, the most valuable, centrally situated green areas are managed in accordance with the class A1 that implies regular fertilization, irrigation and removing plant residues such as lawn clippings, senesced leaves in the fall and pruning offcuts from the site. Areas in class A2 include standard parks, play and sports grounds, courtyards of public property and other grounds managed primarily regarding their usage and functionality. The lowest maintenance class A3 is applied on e.g. buffer belts between built and natural areas and does not involve removing plant biomass or adding resource inputs.

Table 1
Lawn management standards for the three maintenance classes specified in the Finnish Green Area Maintenance Classification (Nuotio, 2014).

Maintenance class	Lawn height range	Grass clippings	Irrigation	Fertilization
A1	4–7 cm	removed	yes	annual
A2	4–12 cm	removed if unsightly	if clearly needed	annual
A3	4–25 cm	usually left on site	no	if clearly needed

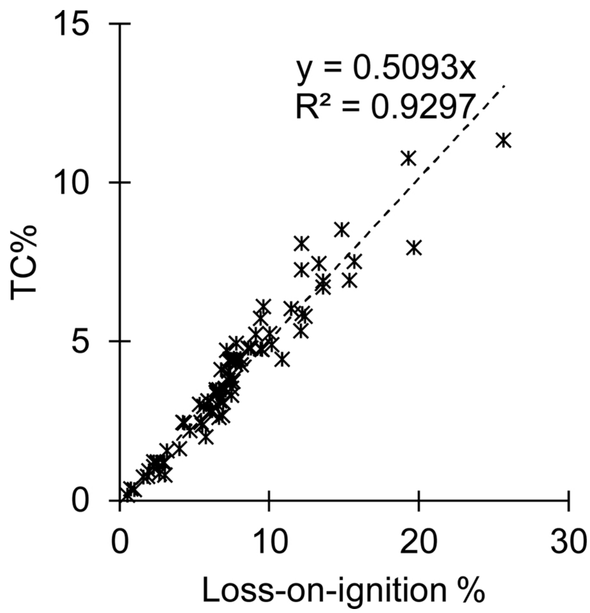

Fig. 1. The relationship between total carbon content (TC % w/w) and loss-on-ignition (LOI %) in 80 randomly chosen park soil samples collected in 2013.

Table 1 illustrates differences in lawn management practices between the three maintenance classes. Park areas in the City of Helsinki were first graded according to the degree of management in the early 1980's. The early four-grade in-house classification was replaced by the Green Area Maintenance Classification in about 1998, after which the practical working instructions have been refined several times, lastly in 2014 (Nuotio, 2014). It should be noted that within one park, there often are areas assigned to two or three different maintenance classes.

Consequently, the 2013 sampling was stratified on the level of park management specified as maintenance class A1, A2 or A3 (Nuotio, 2014). First, each of those 19 municipal parks representing all three maintenance classes within their boundaries, were selected. From amongst these, the 4 parks holding grave sites were left out, resulting in a subgroup of 15 parks with three maintenance classes. A second subgroup of 15 parks was randomly selected from amongst those 298 parks including areas assigned to two different maintenance classes (A2 and A3).

The resulting 30 parks served as the venue for collecting both the second tree sample and the soil samples of this study (see subsection 2.2. Soil survey). Within each park, two or three tree sampling plots were established round the randomly drawn soil sampling sites. The size of the tree plots (79 in all) and the data recorded for them were identical with those applied in 2011. Altogether 184 trees, representing 25 taxa, were registered.

In both tree surveys, DBH was measured for the three largest stems of multi-stem trees, and trees less than 2.5 cm in DBH were left out. The species group classification and relevant equations for total aboveground biomass presented in Jenkins et al. (2003) were then applied to the data, and any taxa encountered that was not included in the original grouping, were classified according to their genus. A biomass to C conversion constant of 0.5 was used when estimating tree C stock (Nowak and Crane, 2002).

2.2. Soil survey

To assess the soil C stocks of urban parks, samples were collected from the 30 municipal parks stratified by maintenance class designations as explained in subsection 2.1. Tree surveys. The actual soil sampling sites within each park were chosen by drawing three random map coordinates, at least 3 m away from any known high-voltage cables. The sampling sites were marked on field maps. The 30 parks were grouped into 5 age classes on grounds of time since park construction or time since the latest large renovation, acquired from the archives and personnel of the City of Helsinki.

In October 2013, each of the 90 random sites in the 30 parks were sampled for soil C content. To study the effects of park management and vegetation, additional soil samples were collected so that all maintenance classes (A1, A2, A3) and vegetation types (lawn, shrubbery and herbaceous perennials) present in each park were sampled. Shrubbery was defined as an area thickly planted with multi-stemmed, small- or medium-sized woody perennials. The additional soil samples were collected in the same way as the three random samples, from appropriate maintenance class and vegetation type areas located nearest to the random sites. Each sampling site was at least 50 cm away from any adjacent hardscape feature and different vegetation type. The presence or absence of organic mulch was recorded for each shrubbery. One soil C sample was composed of at least three soil auger cores (diam. 18 mm). The cores were separated to three depth layers, 0 − 30 cm, 31 − 60 cm and 61 − 90 cm.

To estimate soil C density per unit area (kg m^{-2}), 1–4 bulk density (BD) samples were collected in each park by taking undisturbed soil cores from the topmost soil layer (0 − 6 cm) with a metal tube (inside diameter 57 mm, length 59 mm). Soil BD was sampled at the same sites as soil C content.

The altogether 470 soil samples from 247 sites (247 0 − 30 cm C samples, 143 31 − 60 cm C samples, 27 61 − 90 cm C samples and 53 topsoil BD samples) were dried at 105°C until constant weight and weighed. Samples aimed for determination of soil C content were ground lightly to break any clay aggregates and passed through a 2-mm sieve. The organic matter content was determined by loss-on-ignition (LOI, 2 h at 550°C). The LOI values were calculated per total sample weight assuming no LOI in the fraction larger than 2 mm.

A sub-set of samples (n = 80) was randomly chosen for determination of total C (TC) content using the vario Max CN Element Analyzer (Analysensysteme GmbH, Hanau, Germany). Finnish soils generally have low inorganic C content, so the TC measured was assumed to consist entirely of organic C. Linear correlation analysis between TC content and LOI revealed a strong relationship (R^2 = 0.93, Fig. 1), so the coefficient estimated (0.509) was used to convert all LOI values into TC contents.

2.3. Estimation of C density

Two estimates for mean tree C density (t ha^{-1}) were computed, utilizing the results of the two tree surveys. For the first, park size-specific dataset, tree C density was initially estimated by proportioning the average C stock of the sample plots to the total area of the park concerned. The resulting C densities were used to estimate the mean tree C density of the small, medium-sized and large parks, separately. As the relative frequency of each park size group was known, an estimate for the average C density of trees in all parks of the city could be calculated. For the second tree data, sampled on management regime, the C density of sample plots was first averaged for each maintenance class, then for the total area of constructed municipal parks in Helsinki using the actual share of park area assigned for each maintenance class.

Soil BD values were applied for estimation of TC content per volume for each soil sample. The mean soil C density (kg m^{-2}) was then calculated for each vegetation type in each separate maintenance class. Maintenance class specific soil C densities were calculated by using the area of different vegetation types and sealed surface in relation to the total area of each maintenance class (Helsingin kaupunkiympäristön toimiala, 2014, see subsection 3.3. C density of parks). The overall average C density for park soils in Helsinki was computed using the relative share of park area designated to each maintenance class.

2.4. Statistical analyses

To test the influence of park management, vegetation type, mulching and park age on soil TC content, the non-parametric Kruskal-Wallis test was applied, as the soil data did not conform to ANOVA test assumptions even after transformations. The non-parametric analyses were conducted for each sampling depth separately using SPSS Statistics 25. The effects of maintenance class and vegetation type on soil BD were examined using one-way ANOVA in SAS 9.4 proc GLM.

Pearson *r* correlation was applied to assess the relationship between soil LOI and TC content and to study the association between soil BD and TC content. Pearson *r* correlation was also used to examine the relationship between tree C density and park age in the 2013 tree survey. In the two latter analyses, the TC and C density values were log-transformed to fulfil the normality assumption. Correlation analyses were performed by SAS 9.4 proc REG. Estimates for tree and soil C stocks and soil BD are presented as mean values ± standard deviation (SD).

3. Results

3.1. Tree C stock

The first tree survey gave an average aboveground C storage estimate of 28.1 t C ha^{-1} for constructed parks in Helsinki. In small, medium-sized and large parks, the mean C densities were 22.6 (± 24.7) t C ha^{-1}, 36.5 (± 30.2) t C ha^{-1} and 23.9 (± 11.6) t C ha^{-1}, respectively. Mean tree densities were 147 (± 167) trees ha^{-1} for small parks, 144 (± 62) trees ha^{-1} for medium-sized parks and 172 (± 166) trees ha^{-1} for large parks.

Small trees (DBH 2.5–14.9 cm) dominated the data with a relative frequency of 47 %, yet their contribution to the total C storage was only 3 % (Fig. 2). Two thirds of the total estimated C stock were associated with middle-sized trees (DBH 30–60 cm) (Fig. 2). The average DBH of all trees was 18.8 cm. *Betula* spp., *Sorbus aucuparia* and *Acer platanoides* were the three most common tree taxa. Nearly half (48 %) of the estimated biomass C was in *Tilia* × *vulgaris* and *Betula* spp. trees, indicating that large trees often represented these taxa.

The second tree survey resulted in a roughly similar overall tree C stock estimate (22.1 t C ha^{-1}) as the first, despite differences in sampling. The level of management did not affect ($p > 0.6$) tree C stock in our study (24.1 ± 22.3, 22.1 ± 20.0 and 20.1 ± 18.1 t C ha^{-1} for maintenance classes A1, A2 and A3, respectively). The mean tree density was 105 (± 67) for park areas in maintenance class A1, 116 (± 76) for A2 areas, and 126 (± 85) for areas in the A3 category.

In the second tree survey, the frequency of small trees (DBH 2.5–14.9 cm) was 28 %. The overall average DBH was 24.8 cm. *Acer*

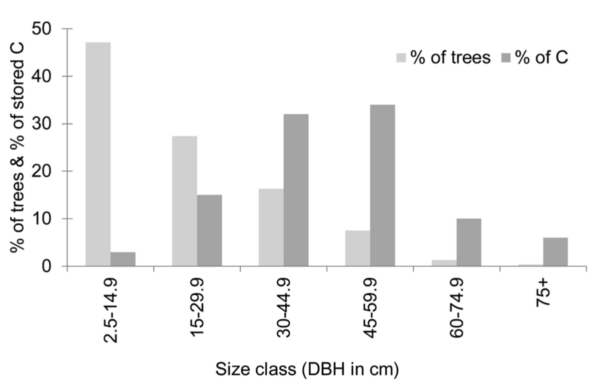

Fig. 2. Frequency of tree size-classes and their contribution to the total carbon storage of park trees surveyed in 2011. Light columns represent the size (DBH in cm) distribution of the trees and dark columns denote the relative share of carbon stored in each tree size-class.

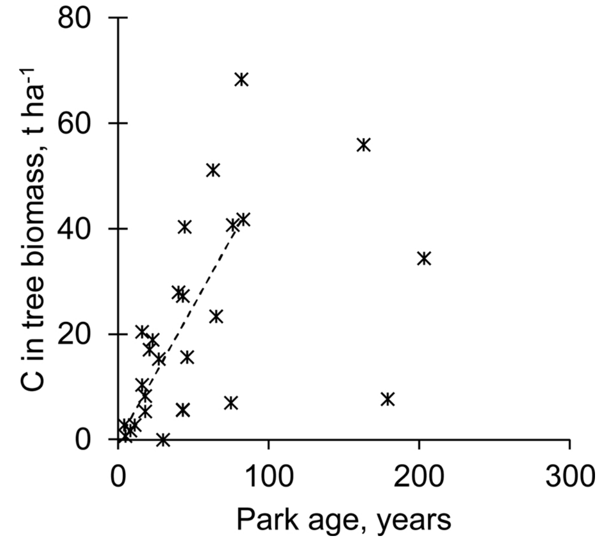

Fig. 3. The relationship between tree C stock and park age for the 30 urban parks sampled in 2013. Correlation between the log-transformed biomass C density and park age was weak, but significant (R^2 = 0.24, p = 0.01; leaving the three oldest parks out of the analysis resulted in R^2 = 0.52, $p < 0.001$). The regression line between soil C (t ha^{-1}) and park age represents parks aged 4–90 years, excluding the three oldest parks.

spp., *Betula* spp. and *Pinus* spp. were the most common tree genera and they contained the majority of tree C stock. Park age (time since initial construction) was clearly related to tree C density up to the park age of approx. 100 years (Fig. 3).

3.2. Soil C stock

The TC content was on average 4.45 ± 2.03 % in the top soil (0 − 30 cm), 3.63 ± 2.18 % in the middle layer (31 − 60 cm) and 2.88 ± 2.52 % in the deepest layer (61 − 90 cm). The intensity of park management influenced soil C stock only slightly. When analysing all vegetation types together, a weak effect was found in the middle soil layer (0 − 30 cm: p = 0.111; 31–60 cm: p = 0.047, 61–90 cm: p = 0.409), for which the mean TC content was somewhat higher in maintenance class A1 (TC = 3.99 ± 1.65 %) than in areas maintained according to class A3 (TC = 3.39 ± 2.71 %). When examining the three vegetation types separately (Table 2), the effect of management was detectable (p = 0.025) only beneath herbaceous perennials in the same middlemost (31–60 cm) soil layer (TC = 4.39 ± 0.99 % for maintenance class A1, TC = 2.88 ± 1.17 % for class A2).

Vegetation influenced TC stock in the topmost (p = 0.000) and middle (p = 0.013) soil layers, when data for the three maintenance classes was pooled. At both depths, the average C stock was higher beneath shrubs (0–30 cm TC = 5.08 ± 2.16 %; 31–60 cm TC = 4.07 ± 1.88 %) than under lawn (0–30 cm TC = 3.92 ± 1.70 %; 31–60 cm TC = 3.38 ± 2.42 %). Topsoil C content was also higher for perennial plantings (0–30 cm TC = 4.88 ± 2.37 %) than for lawn, yet this effect could not be shown in lower sampling layers.

When considering vegetation effects separately for each maintenance class, shrubberies proved to hold higher soil TC stock than lawn in maintenance classes A2 and A3 (Fig. 4). For class A2 areas, the difference proved significant at all sampling depths (Table 2; 0 − 30 cm p = 0.000; 31–60 cm p = 0.010; 61–90 cm p = 0.041). For class A3, only the topsoil held higher TC stock (p = 0.026) beneath shrubs than under lawn. In the intensely managed park areas (class A1), soil TC content did not differ between shrubberies and lawn (Fig. 4).

One third (33 %) of all shrubberies sampled for soil C were covered with organic mulch while the rest of shrub plantings (67 %) were unmulched. Mulching increased topsoil (0 − 30 cm) TC content (p =

Table 2
Soil organic C content (% w/w, mean ± SD) under lawn, shrubberies and herbaceous perennials, managed following maintenance classes A1, A2 and A3. Empty cells denote missing data.

Vegetation type	Sampling depth and maintenance class								
	0–30 cm			31–60 cm			61–90 cm		
	A1	A2	A3	A1	A2	A3	A1	A2	A3
Lawn	4.57 ± 2.36	3.73 ± 1.43	3.85 ± 1.60	4.09 ± 2.23	3.22 ± 2.15	3.16 ± 3.10	1.80 ± 1.51	2.31 ± 2.05	2.08 ± 1.19
Shrubbery	4.93 ± 1.99	5.41 ± 2.54	4.77 ± 1.67	3.44 ± 1.08	4.63 ± 2.33	3.87 ± 1.66	–	7.43 ± 4.57	3.08 ± 0.06
Herbaceous perennials	5.26 ± 2.55	3.84 ± 1.37	3.13 (single observation)	4.39 ± 0.99	2.88 ± 1.17	–	1.85 (single observation)	3.75 ± 0.37	–

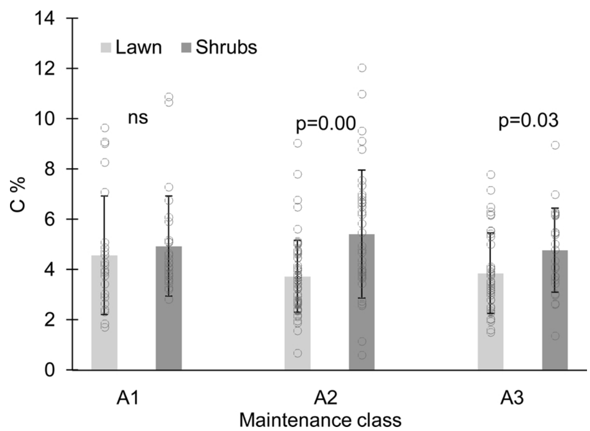

Fig. 4. The comparison of soil TC content (average ± SD, % w/w) at 0 − 30 cm depth under lawn and shrubberies in park areas maintained in accordance with classes A1, A2 and A3. For lawns, n = 24, 65, and 40, and for shrubberies n = 27, 37, and 26, in maintenance classes A1, A2 and A3, respectively.

0.035) from 4.8 ± 2.1% to 5.6 ± 2.2 % on average across maintenance classes. Deeper in the soil the effect of mulch was not detectable.

Park age, defined as years since construction or last major renovation, influenced soil TC content noticeably (p = 0.000, Fig. 5). Parks built or last renovated in 1850–1950 and in 1950–1970, had higher soil TC content than newer parks, built or renovated since 1970. The effect was evident in all soil layers (Fig. 5).

Topsoil BD was negatively related to soil TC content (R^2 = 0.59, Fig. 6). Management level did not affect soil BD (p = 0.39). The average soil BD in the examined parks was significantly (p = 0.00) higher under lawn (0.93 ± 0.18 g cm^{-3}) than under shrubs (0.73 ± 0.23 g cm^{-3}).

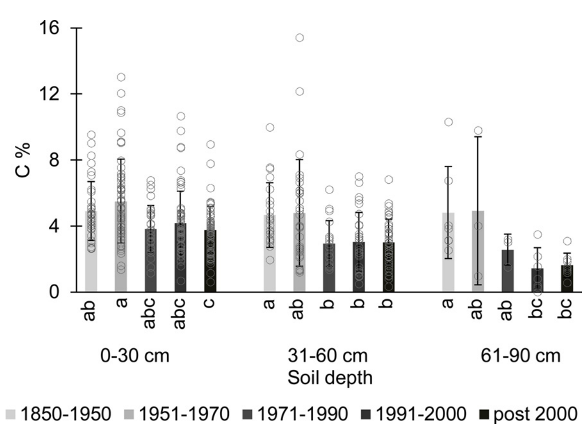

Fig. 5. Soil total carbon content (TC % w/w) in the different age classes of parks. Error bars represent ± SD and letters indicate significant differences in TC between park age classes for each soil layer separately. For the soil layer 0 − 30 cm, n = 29–63 per age class, for the soil layer 31 − 60 cm n ranges from 18 to 45, and for the soil layer 61 − 90 cm, n varies from 3 to 8 per age class.

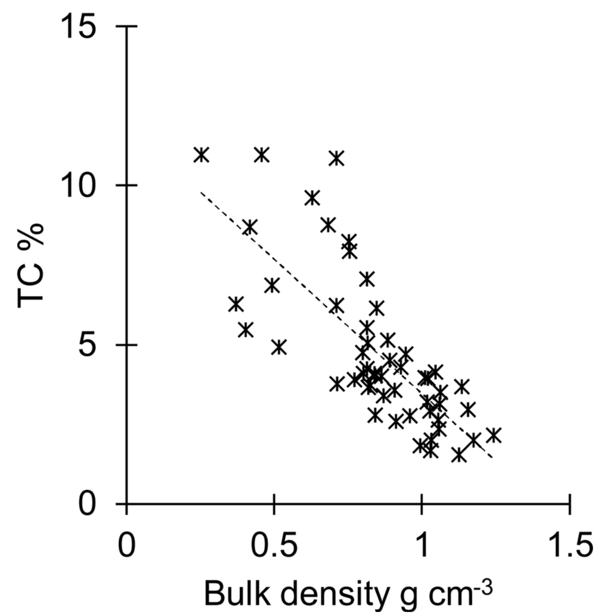

Fig. 6. Topsoil total carbon content (TC % w/w) and bulk density were negatively correlated in the examined parks (TC % log-transformed for analysis; adj. R^2 = 0.59, p < 0.00).

3.3. C density of parks

In park soil under lawn, C density was 19.5, 16.5 and 14.2 kg m^{-2} for park areas maintained in accordance with class A1, A2 and A3, respectively. The corresponding values beneath shrubs were 14.0, 15.0 and 14.2 kg C m^{-2}. For perennial beds, which are mostly maintained according to class A1, soil C density was on average 14.8 kg m^{-2}.

When calculating overall soil C density values for differently managed park areas, the average share of different land-cover types was considered for each maintenance class (Fig. 7). Assuming negligible C beneath impervious surfaces (footpaths, pavements, parking lots, buildings etc.) and for areas classified as “other” (mainly water), yielded soil C density estimates of 89, 107 and 101 t C ha^{-1} for maintenance classes A1, A2 and A3, respectively, with an overall average of 104 t C ha^{-1} for park soils in Helsinki. Quite obviously, the relatively large share of impervious surfaces (Fig. 7) had a substantial impact on soil C storage estimates. Leaving non-vegetated sealed surfaces and water areas aside brought about soil C values of 180, 164 and 142 t C ha^{-1} for maintenance classes A1, A2 and A3, respectively, and a general mean of 155 t C ha^{-1} for park soils beneath vegetation.

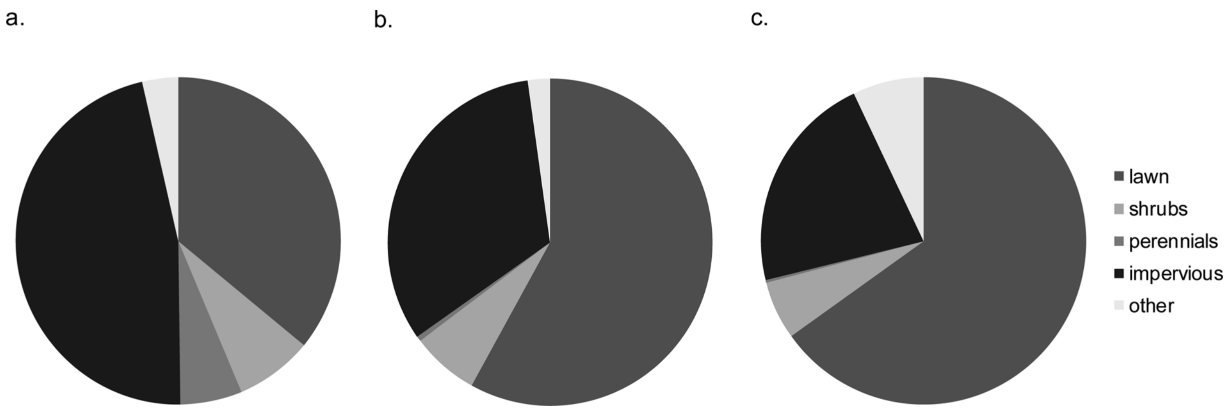

Fig. 7. The proportion of lawn, shrubberies, perennials, impervious surfaces and other, usually water-covered areas in the municipal parks of the city of Helsinki in 2014. The share of each land-cover type is shown separately for areas maintained following maintenance class A1 (a), A2 (b), and A3 (c) (Helsingin kaupunkiympäristön toimiala, 2014).

By adding up the tree ($22-28$ t C ha^{-1}) and soil (104 t C ha^{-1}) carbon storage estimates we attained an average C density value around 130 t C ha^{-1} for constructed urban parks of Helsinki. The soil to tree C stock ratio was 7.5, 7.4 and 7.1 for vegetated park areas maintained in accordance with class A1, A2 and A3, respectively. When soils beneath impervious surfaces, assumed to hold no C, were accounted for, the ratio decreased to 3.7, 4.8 and 5.0 for the three respective maintenance classes.

4. Discussion

4.1. Tree C stocks

The tree C density was 22.1 and 28.1 t C ha^{-1} in the two separate tree surveys we conducted in the public, constructed parks of Helsinki. The C stock estimates are in line with the results of similar studies made elsewhere. In north-eastern China, the average aboveground C density of urban parks was 33.7 t C ha^{-1} in Shenyang (Liu and Li, 2012) and 54.1 t C ha^{-1} in Changchun (Zhang et al., 2015). In Leipzig, Germany, the estimates for green urban areas and cemeteries were 29.4 and 27.8 t C ha^{-1}, respectively (Strohbach and Haase, 2012). City-wise comparisons should be made with caution though, as the sampling methods, land-use classifications and especially the biomass equations applied in each study vary.

We estimated the aboveground biomass of park trees by the generalized equations originally developed for trees grown in forests of the United States (Jenkins et al., 2003). Application of general models may increase the uncertainty of C storage estimates as compared to local equations (Timilsina et al., 2017). However, the sole available allometric equations based on Nordic tree data involve only the four dominant forest tree species (Repola, 2008; Repola and Ulvcrona, 2014), and the Nordic equations require tree height information, not measured in our study. McHale et al. (2009) compared the predictions of allometric equations from traditional forests to urban-based tree biomass predictions. The results revealed that the potential error depends on the species being evaluated and may result both in over- and in underestimation of urban tree biomass so that estimates for tree populations composed of diverse species and various sizes can be more accurate than estimates for a particular species (McHale et al., 2009). The conclusion was that generalized equations, such as those used in our study, may be the best practice when handling highly variable tree data (McHale et al., 2009).

It should be kept in mind, that we surveyed only trees in constructed municipal parks, the total area of which was 900 ha, whereas woodlands maintained by the city cover 4680 ha and the whole area of publicly managed green space is approximately 7000 ha (Helsingin kaupunkiympäristön toimiala, 2014). The tree density in the sampled parks was ca. 150 trees per hectare, whereas woodlands maintained by the city have much higher tree densities, similar to those managed for commercial forestry (i.e. depending on species, site productivity and forest age, in the range of ca. 500–2000 trees per hectare). Increasing tree density in constructed parks could improve urban C balance, but at the same time a high tree density might interfere with some of the numerous goals and service expectations set for urban parks. Open urban green areas are also needed for e.g. sports activities, public events, and even for urban biodiversity (Brunbjerg et al., 2018).

A city-wide estimation of C pools held by the green infrastructure in Helsinki remains undone, but obviously, trees in constructed parks contribute to only a minor part of the total C storage. Moreover, when assessing the C storage services provided by urban trees on a city-wide scale, areas of mixed or private ownership should also be included (Davies et al., 2011; McPherson et al., 2013), as well as contributions from soil respiration and emissions associated with greenery management (McPherson et al., 2015; Tidåker et al., 2017; Velasco et al., 2016).

Most park trees inspected in our study in Helsinki were rather small. In the first sample, the mean DBH was 18.8 cm while almost half (47 %) of the measured trees had a DBH $<$ 15 cm and stored less than 40 kg C per tree. So, a major part of the aboveground C stock was held by large, old trees, similarly as in many previous studies (e.g. Davies et al., 2011; Horn et al., 2015; Zhang et al., 2015). Ensuring a long, productive life span for trees is essential when seeking to increase the C pool of urban vegetation (Nowak et al., 2002), but it is also important to have young and growing trees to replace the old, dying cohort of trees.

We hypothesized that intensive park management and increasing years since construction will raise the amount of C stored in park trees. The results did not support the management practice effect, but the relationship between park age and tree C pool was positive until the age of one hundred years, after which the relationship was diminished. It seems reasonable that the size of trees and the amount of C stored in them will increase with site age, but this is not always the case in urban landscapes. Trammell et al. (2017) identified human interventions as the main reason for the discrepancy between residential yard age and CENTURY modelled tree C stock in Baltimore. Replacement of old trees by young ones may have caused the disruption in the growing C stock trend observed in our data. It seems also that our random sampling plots often fell on wide lawn areas between large trees or avenues, typical for the three oldest parks.

4.2. Soil C stocks

Our results for park soil C concentration in Helsinki (on average 4.07 % at $0-90$ cm depth) were comparable to those reported by Setälä et al. (2016) for urban parks in Helsinki and Lahti, Finland (3.75 % and

3.25 % at 0 − 50 cm beneath evergreen and deciduous trees, respectively). Previous studies on urban parklands have shown widely varying C levels from 0.6 to 11.1%, depending on climatic region, soil type, sampling depth, vegetation and land-use history (Schleuß et al., 1998; Vasenev et al., 2013; Edmondson et al., 2014a; Weissert et al., 2016). Our data revealed substantial amounts of carbon down to 90 cm depth, in agreement with earlier results on subsoil layers in urban parks (Pouyat et al., 2006; Edmondson et al., 2014b; Bae and Ryu, 2015).

The average C density of park soils in our data (10.4 kg m^{-2} for all park soils, 15.5 kg m^{-2} for soils beneath vegetation) clearly exceeds the mean C content in Finnish croplands (4.1–6.7 kg m^{-2} at 0 − 15 cm; Heikkinen et al., 2013), and the C pool of upland forest soils in Finland (approx. 6.3 kg m^{-2} at 0 − 100 cm; Liski et al., 2006). The common practice of not considering subsoil C stocks in agriculture makes the comparison of urban and forest soil C to croplands difficult, however.

Our result is about the size of the mean C density reported by Edmondson et al. (2012) for non-residential greenspace soils (13.2 kg m^{-2} at 0 − 100 cm) in Leicester, UK. Further studies on park soil profiles to a depth of 1 m have indicated C density values from 4.2 kg m^{-2} in Hong Kong to 9.9 kg m^{-2} in Baltimore, USA (Pouyat et al., 2006), and from 3.4 kg m^{-2} for lawn soil to 14.0 kg m^{-2} for wetland in Seoul Forest Park, Republic of Korea (Bae and Ryu, 2015). In the cool northern climate, park soils may store 22.0–35.5 kg C m^{-2} at 0 − 50 cm, as estimated on grounds of a park soil survey conducted in the cities of Helsinki and Lahti by Setälä et al. (2016).

Compared to previous studies in warmer climates, the soil to tree C storage ratio was high varying from 7.1 to 7.5 for vegetated, pervious grounds and from 3.7 to 5.0 for entire park areas in Helsinki. It is hard to find relevant comparison points for these figures, as both soil and tree C stores are quantified in few studies so far. The estimates available range from a soil to tree C ratio of 1.2 in a residential area in Chuncheon, Korea (Jo, 2002) to the ratio of 14.7 calculated for urbanized areas in Wyoming, USA (Pouyat et al., 2006). The overall ratio for the city of Hamburg, Germany, was rated as 2.3 or 3.1 depending on how much C is assumed to be held beneath impervious surfaces (Dorendorf et al., 2015). The estimated mean ratio was 2.8 for cities in the USA (Pouyat et al., 2006), and 4.6 for Leicester, UK (Edmondson et al., 2012). In Finnish forests growing on mineral soils, the ratio is ca. 1.5 (calculated from data presented in Liski et al., 2006).

In the current study, soil C stocks under various impervious surfaces in parks, such as parking lots, paths, buildings, were not assessed. We assume that there is virtually no C beneath sealed surfaces, because due to regular soil freezing in winter, all hardscapes in Finnish parks (paths, pavements, buildings) are built on a 40 − 100 cm thick gravel layer, the depth of soil excavation depending on prevailing load bearing requirements and subsoil quality. Indeed, a pilot-level study conducted in the city of Lahti, southern Finland, showed that the average soil C content 0 − 10 cm beneath asphalt is 0.14 (n = 7) (H. Setälä, unpublished results). In areas of a milder climate, however, soils beneath impervious surfaces may store significant amounts of C (Edmondson et al., 2012; Raciti et al., 2012; Piotrowska-Długosz and Przemysław, 2015).

4.3. Management effects on soil C stocks

The hypothesized positive effect of management on soil C level was noticeable only in the depth of 30 − 60 cm under perennial plantings, while no management effect was found under lawn or shrubs. The somewhat higher C concentration in perennial beds of class A1 than in beds maintained in accordance with class A2 may rather be caused by random plant species differences in the sampled plots than by actual management. The number of different perennial plant taxa was 16 for the A1 plots sampled (n = 35) and 5 for the A2 plots (n = 14). Half of the A2 plots was planted with low-growing, shallow-rooting groundcover perennials.

All in all, our results did not support the hypothesis that intensive management is reflected in the C pools of park vegetation and soil. We used the maintenance class designation of each sample plot (A1, A2, A3) as an indication of management level. However, work specifications for each maintenance class include practices which may have reverse effects on soil C pool, as for example, the maintenance standard for A1 lawns that incorporates irrigation and fertilization, but also removal of grass clippings (Nuotio, 2014). Similarly, shrubs receive fertilizers more often in maintenance class A1 than in A2, yet A1 shrubberies are also kept tidy by frequent litter removal, whereas soils beneath A2 shrubberies are commonly mulched and seldom raked (Nuotio, 2014).

Generally, management regimes such as fertilization or adding organic amendments are assumed to cause the increased C accumulation often observed in urban soils (Edmondson et al., 2014a; Bae and Ryu, 2015; Decina et al., 2016; Trammell et al., 2017). Still, fertilization, irrigation and removal of mown clippings did not impact on soil C levels measured beneath residential lawns in Alabama, USA (Huyler et al., 2014), nor did removal of autumn leaves seem to affect soil C storage in Leicester, UK (Edmondson et al., 2014b). Additional factors, such as timing of fertilization, frequency of grass cutting and differences between plant taxa can intervene in soil C changes (Edmondson et al., 2014b; Huyler et al., 2014; Poeplau et al., 2016; Setälä et al., 2016), making predictions of soil C responses difficult.

4.4. Vegetation effects on soil C stocks

Our second hypothesis on the effects of vegetation on soil C accumulation was valid for maintenance classes A2 and A3 where C stocks were larger under shrubs than beneath lawn. No differences between planting types were observed in the most intensely maintained A1 areas. We assume that soil C increment was largely due to mulching with pine bark, more common in shrubberies sampled from A2 and A3 designated areas than in shrub plantings of maintenance class A1. Accordingly, the impacts of vegetation itself and management measures could not be reliably distinguished in our data.

Edmondson et al. (2014a) report a similar mixed effect of vegetation and management when reporting soil organic C results in Leicester, UK. In their city-wide study, soil C density was not affected by land-cover in non-domestic greenspaces, yet in domestic gardens soil C concentration was larger under trees and shrubs than under herbaceous vegetation. This was supposed to be a consequence of adding organic fertilizers and mulches to trees and shrubs, commonly practised in home gardens.

Setälä et al. (2016) surveyed park soils in Helsinki and Lahti, Finland, finding higher C concentrations under evergreen trees than in park soil beneath lawn. Even other soil properties (pH, organic matter and nitrogen content) were modified by vegetation type, and these changes were generally enlarged as parks aged. It was reasoned that the effects can result from differences in plant litter, growth strategies, or both (Setälä et al., 2016). More empirical research is needed to better distinguish the major effects of management and vegetation.

4.5. Time effects on soil C stocks

In agreement with our third hypothesis, the data revealed a distinct, positive time effect on park soil C concentration across all sampling depths (0 − 90 cm). A similar increment in soil C level over time was found in residential yards by Huyler et al. (2014), but only at 0 − 15 cm depth. Campbell et al. (2014) detected a positive relationship between C concentration and time in the topmost 0 − 5 cm of residential lawn soil, but a negative relationship at 20 − 30 cm. In our data, soil C level stabilized roughly 50 years after construction disturbance, i.e. after park establishment or renovation, similarly as in the previous Finnish park soil survey (Setälä et al., 2016). Likewise, soil organic C accumulation seems to asymptote between 30 and 50 years post disturbance in turf and grassland systems (Conant et al., 2001; Pouyat et al., 2009).

In our data, the number of parks established before 1950 was 11 out of which 8 parks had undergone major renovation. While analysing the time effect we first indicated park age as years since initial construction and found no significant relationship between park age and soil C pool (data not shown). Only after replacing the construction year with the year of last major renovation, the positive time effect became evident. Park renovation usually involves construction measures such as topsoil replacement or profile rebuilding. These common urban land development practices may lead to soil C loss, as shown by Chen et al. (2013). Since parks are currently committed to provision of ecosystem services, park management should be continuous but cautious to keep the site attractive and usable and on the other hand, to minimize all kinds of plant and soil disturbance and C release by management activities.

5. Conclusions

We surveyed the amount of C stored by trees and soil of urban parks in the city of Helsinki, Finland. The results show the importance of park soils, capable of holding manifold C pools in comparison to trees. Both vegetation type and management practices may affect park C budget in many intertwined ways hard to discern, except for the impact of such an obvious C input as bark mulch.

In our study, time was the main driver for soil C accumulation. It seems that in the cool climate of Helsinki, park soil layers down to at least 90 cm accumulate carbon for ca. 50 years after park establishment or other major disturbance that takes place due to management or renovation. To protect existing C stocks in parks, any unnecessary disturbance to trees and soil should be avoided, while at the same time parks should be given enough care to avoid the need for drastic renovation activities.

CRediT authorship contribution statement

Leena Lindén: Conceptualization, Methodology, Supervision, Project administration, Funding acquisition, Data curation, Formal analysis, Software, Writing - original draft, Writing - review & editing. **Anu Riikonen:** Conceptualization, Methodology, Supervision, Project administration, Funding acquisition, Investigation, Data curation, Formal analysis, Software, Visualization, Writing - original draft, Writing - review & editing. **Heikki Setälä:** Conceptualization, Methodology, Supervision, Project administration, Funding acquisition, Writing - original draft, Writing - review & editing. **Vesa Yli-Pelkonen:** Conceptualization, Methodology, Supervision, Project administration, Funding acquisition, Writing - original draft, Writing - review & editing.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Acknowledgements

This work was supported by the Finnish Cultural Foundation; the Helsinki Metropolitan Region Urban Research Program; and the Urban Environment Division of the City of Helsinki. The authors acknowledge M.Sc. Susanna Turunen for provision and analysis of tree data and the two anonymous reviewers for their constructive feedback on our work.

Appendix A. Supplementary data

Supplementary material related to this article can be found, in the online version, at doi:https://doi.org/10.1016/j.ufug.2020.126633.

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