

1 High carbon losses from established growing sites delay the carbon  
2 sequestration benefits of street tree plantings - a case study in Helsinki,  
3 Finland

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11  
12 **Keywords**

13 carbon sequestration

14 tree soil carbon

15 tree biomass equations

16 urban trees

17  
18  
19 List of abbreviations

20 AB aboveground biomass (stem, branches and leaves)

21 ABW aboveground woody biomass (stem and branches)

22 BE biomass equation (see Appendix)

23 C carbon

24 DBH diameter at breast height

25 LOI loss-on-ignition

26  $O_e$  loss-on-ignition estimated (Eq.6) for the periods between LOI samplings

27 SOM soil organic matter

28 SW soil water

29  $T_f$  soil temperature, as measured at tree sites

30 WB woody biomass

31  $W_f$  soil water content, % of weight, as measured at tree sites

32  $\alpha$  LOI and SWC response parameter in the incubation model

33  $\beta$  temperature response parameter in the incubation model

35 **Abstract**

36 We assessed the net carbon (C) sequestration dynamics of street tree plantings based on 10 years of  
37 measurements at two case study sites each with different tree species in Helsinki, Finland. We assessed C  
38 loss from tree soils and tree C accumulation, tested the applicability of pre-existing growth and biomass  
39 equations against observations, and estimated the time point for the beginning of net C sequestration for the  
40 studied street tree plantings. The tree woody biomass C accumulation in the first 10 years after planting was  
41 18--32 kg per tree. At the same time the C loss from the growth media was at least 170 kg per growth media  
42 volume (25 m<sup>3</sup>) per tree. If this soil C loss was accounted for, the net C sequestration would begin, at best,  
43 approximately 30 years after planting. Biomass equations developed for traditional forests predicted more  
44 stem biomass and less leaf and branch biomass than measured for the species examined, but total  
45 aboveground biomass was generally well predicted.  
46

47 **Introduction**

48 Carbon (C) sequestration is one of the ecosystem services that encourage the planting of urban trees  
49 (McPherson *et al.*, 2005). The C sequestration effects of urban tree plantings consists of C stock change in  
50 above- and belowground tree compartments, and soil organic matter (SOM) stock changes related to tree  
51 planting and litter production by planted trees. The belowground C stock of urban trees and soils is not well  
52 known, but there are indications that urban soil C stocks can be substantial (Pataki *et al.*, 2006). In the  
53 traditional, non-urban forests of Scandinavia, the soil C stock occurs predominantly in the superficial layers  
54 and is as large as or larger than that of the vegetation (Liski *et al.*, 2006). In urban greening, trees are planted  
55 traditionally in limited container-like soil spaces or wider structural soil (e.g. Grabosky and Bassuk 1995,  
56 Neal and Whitlow 1997, Kristoffersen 1999) volumes in which the load-bearing properties of the soil have  
57 been enhanced with stony matrices. Fine soil, suitable for tree rooting, is located in the voids of the stone  
58 matrix. In both of these methods, artificial growing media brings C-rich soil into the deep layers. Currently,  
59 Finnish municipalities use SOM contents of 10--12% (measured as loss-on-ignition, LOI) throughout the  
60 standard 1-m-deep growth media in tree plantings (Rakennustietosäätiö, 2010). A square metre of new  
61 traditional tree growth media thus typically has a C stock of approximately 40--50 kg C m<sup>-2</sup> and a structural  
62 soil of 10--20 kg C m<sup>-2</sup>; about 2--10 times more than in traditional upland forest soils in Finland (Liski *et al.*,  
63 2006).  
64

65 In contrast to natural SOM, which has substantial proportions of slowly decomposing fractions, the artificial  
66 growth media organic matter may decompose quickly and lose C to the atmosphere (Bernal *et al.*, 1998).  
67 Soil sealing (by e.g. asphalt or pavement), common in urban environments and predominantly used in  
68 combination with structural tree soils, impairs soil heat and soil water (SW) exchange (Scalenghe and  
69 Marsan, 2009) and limits the C input from above the ground, affecting biomass accumulation and  
70 decomposition. These effects may lead to overall C loss from street tree plantings unless the C sequestered

71 by the tree exceeds the C loss from the growing media. The organic matter in the growing media may be  
72 derived from peat, or partially or entirely from renewable C sources, such as compost. In addition to its use  
73 in growing media as such, peat is a common additive used when composting sewage sludge, kitchen and  
74 food waste etc. (e.g. Himanen and Hänninen 2011), and consequently, also the SOM in compost-derived  
75 growing media tends to contain some fraction of peat. While renewable C originating from short-term  
76 biogenic cycle is commonly considered neutral in regards to climate change, peat-derived CO<sub>2</sub> in energy  
77 production is currently viewed as equivalent to fossil fuel emissions (IPCC 2006). IPCC takes no stand on  
78 other uses of peat however; thus the official C accounting status of peat in growing media is somewhat  
79 unclear. There is a general interest in finding replacements for peat in the growing media industry however,  
80 due to both wetland protection and CO<sub>2</sub> emissions (e.g. Defra 2009).

81  
82 In a street tree planting, growing media C emissions can be compensated and exceeded by the C  
83 sequestration of trees over time. Unfortunately, the size and C stock of urban trees of a given age are not  
84 easily predicted (Peper *et al.*, 2014). Currently, aboveground C stocks in urban trees are estimated with  
85 allometric tree biomass equations (BEs) developed in traditional forests, if urban-based equations are not  
86 available. Root biomass may then be estimated from a set root-shoot ratio despite its large variation between  
87 ecosystems and species (e.g. IPCC 2006). However, the accuracy of traditional forest based BEs in an urban  
88 context has been questioned (McHale *et al.*, 2009). The particular above- and belowground environments of  
89 trees influence both the overall growth rate and biomass distribution within trees (Litton *et al.*, 2007; Zhou *et al.*,  
90 2014). Consequently, the urban environment may lead to biomass distributions different from those  
91 observed in traditional forests, with consequent biomass estimation problems. In addition to biomass  
92 distribution, the tree-related C inputs into urban soil remain largely unknown. Root exudates and litter likely  
93 contribute to soil C stock while, especially in paved areas, the aboveground litter might not, because it is  
94 either removed or moves along, unable to enter the soil under the pavement.

95  
96 The purpose of this study is to estimate the long term carbon dynamics of a street tree planting in the hemi-  
97 boreal city of Helsinki. We collected data from two case study street tree plantings (established according to  
98 current establishment practices in Finland) about long term C stock changes in the growth media and trees.  
99 These were combined with literature based tree growth equations that we tested with separate tree data from  
100 different aged street tree plantings in Helsinki, and literature based biomass equations. At the case study  
101 sites, we assessed the soil C stock changes occurring during the first 10 years after planting, using a LOI  
102 change-based approach. We evaluated the estimate with CO<sub>2</sub> production of soil samples in an incubation  
103 experiment. At these sites we estimated the tree biomass accumulation from direct measurements of the case  
104 study trees. The measurements were compared against literature-based biomass equations to find the  
105 equations corresponding best to the case study observations.

106

107 We used the data to estimate the time needed for street plantings to reach the C compensation point (the  
108 number of years after planting required for the tree C capture to reach the sum of the soil C loss in the first  
109 decade after planting) in Helsinki. Our hypothesis was that the amount of C lost from the growth media of  
110 the case study trees would offset the C uptake of the tree growth during our study period, and an  
111 improvement in average street tree life expectancy would be needed to obtain C sequestration benefits with  
112 current planting practices.

## 114 **Materials and methods**

### 115 *Case study sites*

116 We studied the tree biomass and soil C changes on two separate street sites, located 800 meters apart from  
117 each other, in the Viikki suburb in Helsinki, Finland, (N60°15', E25°03') over 10 years after the  
118 establishment of the street. One street, 250 m in length, was planted with 15 common lime *Tilia × vulgaris*  
119 Hayne trees (Tilia site) and the other (200 m in length) with 22 black alder *Alnus glutinosa* (L.) Gaertn. f.  
120 *pyramidalis* Dippel 'Sakari' trees (Alnus site). The sites were established in the summer of 2002 and the  
121 trees were planted in the autumn of 2002. At both sites, three different premixed structural soil mixes were  
122 applied as a 1-m-deep, 3-m-wide continuous strip (Tilia site) or 15–20-m-long planting pockets for two to  
123 four trees (Alnus site). Cast-iron tree gratings 2.25 m<sup>2</sup> in size were used around the trees, and the streets  
124 outside the grates were paved over with sand-jointed block pavers. The available soil volume per tree was  
125 45--50 m<sup>3</sup> at the Tilia site and 15--30 m<sup>3</sup> at the Alnus site.

127 The structural soils consisted approximately 2/3 by volume of stones ranging from 30 to 120 mm in size and  
128 1/3 of fine soil. In soil mix 1, the fine soil was mainly sand, clay and the SOM source was peat. In soil 2, the  
129 fine soil was derived from composted sewage sludge mixed with peat, sand and pine bark, but the  
130 contribution of peat to the final soil mix SOM could not be determined. In soil 3, the components were fine  
131 gravel, sand, clay and leaf compost (peat was not used in the composting process). For soil 1, the initial LOI  
132 was 6% and for soil 2 20%, according to their respective manufacturers. For soil 3, the initial LOI was 4.4%,  
133 based on the composition and properties of the materials used (7% by volume of leaf compost, 20% of clay  
134 with LOI 8.3% and 3% bark mulch). Soils 1 and 2 were commercial mixes, while soil 3 was specially mixed  
135 for the study sites.

137 The transplanted *Tilia* trees were 8--11 cm in diameter-at-breast height (DBH) and *Alnus* 7--11 cm,  
138 respectively, and both were balled and burlapped. After transplanting, the trees were not pruned (except for  
139 dead and broken branches and shoots growing from rootstocks) until late 2008. Thereafter, the *Tilia* were  
140 pruned about annually to achieve the necessary crown lifting. The *Alnus* trees were not crown-lifted, and  
141 only branches that were damaged or leaned far out from the columnar crown shape were removed.

142

143 *Soil water content and temperature measurements*

144 Each soil mix on both streets was instrumented during the establishment with continuously measuring soil  
145 moisture sensors (Delta T MLx2 (Delta-T Devices Ltd, Burwell, Cambridge, UK), see Riikonen *et al.*, 2011  
146 for installation details) at depths of 10 and 30 cm and with temperature sensors (resistor type KTY81) at  
147 depths of 10, 30 and 60 cm from the surface of the growth media.

148  
149 Data loggers (DP-158; Envic Oy, Turku, Finland) read the temperature and SW sensors from July 2003  
150 onwards at 1–30-min intervals. At the Alnus site, soil moisture sensors at 30 cm depth were installed in 2005  
151 and data from 10 cm depth was used before that. Soil moisture and soil temperature ( $T_f$ ) were averaged to 30-  
152 min means, and missing data, due mainly to periods of datalogger malfunctions, were gap-filled with linear  
153 interpolation. The SW content could only be reliably measured while the  $T_f$  was  $> 0$  °C; periods when ground  
154 was frozen were filtered out and gap-filled linearly (Kornelsen *et al.* 2012) (most missing SW values in  
155 2003: 52%, least missing values in 2006; 7%). The measured volumetric water content was transformed to  
156 percentage of soil weight ( $W_f$ ).

157

158 *Soil sampling and analysis*

159 Soil samples were collected in the autumn 2005, 2008 and 2011 from two pits in each soil mix and site  
160 (2x3x2 sampling pits each year, each located in separate planting pocket at the Alnus site, and at least 10 m  
161 apart at the Tilia site). The average distance from the nearest tree was 2–3 m, depending on parked cars and  
162 other practical considerations. The pavement was removed and a pit with a diameter of 30–50 cm was dug  
163 with hand trowels. In 2005, the pits were dug down to depths of 30 cm (sampling depth 0–30 cm, altogether  
164 12 samples), in 2008 to at least 60 cm and to 90 cm where possible (sampling depths 0–30 cm (n=12), 30–  
165 60 cm (n=12) and 60–90 cm (n=11), altogether 35 samples), and in 2011 to 60 cm (sampling depths 0–30  
166 cm (n=12) and 30–60 cm (n=12), altogether 24 samples). The rocks ( $\geq 30$  mm) in the soil mix were  
167 separated from the fine soil. All the fine soil excavated from each sampling pit was weighed, thoroughly  
168 mixed and a sample of approximately 3 liters was taken from each depth. The sampling pit volume up to  
169 each depth was estimated by measuring the volume of vermiculite required to fill the pit. After each  
170 measurement the pit was vacuumed empty of any vermiculite and filled back with the original soil material  
171 once all measurements were completed.

172

173 The soil samples were divided in parts for further analysis. One subsample of approximately 400 g was dried  
174 at 105 °C and measured for loose dry bulk density, LOI (550 °C, 2 h) and particle-size distribution (dry  
175 sieving and laser diffractometer Coulter LS230; Beckman Coulter Inc., Krefeld, Germany). Another separate  
176 subsample was reserved for incubation to estimate the soil CO<sub>2</sub> production rate. The subsamples for  
177 incubation were stored at 10 °C for 0–30 h before initiation of the incubation experiment.

178

179 *Soil incubation*

180 In 2005 and 2011, from all 12 and 24 soil samples collected, respectively, subsamples were incubated and  
181 measured for CO<sub>2</sub> production. In 2008, subsamples were collected and incubated from all sites at depths of  
182 0-30 cm and a random sample of one-half of the sites at 30-60 and 60-90 cm. All visible roots were picked  
183 out from the soil samples, and eight 20-ml portions from each sample were placed in 120-ml incubation  
184 bottles and weighed. The bottles were then flushed with compressed air at atmospheric CO<sub>2</sub> concentration  
185 and sealed. The bottles were immediately placed randomly at 5, 10, 15 and 25 °C, two bottles at each  
186 temperature. For each soil sample, two bottles were filled with air only and their CO<sub>2</sub> concentration was used  
187 as a background level in calculating the results.

188  
189 After 24 h, the bottles were placed in an ice bucket and gas samples were taken and analysed immediately  
190 for CO<sub>2</sub> content with gas chromatography. CO<sub>2</sub> production during the incubation time ( $R$ , l CO<sub>2</sub> l<sup>-1</sup> h<sup>-1</sup>) was  
191 calculated for each subsample, based on the CO<sub>2</sub> concentration in the incubation bottles and measured by gas  
192 chromatography (Hewlett-Packard Co., Palo Alto, CA, USA; Jaakkola and Simojoki, 1998), as:

$$193 \quad R = (CO_{2\ inc} - CO_{2\ ref}) / h \quad \text{Eq. 1}$$

194 where CO<sub>2,inc</sub> (l CO<sub>2</sub> l<sup>-1</sup>) is the CO<sub>2</sub> content of the incubation bottle after incubation, CO<sub>2,ref</sub> (l CO<sub>2</sub> l<sup>-1</sup>) is the  
195 CO<sub>2</sub> content of the air used in filling the bottles prior to incubation and  $h$  is the incubation time in hours.  
196 Then the C loss (g CO<sub>2</sub> g<sup>-1</sup> h<sup>-1</sup>) was calculated as:

$$197 \quad (R V_b / M_s) v_m M_{CO_2} \quad \text{Eq. 2}$$

198 where  $R$  is the CO<sub>2</sub> produced in the incubation bottle per hour of incubation (l CO<sub>2</sub> l<sup>-1</sup>),  $V_b$  is the volume of  
199 the incubation bottle (l),  $M_s$  is the weight of the incubated sample (g),  $v_m$  is the gas constant (l mol<sup>-1</sup>),  $M_{CO_2}$   
200 is the molar mass of CO<sub>2</sub>. C loss in the incubation experiment (g g<sub>SDW</sub> h<sup>-1</sup>) ( $P$ ) was calculated from the CO<sub>2</sub>  
201 production:

$$202 \quad P = g \text{ CO}_2 \text{ g}^{-1} \text{ h}^{-1} (M_C / M_{CO_2}) \quad \text{Eq. 3}$$

203 where  $M_C$  is the molar mass of C (g mol<sup>-1</sup>).

204 *Calculation of soil C storage change*

205 The change in soil C stock was calculated as LOI change between the sampling times and from the initial  
206 value for different soils and sites (2002--2011). The proportion of C in the LOI used in the calculation was  
207 0.56 (Hoogsteen *et al.*, 2015). Estimates of soil C stock changes were calculated per standard 25 m<sup>3</sup> of soil  
208 per tree (Rakennustietosäätiö, 2010) instead of the actual soil volume allocated for each tree at the study  
209 sites, to allow easier comparison between sites, soil mixes and tree species. This estimate was compared

210 with one based on the incubation experiment. Since the first soil sampling for incubation was performed in  
211 2005, the comparison covered the years between 2005 and 2011.

212

213 The incubation-based estimate was obtained in two steps. First, we developed an Arrhenius-type model to  
214 predict C loss as a function of the measured  $T_f$ , SW and LOI content and parameterized it with the soil  
215 incubation data (Eq. 4). The three estimated parameters were used in the model (Eq. 5); the intercept ( $\lambda$ ),  
216 combined LOI and SW response parameter  $\alpha$  and parameter  $\beta$  describing the temperature response. After the  
217 testing of year-, soil- and site- specific datasets, the least number of separate models that presented no  
218 heteroscedasticity problems were chosen. As a result, the model was fitted separately for each soil type and  
219 site ( $n = 40$  in each combination):

220

$$221 \ln(P) = \lambda + (\alpha \ln(O) \ln(W)) + \beta T, \quad \text{Eq. 4}$$

222 where  $O$  is the LOI of the soil sample (% of dry weight),  $W$  (% of dry weight) is the SW of the soil sample  
223 and  $\lambda$ ,  $\alpha$  and  $\beta$  were the fitted parameters. The model was then applied to predict the soil C loss, based on the  
224 measured  $T_f$  and water content at the studied sites:

$$225 C \text{ loss} = \exp(\lambda + (\alpha (\ln O_e) \ln W_f) + \beta T_f + (RMSE/2)^2), \quad \text{Eq. 5}$$

226 where  $\lambda$ ,  $\alpha$  and  $\beta$  are the parameters fitted in the preceding step,  $W_f$  is the measured SW content (% weight)  
227 and  $T_f$  is soil temperature ( $^{\circ}\text{C}$ ) hourly averages. The correction for unbiasedness (Baskerville, 1972) was  
228 calculated from the root-mean-squared error (RMSE) of the parameter estimation. The initial LOI ( $O_{e \text{ ini}}$ ) at  
229 the starting point of the model was generated with a combined two first-order kinetic model (Bernal *et al.*,  
230 1998), based on the initial LOI and measurements from the soil samples taken in 2005, 2008 and 2011 as:

$$231 O_{e \text{ ini}} = LOI_{2002} - (LOI_{2002} (1 - \exp(-K_r t_m)) + S (1 - \exp(-K_s t_m))) \quad \text{Eq. 6}$$

232 where  $LOI_{2002}$  is the LOI of the soil at  $T_0$ ,  $t_m$  is time after establishment (in months), and  $K_s$ ,  $K_r$  and  $S$  are the  
233 fitted parameters. After the first time step of the incubation model, subsequent  $O_e$  values were obtained  
234 dynamically from the periodic  $\text{CO}_2$  production estimates calculated with the model. Half-hourly  
235 measurements of  $T_f$  at depths of 10 cm were used for depths of 0--20 cm, at 30 cm for depths of 20--45 cm  
236 and at 60 cm for depths of 45--100 cm. The SW measured at 10 cm was used for the 0--20-cm layer and  
237 measurement from 30 cm for lower levels.

238

239 The incubation model based total C loss for each  $25 \text{ m}^3$  of tree soil was calculated as:

$$240 \text{Total } C_{\text{inc}} \text{ loss} = C_{\text{tot}} \sigma v 25 \quad \text{Eq. 7}$$

241 where  $\sigma$  is the loose soil bulk density ( $\text{kg m}^{-3}$ ) and  $v$  is the proportion of fine soil in the structural soil ( $\text{m}^3 \text{ m}^{-3}$ )  
242  $^3$ ).

243 The LOI based C loss ( $\text{kg per } 25 \text{ m}^3$  of tree soil) was separately calculated, based on the LOI change between  
244 the soil construction and year 2011:

$$245 \quad \text{Total } C_{LOI} \text{ loss} = ((L_{2002} c \sigma v) - (L_{2011} c \sigma v)) 25 \quad \text{Eq. 8}$$

246 where  $L_{2011}$  is the measured loss on ignition ( $\text{g g}^{-1}$ ) measured in 2011,  $c$  is estimated proportion of C in the  
247 LOI (0.56; Hoogsteen *et al.*, 2015),  $\sigma$  is loose soil bulk density ( $\text{kg m}^{-3}$ ) and  $v$  is proportion of fine soil in the  
248 structural soil ( $\text{m}^3 \text{ m}^{-3}$ ).

#### 249 *Tree biomass measurements*

250 The development of tree aboveground woody biomass (ABW) at the study sites was calculated from  
251 measurements made of same 6--12 trees per species in August 2005, 2008 and 2011, and leaf biomass from  
252 measurements taken in all years from 2004 to 2011. The selection of observed trees (a combination of  
253 systematic and random sampling, each soil type at site was equally represented) and leaf area measurements  
254 are described in Riikonen *et al.* (2011). Tree trunk diameter was measured at soil surface, at height of 130  
255 cm and above all first-degree branches. The height and diameter of each first-degree branch were recorded.  
256 The living crown was divided horizontally into three equally long sections, and leaf samples were taken from  
257 the median diameter branch of each crown section for leaf area and dry weight measurements. A relationship  
258 was fitted between the branch basal area and leaf area and used to predict the leaf biomass for all branches  
259 within the tree crown (Riikonen *et al.* 2011).

260  
261 The branch woody biomass (WB) for *Tilia* was based on 206 and for *Alnus* 46 branches, bulked per species,  
262 pruned from the study site trees. A power function was fitted to estimate the branch dry biomass, based on its  
263 diameter:

$$264 \quad M_{wb} = \delta D_b^\mu \quad \text{Eq. 9}$$

266 Here,  $M_{wb}$  is the woody biomass of the branch (kg),  $D_b$  is branch diameter (mm), and  $\delta$  and  $\mu$  were the  
267 estimated parameters. The biomass of all branches in each tree was calculated based on the branch diameters  
268 measured. The tree trunk volume was calculated from the trunk diameter measurements as stacked cylinders  
269 and converted to biomass with specific gravity for *Tilia americana* L. (0.40) and *Alnus rubra* Bong. (0.43)  
270 (Alden, 1995).

271 Living tree roots encountered in the sampling pits during soil sampling in 2008 and 2011 were collected for  
272 root biomass estimation, washed and weighed ( $R_f$ ). Additionally, small roots that could not be separated in



273 the field were separated from the dried soil sample, weighed and calculated per kg of soil ( $R_d$ ). The total dry  
274 root biomass for each sampling pit was calculated, based on the roots collected in the soil sampling as:

$$275 \text{ Root DW} = ((R_f j) + (R_d M_p)) / V_p \quad \text{Eq. 10}$$

276 where  $j$  is the root dry-to-fresh weight ratio,  $M_p$  is total mass of the collected fine soil sample from the  
277 sampling pit (kg) and  $V_p$  is volume of the pit ( $\text{m}^3$ ).

278 The total tree root dry biomass ( $\text{g m}^{-3}$ ) was estimated from roots in the collected soil samples, assuming the  
279 root density was equal to the average of all sampling pits at each site. This value was assumed to represent  
280 root density for the entire root system. It is estimated that the root system covers an area with a radius of 38 x  
281 the DBH of the tree in question (Day *et al.*, 2010), which exceeded the tree soil dimensions already in 2008;  
282 thus it was assumed that roots had grown throughout the entire soil volume. We assumed that the root  
283 density in the root ball at planting time was equal to the value in 2011 and calculated an estimate of root  
284 biomass at planting, based on the root ball volume (diameter 100 cm).

#### 286 *C stocks in branch prunings and leaf litter*

287 The contribution of exported biomass fractions (tree litter C and pruned branches) to tree C sequestration  
288 was estimated with an exponential decay function (Olson, 1963), to account for these non-living  
289 compartments of tree sequestered C at a given time. The function was applied for each fallen leaf and  
290 pruned branch cohort separately for each year:

$$292 M_r = M_0 \exp(-kt) \quad \text{Eq. 11}$$

293 where  $M_r$  is the remaining leaf or branch biomass,  $M_0$  is leaf or branch biomass (kg) at  $t_0$  (years),  $k$  is decay  
294 factor (*Tilia* leaves: 0.24, Hobbie *et al.*, 2006, *Alnus* leaves: 0.6, Dilly and Munch, 1996, and branches of  
295 both species: 0.22, Perruchould *et al.*, 1999) and  $t$  is years passed. We assumed that the C in the leaves was  
296 initially equal to the total C content of that year's leaf cohort and was lost at a rate equal to the total mass  
297 loss. The WB removed by pruning was measured for one half of the trees, selected randomly, at each  
298 pruning.

#### 299 *Prediction of tree DBH, biomass and C*

300 To estimate the compensation point (the number of years after planting required for the tree C capture to  
301 reach the sum of the soil C loss in the first decade after planting, not accounting for possible soil C changes  
302 after first 10 years) for the case study trees, we needed to predict the long-term tree growth and biomass  
303 accumulation. For this we needed models that predict both the tree growth over time and how the biomass is  
304 distributed among the various tree parts. We compared existing DBH growth models (we found two for  
305 *Tilia*, but only one for *Alnus*) (Table 2) with observations from sample trees that we collected for this

306 purpose. The biomass and C stock predictions were based on BEs (Table 2, Appendix), assessed and selected  
307 by comparing the BE predictions with the measured values explained above.

308  
309 We collected the DBH values from 19 *Tilia* sp. street tree plantings in Helsinki with known planting year  
310 and at least six trees still remaining of the originally planted. We used either the latest DBH data from the  
311 City of Helsinki street tree database or measured at least six randomly chosen trees per planting in 2012—  
312 2014. Only two known *Alnus glutinosa* f. *pyramidalis* plantings in Helsinki are older than the case study site.  
313 Two datasets of earlier DBH measurements were available for one of the plantings and one set for the other,  
314 and additionally, all trees were measured for DBH in 2014: altogether, the resulting dataset on *Alnus* DBH  
315 had 5 data points (average DBH at given age). For *Tilia*, from 6 to 68, on average 22 trees were measured to  
316 attain the average DBH for a planting site, and for *Alnus*, one planting had 22 and the other, 91 trees.

317  
318 To predict the C content of the pruned branches, we applied a pruning regime the City of Helsinki aims for  
319 with street trees: the trees were pruned 1, 3, 5, 7, 10, 15 and 20 years after planting and thereafter every 10  
320 years. In the first 20 years, we applied a pruning of 25% and thereafter 15% of the branch biomass at each  
321 pruning.

322  
323 In all conversions from woody biomass to C, we used a 45% C content for both species. We used C content  
324 of 47.6% for *Tilia* (Niinemets, 1999) and 45.8% for *Alnus* (Browaldh, 1997) leaves, respectively. For  
325 purposes of predicting future C stocks of tree planting, we assumed there was no further soil C stock change  
326 after the first 10 years.

327  
328 *Statistical analysis*

329 The mean LOI change-based C loss for each site and soil type, and similarly, the measured and BE-predicted  
330 tree biomass compartments for individual trees in 2011, were each log-transformed and compared using 2-  
331 sided Tukey's test. The incubation-based C loss prediction model parameters were estimated with SAS  
332 procedure MODEL (linear regression, Eq. 4), and the residuals were assessed with White's  
333 heteroscedasticity test. The relationship between sample branch diameter and biomass in the tree biomass  
334 measurements was determined with nonlinear regression (Eq. 9). DBH growth models for *Tilia* were  
335 compared to measurements by calculating RMSE and bias and testing the significance of the latter with t-  
336 test. The p value required for significant difference and/or effect was set at  $\leq 0.05$  for all statistical analyses.

337

## 338 **Results**

339 *Soil C storage change*

340 The soil C stock at the time of establishment was approximately 400, 900 and 250 kg of C per tree in the 25  
341 m<sup>3</sup> of tree soil in soils 1, 2 and 3, respectively. The average loss of C in 2002—2011 across the soils was  
342 approximately 290 kg per 25 m<sup>3</sup>, as calculated from the LOI change (Figure 1). The highest proportional LOI  
343 loss in comparison to the original LOI was measured in soil 3; only about 1/3 remained in 2011 (Figure 2).  
344 The peat-based soil 1 maintained its LOI best, retaining on average, 56% of the original LOI value, with C  
345 loss estimates of 100-250 kg per 25 m<sup>3</sup>.

346 Based on the incubation model, the average annual soil C loss per m<sup>3</sup> in 2005—2011 was 0.41 kg, resulting  
347 in a slightly lower C loss over time in all soils than in the LOI loss-based estimate (Figure 3). Model  
348 parameter  $\beta$  describing the temperature response was highest in soil 2, but parameter  $\alpha$  combining the effects  
349 of LOI and SW was highest in soil 3 (Table 3). Periods of extrapolation to  $T_f$  values outside the incubation  
350 experiment range accounted for 17.5% of the C loss predicted by the model. The gap-filling in the SW data  
351 produced only 0.54% of the C loss estimate.

352  
353 The particle-size distributions showed that in 2002, 83% of the samples by weight on average were < 2 mm,  
354 but in 2011 the corresponding value was only 58%. Based on laser diffraction analysis, the clay content of all  
355 samples was < 1%.

#### 356 357 *Tree biomass C*

358 Based on biomass measurements, an average *Tilia* tree showed approximately 25 kg of C in its WB in 2011  
359 (Table 4) and sequestered approximately 18 kg C in total WB per tree in 2003—2011. The WB of an average  
360 *Alnus* tree had a stock of 37.5 kg of C in 2011. Including the C in the roots, C sequestration in the *Alnus* WB  
361 in 2003—2011 showed a total of 32 kg.

362 The branches used to predict *Tilia* and *Alnus* branch biomass, based on branch diameter, showed fairly good  
363 correlation between branch diameter and biomass; the R<sup>2</sup> values were 0.93 and 0.91, respectively. The  
364 estimated leaf and litter contribution to the total C stock was high, especially for *Tilia*: 7.3 kg per tree in  
365 2011 (Table 4). The prunings contributed relatively little to the C stock in either species. Adding the C  
366 estimated to remain in the leaf litter and pruned branches to the tree C sequestration estimates brought the  
367 sequestered C in 2011 to 26 kg per *Tilia* tree and for *Alnus* to 38 kg per tree.

368  
369 There were significant differences in all aboveground biomass compartments measured between the tree  
370 species, but only two of the seven BEs tested (Appendix) produced significantly different total or ABW  
371 estimates from the measured biomass (Table 5). However, all of the BEs showed significantly higher stem  
372 biomass and lower branch and leaf biomass in comparison to the measurements.

373  
374

375 *Tree C sequestration vs. soil C storage change*

376 Based on the estimates of soil C loss and tree C sequestration, the *Alnus* trees sequestered 10—15% of the  
377 mean soil C loss by 2011, while the *Tilia* trees achieved some 10% sequestration in comparison to the mean  
378 C losses from the tree soils, but the uncertainties in both soil C loss and tree C sequestration remain  
379 considerable. In the highest C loss soil, less than 10% of the C loss was sequestered by the trees, while in  
380 comparison to the lowest loss soil and site, the *Alnus* trees approached 40% sequestration in 2011. These  
381 percentages include the C in leaf litter and prunings, as estimated for 2011. Considering only the peat-based  
382 soil 1, *Alnus* trees had sequestered about 30 kg of C, while the C loss from soil 1 was estimated as 170 kg  
383 (average over the two sites, Figure 1).

384  
385 Of the available DBH growth models for long-term biomass accumulation estimation, DBH model 2  
386 coincided better with the measurements (Figure 6) with a relative RMSE of 20% and a bias of 0.7 cm, while  
387 DBH model 1 showed tree sizes similar to those at the best sites measured in Helsinki with relative RMSE of  
388 28% and a bias of -7.4 cm. Neither model predictions were significantly different from measurements  
389 however. The data collected from the *Alnus* plantings in Helsinki were too limited to assess the applicability  
390 of the prediction.

391 If *Tilia* growth followed prediction 2, the estimated C capture in the woody biomass of the *Tilia* trees would  
392 reach the mean soil C loss of the first decade about 55 years after transplanting (Figure 7) and the smallest  
393 soil C loss in this study (Soil 1 at the *Alnus* site, Figure 1) in about 30 years. The prediction for *Alnus* was  
394 more uncertain still, but it showed a biomass accumulation curve very similar to that in prediction 2 for *Tilia*,  
395 with *Alnus* some 5 years ahead (Figure 7). Taking the estimated litter and prunings C stock into account  
396 moved the previous compensation point estimates forward by 3 years for *Tilia*, but in the *Alnus* C stock, the  
397 litter and prunings had little effect.

398

399 **Discussion**

400 *Soil C storage change*

401 High C losses from tree planting soils were seen soon after soil construction. In the two compost-containing  
402 soil mixes studied, the LOI was halved in less than 10 years and most of the LOI loss took place before the  
403 first soil sampling in 2005. In addition to our study, rapid SOM decline has been demonstrated for other  
404 compost-derived growth media, especially when the compost was not sufficiently mature (e.g. Bernal et al.,  
405 1998; Sanchez-Monedero et al., 2004; Vidal-Beaudet et al., 2012). In the entirely peat-based soil mix 1, the  
406 LOI loss was not as high, consistent with studies of greenhouse growth media (e.g. Prasad and O'Shea,  
407 1997; Prasad and Maher, 2003).

408

409 The incubation model gave lower values for C loss than the LOI loss measured, and the parameters for the  
410 model differed clearly between sites. Sampling structural soil is complicated by the presence of load-bearing  
411 stones, and consequently the samples incubated were heavily disturbed. This may have affected the resulting  
412 CO<sub>2</sub> production rates. The absence of active tree roots may also have had a suppressing effect on SOM  
413 mineralization (Linden *et al.*, 2014). The estimated parameters showed the highest temperature response in  
414 the highest LOI soil mix, likely due to the higher amount of decomposable organic material. The water  
415 content and LOI response were more varied, probably because the ranges of these variables did not entirely  
416 overlap in the various soils and sites. Overall, the number of samples per soil and site was rather low, and the  
417 site differences may be exaggerated. On the other hand, the differences between sites, such as the planted  
418 tree species (e.g. Bomberg and Timonen 2009) or soil water status (e.g. Williams and Rice 2007) in long  
419 term may have led to different soil microbial community and SOM decomposition on initially similar soil  
420 mixes.

421  
422 Based on the change in particle-size distribution and visual inspection of soil samples, an increase in coarse  
423 material was seen in the fine fraction after soil construction, originating from breakage of stones in the  
424 structural soil during handling and tamping. Assuming the change in finer fractions was similar to that seen  
425 in the coarse fraction, an addition of some 25% in volume can be roughly estimated. This would reduce the  
426 LOI loss estimates by 20%. From Figure 7 we can assess that such an error has a relatively small impact on  
427 the compensation point estimates. The clay content of the samples was measured with the laser diffraction  
428 method, which gives lower values for clay content than does the pipette method (2–3 times less; Taubner *et*  
429 *al.*, 2009). The clay content was low nevertheless, and no clay correction was applied in the C stock  
430 calculation.

431  
432 At the time of the site establishment, the LOI of the growth media was not separately measured. The initial  
433 LOI values were bulk values from the growth media trade descriptions (soils 1 and 2) and the LOI values of  
434 the organic matter used in the soil mix for soil 3 (specially prepared > 200- $\mu$ m<sup>3</sup> mix). The LOI change  
435 estimates were net C changes in the soil, i.e. the possible contribution of root litter to the soil C stock was  
436 included in the net effect calculation; it would add SOM to the soil and thus reduce the soil C loss observed.  
437 The agreement seen between the LOI and incubation-based model, implies that the uncertainties were  
438 probably not major. However, the estimates of LOI half-life are especially sensitive to the types of  
439 uncertainties present in the data.

440  
441 *Tree C sequestration*

442 For the relatively small trees measured in this study, the differences between the various BEs for total  
443 aboveground biomass (AB) or total ABW and the measured biomass were fairly small for five of the seven  
444 equations. The only model for urban *Tilia* trees (McHale *et al.*, 2009) gave much higher values than the other

445 BEs, but the authors noted that the trees were irrigated and fertilized. For *Alnus*, BE 5a deviated furthest  
446 from the measured biomass. The source of the equation also gave separate BEs for the stem and branches  
447 (BEs 5b and 5c); summing these up resulted in better estimates (Table 5).

448

449 While the total aboveground BEs performed fairly well, the equations for the various AB compartments –  
450 leaves, branches and trunk – did not. In *Alnus*, this may have been due to the different crown form (f.  
451 *pyramidalis*) in comparison to the BEs and the lack of large branches in the biomass samples. For *Tilia*,  
452 however, the branch biomass was more likely under- than overestimated, because large branches in the  
453 biomass samples were often reduction-pruned previously. The differences between the measurements and  
454 BE predictions imply that in *Tilia* street trees, the biomass distribution within the crown likely differs from  
455 that of *Tilia* in traditional forests. Similar findings were recently obtained for three other species in open sites  
456 (Zhou *et al.*, 2014); stem biomass was low and branch biomass was high, but total aboveground biomass  
457 agreed somewhat with traditional forest based equations. More data needs to be gathered however, as our  
458 study only concerns a small number of trees from two species.

459

460 The leaf biomass of the trees measured was also consistently higher than the BEs predicted. In our data, the  
461 *Tilia* trees annually invested approximately 40--50% and *Alnus* 30--40% of the total aboveground C increase  
462 into leaves, which is on the high side but within the range for similarly aged stands in allocation studies  
463 reviewed by Litton *et al.* (2007). The roots of the *Tilia* trees appeared to have relatively more C stock than  
464 the roots of the *Alnus* trees. This may have resulted from the uncertainties in root sampling however, since  
465 the variation between samples was high. Based on the literature, belowground biomass can be estimated to  
466 be 23% of the AB (Chojnacky *et al.*, 2014), giving estimates surprisingly close to the measurements, so  
467 while our data is very uncertain, using literature based values instead would have had little effect on the  
468 overall results.

469

470 The contribution of litter and prunings to overall tree C sequestration appeared large for the still rather small  
471 case study trees. However, the literature-based decay factors may not have performed well under our  
472 conditions; the contradictory results concerning urban vs. rural litter decomposition rates (Pouyat *et al.*,  
473 1997; Pouyat and Carreiro, 2003; Nikula *et al.*, 2010; Dorendorf *et al.*, 2015) indicate that the processes  
474 related to urban litter are not yet well understood. When the soil surface is sealed, the aboveground litter is  
475 likely lost from the tree-soil system, warranting leaving it out of the C sequestration estimates. However, the  
476 C input in leaves was quite high in the case study trees, indicating that improving leaf C retention and  
477 longevity may be one key factor in improving urban tree C sequestration.

478

479 *Tree C sequestration vs. soil C storage change*

480 In tree C sequestration, the favourable growth rate of trees is critical, because biomass is a function of tree  
481 size, and tree size growth is dependent on the growing conditions. The uncertainty in tree C sequestration

482 predictions in general is thus mainly linked with the difficulty in predicting tree growth rate and mortality  
483 (Strohbach *et al.*, 2012); this is easily seen in the DBH data of existing *Tilia* plantings in Helsinki (Figure 6).  
484 Identifying the remaining original trees and the planting year data was not easy in all cases, which may  
485 explain some of the variation. Since the BEs appeared to underestimate branch biomass, the biomass in the  
486 prunings was likely also underestimated. This may have been compensated for somewhat, because the  
487 pruned branch biomass was not removed from the predicted tree biomass.

488  
489 In forest ecosystems, trees accumulate C as they grow, but soil C dynamics are dependent on litter input and  
490 decomposition, which can be affected by disturbances such as tree felling. Research on traditional forest soil  
491 C stock temporal dynamics is often contradictory and confounded by the various initial states (e.g. fire,  
492 harvest or plantation) of the system (Yang *et al.*, 2011). Similarly, planting of urban trees is usually preceded  
493 by a land-use change, making it difficult to draw parallels with native ecosystems; however, there are some  
494 studies showing recovery of urban soil C stocks over time (e.g. Golubiewski, 2006; Bae and Ryu, 2015;  
495 Setälä *et al.* 2016). In managed traditional forests of Finland, soil C stock decreases for approximately 20  
496 years after clearcutting (Peltoniemi *et al.*, 2004) but net C sequestration in the ecosystem can be reached  
497 after some 10--15 years (Kolari *et al.*, 2004). The most positive compensation point estimates in this study  
498 were similar, yet we must conclude that the often stated expected street tree lifespan of 20--30 years (Roman  
499 and Scatena, 2011) appears unlikely to bring tree C sequestration benefits in soil mixes in the study, even if  
500 only the entirely peat-originating loss of C on soil 1 is considered accountable. Fortunately, the annual C  
501 capture of young trees can be expected to improve for several decades (Pregitzer and Euskirchen, 2004).

502  
503 It is also likely that the belowground litter C input will increase the soil C stock over time. A pattern of initial  
504 soil C loss, followed by soil C stock increase, in conjunction with change to urban land use has been  
505 emerging in the USA (Pataki *et al.* 2006). In our study, soil C loss was high in the beginning, but stabilized  
506 after the first few years (Figure 2); it is possible that soil C will start to increase as time passes, paralleling  
507 this type of development. The estimates of compensation point timing are thus very uncertain in relation to  
508 possible soil C stock increase over time and must be considered as only suggestions for the minimum tree  
509 life expectancy needed to produce net C sequestration by planting street trees in artificial growth media. The  
510 design of our study was not particularly well suited for generalization of the results but rather it  
511 demonstrated possible long term C dynamics in urban planting schemes. There are many C expenses related  
512 to street trees that we are not accounting for, such as C cost of maintenance (McPherson *et al.* 2015) and cast  
513 iron tree grates and trunk guards. Despite these shortcomings, the range of results indicate that the soil C  
514 dynamics need attention in urban tree C sequestration estimates overall.

515  
516 Both peat and compost derived organic matter are currently used in growth media; the latter appears to be the  
517 more sustainable choice, although the sustainability determined is greatly dependent on the assessment

518 method chosen (Defra, 2009). The use of peat in composting process adds to the difficulty of choosing  
519 suitable tree soil. While high SOM contents have long been considered beneficial for soil quality (Reeves,  
520 1997), perhaps the amount of SOM used in tree soils could be optimized more carefully. Biochar and other  
521 e.g. clay-stabilized C soil additives appear very attractive new options for C sources in artificial growth  
522 media due to their relative stability in comparison to compost (Bolan *et al.* 2012, Ameloot *et al.* 2013).

523

524 However, the C cost of tree planting, or the possible lack of C-related overall benefits, should not discourage  
525 people from planting street trees. C sequestration is only one of the ecosystem services provided by trees,  
526 and its value is usually estimated to be small in comparison to storm water management, property value and  
527 energy-saving benefits, not to mention recreational and cultural values and human health benefits  
528 (McPherson *et al.*, 2005). Rather, these results should encourage more investment and interest in tree  
529 planning, establishment and maintenance to ensure improvements in urban tree lifespan and eventual tree C  
530 sequestration. Overall, using local soils and less and lighter infrastructure in tree plantings would likely lead  
531 to higher net C sequestration. Unfortunately, these appear to be a rare option in the midst of efforts towards  
532 more efficient urban land use.

533

## 534 **Conclusions**

535 In considering the C sequestration capacity of street trees, the initial C losses from peat-containing soils after  
536 site establishment should not be overlooked. Due to the exponential nature of the increase in tree C stock  
537 over time, tree planting-related C expenses are best compensated for with a long tree life expectancy. This is  
538 best achieved with adequate consideration of tree growth requirements over their entire life cycle.

539 Compensating for the smallest measured single tree soil C stock loss of 100 kg by C sequestration of the tree  
540 in this case study was predicted to require at least 30 years. This exceeds current estimates of average street  
541 tree lifespan. Especially the estimates of tree root C stock and soil C development in the future were  
542 uncertain however.

543

544 Further research on biomass distribution within urban trees, both above and under ground, is needed to  
545 improve the accuracy of urban C stock and C stock change estimates. The use of traditional forest based BEs  
546 for urban trees may lead to errors when specific AB compartments are estimated. This causes further  
547 problems in assessment of the effects of leaf litter and prunings in urban tree C sequestration. Cutting the C  
548 expenses of the tree planting and focusing on the longevity of pruning and leaf litter C may aid in balancing  
549 tree-related C effects.

550



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558

559 **References**

560 Alden, H.A., 1995. Hardwoods of North America. General Technical Report-Forest Products Laboratory,  
561 USDA Forest Service, (FPL-GTR-83).

562

563 Ameloot, N., Graber, E. R., Verheijen, F. G., De Neve, S. 2013. Interactions between biochar stability and  
564 soil organisms: review and research needs. *European Journal of Soil Science* 64, 379-390.

565

566 Bae, J., Ryu, Y. 2015. Land use and land cover changes explain spatial and temporal variations of the soil  
567 organic carbon stocks in a constructed urban park. *Landscape and Urban Planning* 136, 57-67.

568

569 Baskerville, G. L., 1972. Use of logarithmic regression in the estimation of plant biomass. *Canadian Journal*  
570 *of Forest Research* 2, 49-53.

571

572 Bernal, M.P., Sanchez-Monedero, M.A., Paredes, C., Roig, A., 1998. Carbon mineralization from organic  
573 wastes at different composting stages during their incubation with soil. *Agriculture, Ecosystems &*  
574 *Environment* 69, 175-189.

575

576 Bolan, N.S., Kunhikrishnan, A., Choppala, G.K., Thangarajan, R., Chung, J. W., 2012. Stabilization of  
577 carbon in composts and biochars in relation to carbon sequestration and soil fertility. *Science of the Total*  
578 *Environment* 424, 264-270.

579

580 Bomberg, M., Timonen, S. 2009. Effect of tree species and mycorrhizal colonization on the archaeal  
581 population of boreal forest rhizospheres. *Applied and environmental microbiology*, 75, 308-315.

582

583 Brenneman, B.B., Gardner, W.E., Schoenhofen, L.H., Marsh, P.L., 1978. Biomass of species and stands of  
584 West Virginia hardwoods. In: *Proceedings of Central Hardwood Forest Conference II* (ed. Pope, P.E.). West  
585 Lafayette, Purdue University.

586

587 Browaldh, M., 1997. Nitrogen release from leaves and prunings of different tree species used as green  
588 manures. *Biological Agriculture & Horticulture* 14, 309-322.  
589

590 Bunce, R.G.H., 1968. Biomass and production of trees in a mixed deciduous woodland. I. Girth and height  
591 as parameters for the estimation of tree dry weight. *Journal of Ecology* 56, 759–775.  
592

593 Böckmann, T., 1990. Ertragstafel für Winterlinde (*Tilia cordata* Mill.) in Niedersachsen und Nordhessen.  
594 *Allgemeine Forst- und Jagdzeitung* 162, 28-34.  
595

596 Chojnacky, D.C., Heath, L.S., Jenkins, J.C., 2014. Updated generalized biomass equations for North  
597 American tree species. *Forestry* 87, 129–151.  
598

599 Day, S.D., Wiseman, E., Dickinson, S.B., Harris J.R., 2010. Contemporary concepts of root system  
600 architecture of urban trees. *Arboriculture & Urban Forestry* 364, 149-159.  
601

602 Defra, 2009. A preliminary assessment of the greenhouse gases associated with growing media materials.  
603 IFO154. Research Final Report. SID 5. <http://randd.defra.gov.uk>. Retrieved 05/2015.  
604

605 Dilly, O., Munch, J.-C., 1996. Microbial biomass content, basal respiration and enzyme activities during the  
606 course of decomposition of leaf litter in black alder (*Alnus glutinosa* (L.) Gaertn.) forest. *Soil Biology and*  
607 *Biochemistry* 28: 1073-1081.  
608

609 Dorendorf, J., Wilken, A., Eschenbach, A., Jensen, K., 2015. Urban-induced changes in tree leaf litter  
610 accelerate decomposition. *Ecological Processes* 4, 1-16.  
611

612 Golubiewski, N. E., 2006. Urbanization increases grassland carbon pools: Effects of landscaping in  
613 Colorado's front range. *Ecological Applications* 16, 555-571.  
614

615 Grabosky, J., Bassuk, N., 1995. A new urban tree soil to safely increase rooting volumes under sidewalks.  
616 *Journal of Arboriculture* 21, 187-187.  
617

618 Hobbie, S.E., Reich, P.B., Oleksyn, J., Ogdahl, M., Zytowski, R., Hale, C., Karolewski, P., 2006. Tree  
619 species effects on decomposition and forest floor dynamics in a common garden. *Ecology* 87, 2288-2297.  
620

621 Hoogsteen, M.J.J., Lantinga, E.A., Bakker, E.J., Groot, J.C.J., Tittonell, P.A., 2015. Estimating soil organic  
622 carbon through loss on ignition: effects of ignition conditions and structural water loss. *European Journal of*  
623 *Soil Science* 66, 320-328.

624  
625 Hughes, M.K., 1971. Tree biocontent, net production and litter fall in a deciduous woodland. *Oikos* 22, 62–  
626 73.  
627 Himanen, M., Hänninen, K. 2011. Composting of bio-waste, aerobic and anaerobic sludges—ludges, aerobic  
628 and anaerobic sludgesdeciduous woodland. *OiBioresource technology* 102, 2842-2852.  
629  
630 IPCC 2006, 2006 IPCC Guidelines for National Greenhouse Gas Inventories, Prepared by the National  
631 Greenhouse Gas Inventories Programme, Eggleston H.S., Buendia L., Miwa K., Ngara T., and Tanabe K.  
632 (eds). Published: IGES, Japan.  
633  
634 Jaakkola, A., Simojoki, A., 1998. Effect of soil wetness on air composition and nitrous oxide emission in a  
635 loam soil. *Agricultural and Food Science in Finland* 7, 491-505.  
636  
637 Johansson, T., 1999. Dry matter amounts and increment in 21- to 91-year-old common alder and grey alder  
638 and some practical implications. *Canadian Journal of Forest Research* 29, 1679–1690.  
639  
640 Johansson, T., 2000. Biomass equations for determining functions of common and grey alder growing on  
641 abandoned farmland and some practical implications. *Biomass and Bioenergy* 18, 147–159.  
642  
643 Kolari P., Pumpanen J., Rannik Ü, Ilvesniemi H., Hari P., Berninger F., 2004. Carbon balance of different  
644 aged Scots pine forests in southern Finland. *Global Change Biology* 10, 1106–1119.  
645  
646 Kornelsen, K., Coulibaly, P., 2012. Comparison of interpolation, statistical, and data-driven methods for  
647 imputation of missing values in a distributed soil moisture dataset. *Journal of Hydrologic Engineering* 19,  
648 26-43.  
649  
650 Kristoffersen, P., 1999. Growing trees in road foundation materials. *Arboricultural Journal* 23, 57-76.  
651  
652 Larsen, F.K., Kristoffersen, P., 2002. Tilia's physical dimensions over time. *Journal of Arboriculture* 28,  
653 209-214.  
654  
655 Linden, A., Heinonsalo, J., Buchmann, N., Oinonen, M., Sonninen, E., Hiltunen, E., & Pumpanen, J., 2014.  
656 Contrasting effects of increased carbon input on boreal SOM decomposition with and without presence of  
657 living root system of *Pinus sylvestris* L. *Plant and Soil* 377, 145-158.  
658

659 Liski, J., Lehtonen, A., Palosuo, T., Peltoniemi, M., Eggers, T., Muukkonen, P., and Mäkipää, R., 2006.  
660 Carbon accumulation in Finland's forests 1922-2004 - an estimate obtained by combination of forest  
661 inventory data with modelling of biomass, litter and soil. *Annals of Forest Science* 63, 687-697.  
662  
663 Litton, C.M., Raich, J.W., Ryan, M.G., 2007. Carbon allocation in forest ecosystems. *Global Change*  
664 *Biology* 13, 2089-2109.  
665  
666 McHale M., Burke, I., Lefsky, M., Peper, P., McPherson, E., 2009. Urban forest biomass estimates: is it  
667 important to use allometric relationships developed specifically for urban trees? *Urban Ecosystems* 12, 95-  
668 113.  
669  
670 McPherson, E. G., Kendall, A., Albers, S., 2015. Life cycle assessment of carbon dioxide for different  
671 arboricultural practices in Los Angeles, CA. *Urban Forestry & Urban Greening* 14, 388-397.  
672  
673 McPherson, G., Simpson, J.R., Peper, P.J., Maco, S.E., & Xiao, Q., 2005. Municipal forest benefits and costs  
674 in five US cities. *Journal of Forestry* 103, 411-416.  
675  
676 Neal, B.A., Whitlow, T.H., 1997. Using tree growth rates to evaluate urban tree planting specifications.  
677 *Journal of Environmental Horticulture* 15, 115-118.  
678  
679 Niinemets, Ü., 1999. Energy requirement for foliage formation is not constant along canopy light gradients  
680 in temperate deciduous trees. *New Phytologist* 141, 459-470.  
681  
682 Nikula, S., Vapaavuori, E., & Manninen, S., 2010. Urbanization-related changes in European aspen (*Populus*  
683 *tremula* L.): Leaf traits and litter decomposition. *Environmental Pollution* 158, 2132-2142.  
684  
685 Olson, J.S., 1963. Energy storage and the balance of producers and decomposers in ecological systems.  
686 *Ecology* 44, 322-331.  
687  
688 Pataki, D.E., Alig, R.J., Fung, A.S., Golubiewski, N.E., Kennedy, C.A., McPherson, E.G., Nowak, D.J.,  
689 Poyat, R. V., Romero Lankao, P., 2006. Urban ecosystems and the North American carbon cycle. *Global*  
690 *Change Biology* 12, 2092-2102.  
691  
692 Peltoniemi, M., Mäkipää, R., Liski, J., Tamminen, P. 2004. Changes in soil carbon with stand age – an  
693 evaluation of a modelling method with empirical data. *Global Change Biology* 10: 2078-2091.  
694

695 Peper, P.J., Alzate, C.P., McNeil, J.W. , Hashemi, J., 2014. Allometric equations for urban ash trees  
696 (*Fraxinus* spp.) in Oakville, Southern Ontario, Canada, *Urban Forestry & Urban Greening* 13, 175-183.  
697

698 Perala D,A., Alban D.H., 1994. Allometric biomass estimators for Aspen-dominated ecosystems in the upper  
699 Great Lakes. Research Paper NC-314. USDA Forest Service, North Central Experiment Station, St. Paul,  
700 MN.  
701

702 Perruchould, D., Joos, F., Fischlin, A., Hajdas, I., Bonani, G., 1999. Evaluating timescales of carbon  
703 turnover in temperate forest soils with radiocarbon data. *Global Biogeochemical Cycles* 13, 555-573.  
704

705 Pouyat, R.V., Carreiro, M.M., 2003. Controls on mass loss and nitrogen dynamics of oak leaf litter along an  
706 urban-rural land-use gradient. *Oecologia* 135, 288-298.  
707

708 Pouyat, R.V., McDonnell, M.J., Pickett, S.T., 1997. Litter decomposition and nitrogen mineralization in oak  
709 stands along an urban-rural land use gradient. *Urban Ecosystems* 1, 117-131.  
710

711 Prasad, M., Maher, M.J., 2003. Stability of peat alternatives and use of moderately decomposed peat as a  
712 structure builder in growing media. In: South Pacific Soilless Culture Conference-SPSCC 648, pp. 145-151.  
713

714 Prasad, M., O'Shea, J., 1997. Relative breakdown of peat and non-peat growing media. In: International  
715 Symposium on Growing Media and Hydroponics 481, pp. 121-128.  
716

717 Pregitzer, K.S., Euskirchen, E.S., 2004. Carbon cycling and storage in world forests: biome patterns related  
718 to forest age. *Global Change Biology* 10, pp. 2052–2077.  
719

720 Rakennustietosäätiö RTS, 2010. InfraRYL 2010, infrarakentamisen yleiset laatuvaatimukset 2010 (Quality  
721 guidelines of infrastructure construction; language Finnish). Osa 1, Väylät ja alueet. Helsinki: Rakennustieto,  
722 2010.  
723

724 Reeves, D. W., 1997. The role of soil organic matter in maintaining soil quality in continuous cropping  
725 systems. *Soil and Tillage Research* 43, 131-167.  
726

727 Riikonen, A., Lindén, L., Pulkkinen, M., Nikinmaa, E., 2011. Post-transplant crown allometry and shoot  
728 growth of two species of street trees. *Urban Forestry & Urban Greening* 10, 87-94.  
729

730 Roman, L.A., Scatena, F.N. 2011. Street tree survival rates: Meta-analysis of previous studies and  
731 application to a field survey in Philadelphia, PA, USA. *Urban Forestry & Urban Greening* 10, 269-274.

732

733 Sanchez-Monedero, M.A., Mondini, C., De Nobili, M., Leita, L., Roig, A. 2004. Land application of  
734 biosolids. Soil response to different stabilization degree of the treated organic matter. *Waste Management*  
735 24, 325-332.

736

737 Scalenghe, R., Marsan, F.A., 2009. The anthropogenic sealing of soils in urban areas. *Landscape and Urban*  
738 *Planning* 90, 1-10.

739

740 Schober, R., 1987. Ertragstabellen wichtiger Baumarten bei verschiedener Durchforstung. J.D. Dauerländer's  
741 Verlag, Frankfurt am Main, Germany, 166 p. In German.

742

743 Setälä, H. M., Francini, G., Allen, J. A., Hui, N., Jumpponen, A., Kotze, D. J., 2016. Vegetation type and  
744 age drive changes in soil properties, nitrogen, and carbon sequestration in urban parks under cold climate.  
745 *Frontiers in Ecology and Evolution* 4, 93.

746

747 Strohbach, M.W., Arnold, E., Haase, D., 2012. The carbon footprint of urban green space – A life cycle  
748 approach. *Landscape and Urban Planning* 104, 220-229.

749

750 Taubner, H., Roth, B., Tippkötter, R., 2009. Determination of soil texture: Comparison of the sedimentation  
751 method and the laser-diffraction analysis. *Journal of Plant Nutrition and Soil Science* 172, 161-171.

752

753 Ter-Mikaelian, M.T., Korzukhin, M.D., 1997. Biomass equations for sixty-five North American tree species.  
754 *Forest Ecology and Management* 97, 1-24

755

756 Vidal-Beaudet, L., Grosbellet, C., Forget-Caubel, V., Charpentier, S., 2012. Modelling long-term carbon  
757 dynamics in soils reconstituted with large quantities of organic matter. *European Journal of Soil Science* 63,  
758 787-797.

759

760 Williams, M. A., Rice, C. W. 2007. Seven years of enhanced water availability influences the physiological,  
761 structural, and functional attributes of a soil microbial community. *Applied Soil Ecology*, 35, 535-545.

762

763 Yang, Y., Luo, Y., Finzi, A. C. 2011. Carbon and nitrogen dynamics during forest stand development: a  
764 global synthesis. *New Phytologist* 190, 977-989.

765

766 Zhou, X., Schoeneberger, M.M., Brandle, J.R., Awada, T.N., Chu, J., Martin, D.L., Li, J., Li, Y., Mize,  
767 C.W., 2014. Analyzing the uncertainties in use of forest-derived biomass equations for open-grown trees in  
768 agricultural land. *Forest Science* 61, 144-161.  
769

DRAFT

770 Table 1. A summary of major measurements performed for the case study (Tilia and Alnus sites in 2003-  
771 2011), and numbers of samples taken and/or analyzed, at Tilia and Alnus sites. T= temperature (°C) D=  
772 diameter (mm), BM= biomass (g).  
773

Site and measurement	Soils				Trees			
	T <sub>f</sub> (every 30 min)	W <sub>f</sub> (every 30 min)	LOI, loose BD, particle size distribution	T, SWC, LOI response of CO <sub>2</sub> production	Trunk and branch D	Leaf area	Prunings (BM and D/BM ratio)	Root BM
Tilia site	2003-2011, 3 sensors at 30, 60, and 90 cm depth, respectively	2003-2011, 3 sensors at 30 and 60 cm depth, respectively	2005 (n=6), 2008 (n=15), 2011 (n=12)	2005 (n=6), 2008 (n=12), 2011 (n=12)	2005 (n=8), 2008 (n=7), 2011 (n=6)	2004-2011 trees, 3 branches per tree	Every time trees were pruned. (2008-2011, n=7)	2005, 2008, 2011 (n=6)
Alnus site	2003-2011, 3 sensors at 30, 60, and 90 cm depth, respectively	2003-2011, 3 sensors at 30 and 60 cm depth, respectively	2005 (n=6), 2008 (n=15), 2011 (n=12)	2005 (n=6), 2008 (n=12), 2011 (n=12)	2005 (n=10), 2008 (n=7), 2011 (n=6)	2004-2011 trees, 3 branches per tree	Whenever pruned (2010, n=11)	2005, 2008, 2011 (n=6)

774  
775 Table 2. Details of the DBH growth and biomass C accumulation forecast models. BE s (biomass equations)  
776 referred can be found in the Appendix.

	Forecast 1 for <i>Tilia</i>	Forecast 2 for <i>Tilia</i>	Forecast 1 for <i>Alnus</i>
<b><i>DBH growth forecast</i></b>			
	Street trees (Larsen and Kristoffersen, 2002)	Traditional forest trees (Yield class III, Böckmann, 1990)	Traditional forest trees (Yield class III, Schober, 1987)
<b><i>Biomass accumulation by compartment</i></b>			
Aboveground woody		BE 1	BE 7a minus leaves (7b)



Roots	23% of aboveground woody biomass (Chojnacky <i>et al.</i> , 2014)	23% of aboveground woody biomass (Chojnacky <i>et al.</i> , 2014)
<b>Litter</b>		
Leaf	BE 3b	BE 7b
Branch (prunings)	BE 3c	BE 5b
<i>Decay function; parameters (leaves, branches)</i>	Olson 1963; 0.24 (Hobbie <i>et al.</i> , 2006), 0.22 (Perruchould <i>et al.</i> , 1999)	Olson 1963; 0.60 (Dilly and Munch, 1996), 0.22 (Perruchould <i>et al.</i> , 1999)

777

778 Table 3. Parameter values  $\pm$  SD,  $R^2$  and White's heteroscedasticity test p-value for soil and site specific P  
779 prediction models (Eq. 4). The parameters  $\lambda$  (intercept),  $\alpha$  (SW and LOI parameter), and  $\beta$  (temperature  
780 response parameter) were used to predict C loss (Eq. 5).

Site	Soil mix	$\lambda$	$\alpha$	$\beta$	Adj. $R^2$	White's test p-value
Tilia	1	-18.02 $\pm$ 0.68	0.083 $\pm$ 0.013	0.08 $\pm$ 0.149	0.51	0.19
	2	-19.43 $\pm$ 0.64	0.083 $\pm$ 0.011	0.44 $\pm$ 0.078	0.68	0.57
	3	-17.81 $\pm$ 0.17	0.065 $\pm$ 0.008	0.34 $\pm$ 0.083	0.65	0.08
Alnus	1	-19.16 $\pm$ 0.21	0.084 $\pm$ 0.011	0.48 $\pm$ 0.083	0.71	0.43
	2	-18.70 $\pm$ 0.26	0.108 $\pm$ 0.008	0.16 $\pm$ 0.034	0.84	0.82
	3	-18.91 $\pm$ 0.14	0.090 $\pm$ 0.006	0.67 $\pm$ 0.142	0.85	0.50

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784

785 Table 4. Estimated C in the *Tilia* and *Alnus* aboveground stem, branch, roots and total C in woody biomass; cumulative original C in leaves and pruned  
 786 branches and remaining C in pruned branches and leaf litter during the period examined from 2002 to 2011 (kg per tree  $\pm$  SD when estimable). The remaining  
 787 C in the leaves and branches was calculated for each cohort separately and summed up. ABW = aboveground woody biomass.

	Year	Stem	Branches	ABW	Roots	Total woody	Leaf C remaining	Prunings C remaining	Total litter + prunings	Sum
<i>Tilia</i>	At planting	n/a	n/a	6.7*	0.2	6.9*	0	0	0	6.9*
	2005	2.5 $\pm$ 0.4	2.3 $\pm$ 0.6	4.9 $\pm$ 0.9	n/a	6.3**	1.5	0	1.5	7.8**
	2008	5.4 $\pm$ 2.1	4.1 $\pm$ 2.1	9.5 $\pm$ 4.0	3.4 $\pm$ 5.2	12.9	3.9	0	3.9	16.8
	2011	8.4 $\pm$ 4.1	8.2 $\pm$ 2.8	16.6 $\pm$ 6.7	8.5 $\pm$ 9.7	25.1	7.3	0.6	7.9	33.0
	C stock increase	n/a	n/a	9.9	8.3	18.2	7.3	0.6	7.9	26.1
<i>Alnus</i>	At planting	n/a	n/a	5.6*	0.1	5.7†	0	0	0	5.7*
	2005	4.2 $\pm$ 0.8	2.9 $\pm$ 0.6	7.1 $\pm$ 1.3	n/a	9.2**	2.1	0	2.1	11.3**
	2008	8.2 $\pm$ 1.3	6.9 $\pm$ 1.5	15.1 $\pm$ 1.0	3.1 $\pm$ 4.1	16.5	3.8	0	3.8	22.0
	2011	17.6 $\pm$ 2.8	12.6 $\pm$ 2.7	30.2 $\pm$ 4.3	7.3 $\pm$ 7.1	36.0	6.3	0.1	6.3	43.9
	C stock increase	n/a	n/a	24.6	7.2	31.8	6.3	0.1	6.3	38.2

788  
 789 \* ABW with BE 1.

790 † ABW with BE 5a

791 \*\*Root biomass estimated as 23% of ABW (Chojnacky *et al.*, 2014) added to total woody biomass

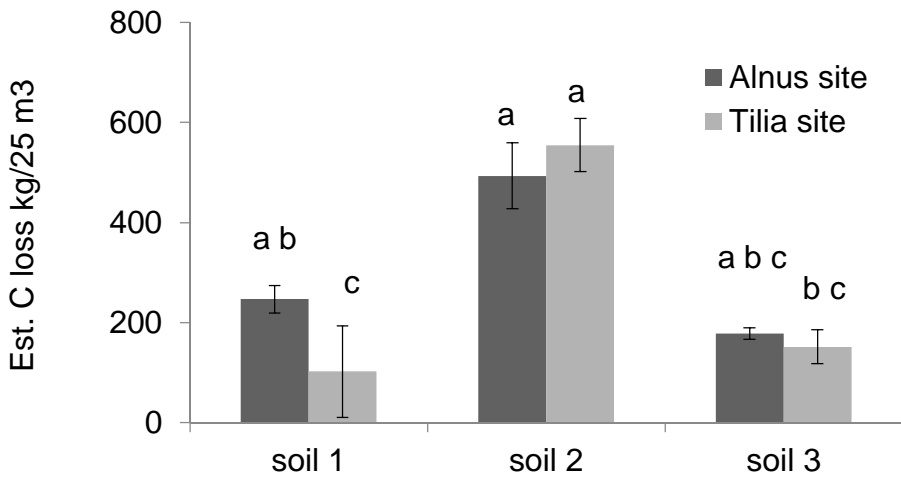
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793 Table 4. Biomass of the various tree compartments as measured (kg per tree  $\pm$  SD, n = 6—12) and predicted  
 794 from the DBH with biomass equations (average of the trees included in the biomass measurements) for each  
 795 taxon in 2011. The number and letter in superscript indicate the equation used (Appendix). ABW =  
 796 aboveground woody biomass, AB = aboveground biomass, including leaves. \* denotes that the biomass  
 797 model prediction is significantly different from the measurement results (Tukey's 2-sided t-test).

	Stem	Branch	Leaf	ABW	AB
<i>Tilia</i> meas.	18.6 $\pm$ 9.1	18.2 $\pm$ 6.2	4.5 $\pm$ 1.0	36.8 $\pm$ 14.9	41.3 $\pm$ 15.3
<i>Tilia</i> pred.	36.1 $\pm$ 13.5* <sup>3d + 3e</sup>	7.5 $\pm$ 3.2* <sup>3d</sup>	1.4 $\pm$ 0.5* <sup>3b</sup>	43.2 $\pm$ 17.0 <sup>1</sup> , 79.3 $\pm$ 26.4* <sup>4</sup>	47.8 $\pm$ 19.4 <sup>2</sup> 42.0 $\pm$ 15.9 <sup>3a</sup>
<i>Alnus</i> meas.	39.1 $\pm$ 6.3	28.0 $\pm$ 5.9	6.8 $\pm$ 1.3	67.1 $\pm$ 9.6	73.9 $\pm$ 10.8
<i>Alnus</i> pred.	61.0 $\pm$ 7.4* <sup>5c</sup> 69.3 $\pm$ 6.4* <sup>6d</sup> 60.7 $\pm$ 6.5* <sup>7d</sup>	12.8 $\pm$ 1.6* <sup>5b</sup> 5.3 $\pm$ 0.7* <sup>6c</sup> , 7.8 $\pm$ 1.3* <sup>7c</sup>	1.0 $\pm$ 0.1* <sup>6b</sup> 1.8 $\pm$ 0.1* <sup>7b</sup>	48.0 $\pm$ 5.6* <sup>5a</sup> 73.7 $\pm$ 8.9 <sup>5b+c</sup> 73.5 $\pm$ 7.3 <sup>6a-b</sup> 69.1 $\pm$ 7.8 <sup>7a-b</sup>	74.5 $\pm$ 7.4 <sup>6a</sup> 70.9 $\pm$ 8.0 <sup>7a</sup>

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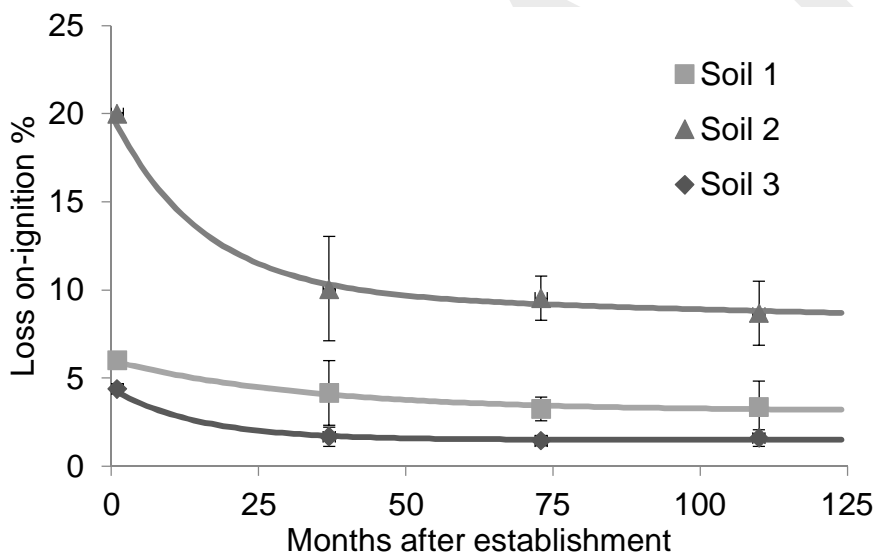
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801 Figure 1. Estimates of cumulative C loss ( $\pm$  SD) estimated per the 25 m<sup>3</sup> of the three different tested tree  
 802 soils from 2002 to 2011, based on the soil loss-on-ignition change. The mean values indicated with the same  
 803 letter do not differ significantly.

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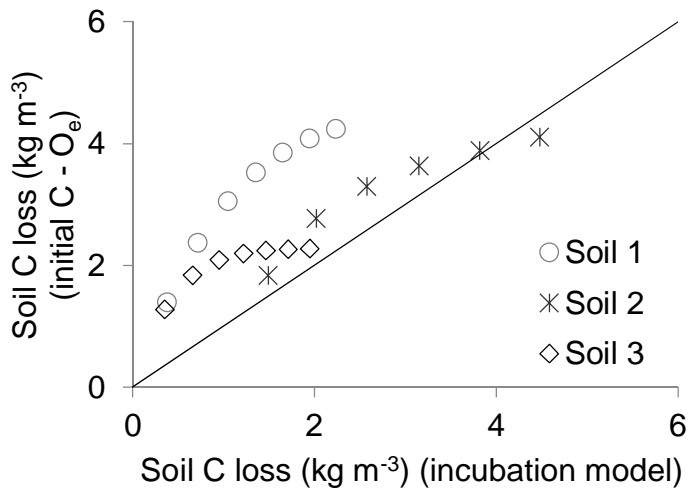
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808 Figure 2. Initial and measured average loss-on-ignition ( $\pm$  SD) for each soil mix fine soil fraction at each  
 809 sampling time (markers), and estimates for LOI between LOI sampling from the incubation model ( $O_e$ )  
 810 (lines, Eq. 6) for each month after establishment.

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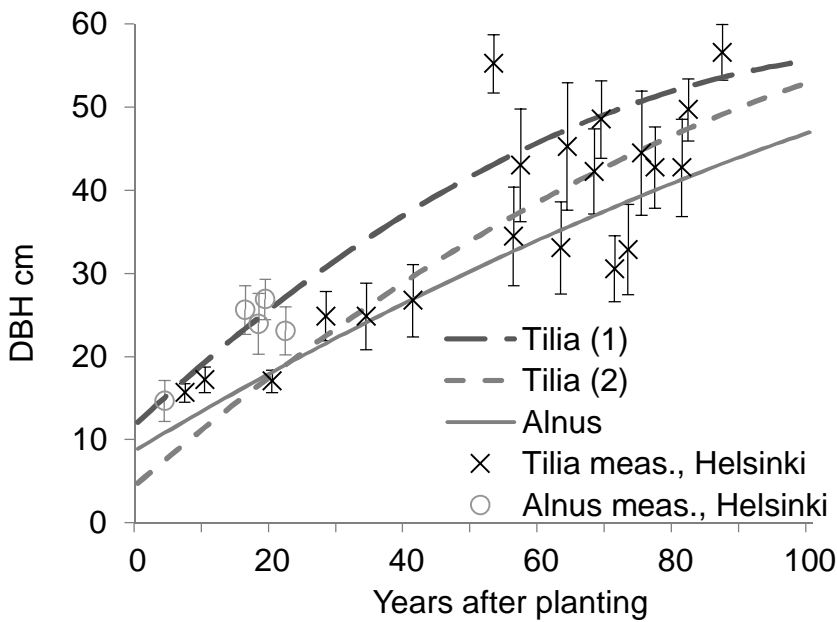


812

813 Figure 3. Cumulative C loss in 2006–2011, based on the  $O_e$  value (Eq. 6; Figure 2) on the X axis and the  
 814 incubation model on the Y axis, with early 2005 as the starting point.

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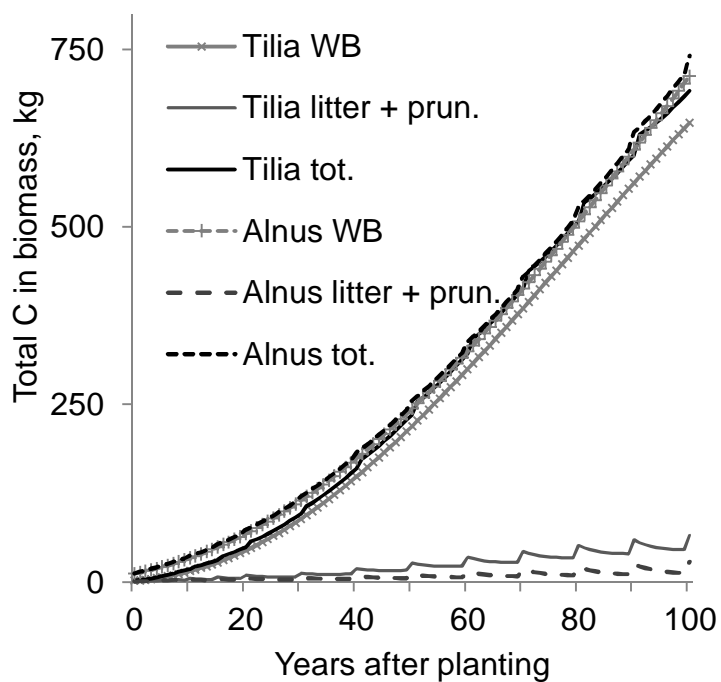
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818 Figure 6. Measured and predicted DBHs for *Tilia* and *Alnus*. The DBH predictions, two for *Tilia* and one for  
 819 *Alnus*, are based on literature (see Table 2). The crosses mark the average DBHs measured from the street  
 820 tree *Tilia* plantings in Helsinki ( $\pm$  standard deviation, SD) plotted against the number of years after planting.  
 821 The average DBH ( $\pm$  SD) of the two known older *Alnus glutinosa* f. *pyramidalis* plantings in Helsinki, for  
 822 one at two and for the other at three available time points, are marked with circles. Average predicted DBH  
 823 growth rates at 0-20 years, 0.65, 0.69, and 0.46  $\text{cm yr}^{-1}$ , at 20-40 years, 0.57, 0.56 and 0.42  $\text{cm yr}^{-1}$ , and >40  
 824 years, 0.40, 0.31 and 0.34  $\text{cm yr}^{-1}$  for *Tilia* predictions 1 and 2, and *Alnus* prediction, respectively.

825



826  
 827 Figure 7. Predictions for tree C sequestration in woody biomass (WB, including roots), litter and pruned  
 828 branches (litter + prun.) and the sum of the previous (tot.) at 0—100 years after planting. The *Tilia* C stock  
 829 was calculated based on DBH prediction 2 (Figure 6).

830  
 831

832 **Appendix.**

833 Parameters and references for the allometric equations (BEs) for the total AB and branch, trunk and leaf  
 834 biomass used in the calculations in this study. Equations 1--4 are for *Tilia* sp. and 5--7 for *Alnus glutinosa*.  
 835 The DBH ranges for which the equation was developed are listed. The equation form is  $M = aD^b$ , where M =  
 836 biomass (kg) and D = DBH, unless noted otherwise.

	Allom. equation no.	Parameter <i>a</i>	parameter <i>b</i>	Biomass compartment	DBH range and unit	Reference
	1	-5.49	2.45	woody abovegr.*	3--15 cm	Bunce, 1968†
	2	0.062	2.53	total abovegr.	5--50 cm	Brenneman, 1978, ref. Ter- Mikaelian and Korzukhin, 1997 ††
<i>T</i> <i>il</i> <i>i</i> <i>a</i>	3a	0.087	2.35	total abovegr.		
	3b	$4.90 \times 10^{-3}$	2.09	Leaves ‡		
	3c	$6.59 \times 10^{-3}$	2.68	branches	4--47 cm	Perala and Alban, 1994††
	3d	$4.99 \times 10^{-2}$	2.40	stem wood		
	3e	$4.32 \times 10^{-2}$	2.03	stem bark		
	4	$9.40 \times 10^{-2}$	2.04	woody abovegr.**	n/a, cm	McHale <i>et al.</i> , 2009†
	5a	$8.60 \times 10^{-2}$	2.35	woody abovegr.		
	5b	$1.47 \times 10^{-2}$	2.52	branches	n/a, cm	Hughes, 1971
	5c	$8.42 \times 10^{-2}$	2.45	stem		
	6a	$3.09 \times 10^{-3}$	2.02	total abovegr.		
<i>A</i> <i>l</i> <i>n</i> <i>u</i> <i>s</i>	6b	$3 \times 10^{-6}$	2.55	leaves	120--280 mm	Johansson, 1999
	6c	$3 \times 10^{-6}$	2.88	branches		
	6d	$5.61 \times 10^{-3}$	1.89	stem		
		7a	$7.90 \times 10^{-4}$	2.29	total abovegr.	
	7b	$2.39 \times 10^{-3}$	1.33	leaves		
	7c	$6 \times 10^{-7}$	3.28	branches	20--170 mm	Johansson, 2000
	7d	$1.19 \times 10^{-3}$	2.17	stem		

837  
 838 \*  $\ln M = a + b (\ln G)$ , G = girth (cm)  
 839 \*\* predicts volume; converted to mass with specific gravity of 0.40 (Alden, 1995).  
 840 † for *Tilia cordata*  
 841 †† for *Tilia americana*  
 842 ‡ Correction factor 1.13