

1 **Effect of grassland cutting frequency on soil carbon storage**
2 **- A case study on public lawns in three Swedish cities**

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12 **Abstract**

13 Soils contain the largest terrestrial carbon pool and thus play a crucial role in the global carbon
14 cycle. Grassland soils have particularly high soil organic carbon (SOC) stocks. In Europe (EU
15 25), grasslands cover 22% of the land area. It is therefore important to understand the effects of
16 grassland management and management intensity on SOC storage. City lawns constitute a
17 unique study system in this context, since they provide a high functional diversity and thus a
18 wide range of different management intensities per unit area. In this study we investigated
19 frequently mown (on average 8 times per season) utility lawns and rarely mown (once per
20 season) meadow-like lawns at three multi-family housing areas in each of three Swedish cities,
21 Uppsala, Malmö and Gothenburg. The two different lawn types were compared regarding their
22 aboveground net primary production (NPP) and SOC storage. In addition, root biomass was
23 determined in Uppsala. We found significantly higher aboveground NPP and SOC
24 concentrations and significantly lower soil C:N ratio for the utility lawns compared with the
25 meadow-like lawns. On average, aboveground NPP was 24% or 0.7 Mg C ha⁻¹ yr⁻¹ higher and
26 SOC was 12% or 7.8 Mg ha⁻¹ higher. Differences in SOC were well explained by differences in
27 aboveground NPP (R²=0.39), which indicates that the increase in productivity due to more
28 optimum CO₂-assimilating leaf area, leading to higher carbon input to the soil, was the major

29 driver for soil carbon sequestration. Differences in soil C:N ratio indicated a more closed N cycle
30 in utility lawns, which might have additionally affected SOC dynamics. We did not find any
31 difference in root biomass between the two management regimes, and concluded that cutting
32 frequency most likely only exerts an effect on SOC when cuttings are left on the surface.

33

34 **1 Introduction**

35 Soils contain the largest terrestrial carbon pool (Chapin et al., 2009). The balance of soil organic
36 carbon (SOC) inputs and outputs is therefore critical for the global carbon balance and thus for
37 the concentration of greenhouse gases in the atmosphere. Globally, 3650 Mha or 68% of the total
38 agricultural area is under permanent pasture or meadows (Leifeld et al., 2015). In Europe (EU-
39 25), grassland covers 22% of the land area (Soussana et al., 2007). Grassland soils store among
40 the highest amounts of SOC, which is primarily related to the high belowground carbon input by
41 roots and their exudates (Bolinder et al., 2012). Soils rich in SOC are potential hot-spots for CO₂
42 emissions, when a management or land-use change-induced imbalance in carbon input and
43 output occurs. It is therefore important to understand the effect of management practices on
44 grassland SOC storage. It has been demonstrated that the type, frequency and intensity of net
45 primary production (NPP) appropriation (harvest) can play a crucial role for the carbon balance
46 and SOC stocks of grassland ecosystems (Soussana et al., 2007).

47 One direct management intensity effect on SOC, which is mediated by grazing, cutting or
48 fertilisation regime, is obviously the change in carbon input via the degree of biomass extraction
49 and altered photosynthetic activity (Wohlfahrt et al., 2008a). Furthermore, above- and below-
50 ground allocation patterns may change with cutting frequency (Seiger and Merchant, 1997).
51 Recently, Leifeld et al. (2015) reported faster root turnover in moderately and intensively
52 managed alpine grasslands than at less intensively grazed sites. They concluded that
53 management is a key driver for SOC dynamics and should be included in future predictions of
54 SOC stocks. Nutrient status, species composition and diversity are highly management-
55 dependent and interfere with the carbon cycle in several ways, including effects on the
56 decomposer community and its substrate use efficiency (Ammann et al., 2007; Kowalchuk et al.,
57 2002).

58 Management effects on SOC are presumably smaller than land use change effects such as
59 conversion from permanent pasture to arable land (Poeplau and Don, 2013) and might not be
60 visible in the short term. To assess those changes, it is therefore important to find suitable study
61 systems with long-lasting strong contrasts in management intensity over a limited spatial scale
62 and with limited soil variability. For agroecosystems, this situation is usually created in long-
63 term experiments, which are designed to study such questions. In a global compilation of all
64 existing agricultural long-term field experiments, only 49 out of >600 experiments are listed as
65 including permanent grassland (pasture or meadow) (Debreceeni and Körschens, 2003). Thus,
66 the current quantitative and mechanistic understanding of grassland management effects on SOC
67 stocks is certainly limited, since existing studies are often strongly confounded by external
68 factors such as elevation gradients (Leifeld et al., 2015; Zeeman et al., 2010). As an alternative to
69 long-term plot experiments, urban areas can be appropriate study systems. Lawns, public green
70 areas and parks are omnipresent in urban areas and are usually managed in a similar way for a
71 long time, so that depending on the prior land-use type equilibrium SOC stocks might be
72 approximated (Raciti et al., 2011). Over a comparatively small spatial scale, a wide range of
73 different management intensities can be present.

74 Urban areas are more rapidly expanding than any other land-use type (Edmondson et al., 2014).
75 Turfgrass lawns cover the majority of all green open spaces in urban landscapes (70-75%)
76 according to Ignatieva et al. (2015). It has been estimated that turfgrass lawns cover
77 approximately 16 M ha of the total US land area, which in the 1990s was three-fold the area of
78 irrigated maize (Milesi et al., 2005; Qian et al., 2010). Although robust global estimates of the
79 coverage of turfgrass lawns are scarce, these few existing figures indicate the potential
80 importance of lawn management for the global carbon cycle. Furthermore, several studies
81 reported higher SOC stocks under urban land-use as compared to surrounding soils, which might
82 be a feature of high management intensity in urban ecosystems (Edmondson et al., 2012; Pouyat
83 et al., 2009). There is thus a need to quantify the carbon footprint of differently managed lawns,
84 for which SOC is of major importance. In the transdisciplinary Swedish LAWN project
85 (<http://www.slu.se/lawn>), lawns were studied from social, ecological and aesthetic perspectives
86 (Ignatieva et al., 2015).

87 In this study, as part of the LAWN project, we analysed two types of lawn under different
88 management intensity (cutting frequency) associated with multi-family housing areas, which
89 were intensively monitored at three sites in each of three Swedish cities. The objectives of the
90 study were: i) to examine how cutting frequency affected NPP, SOC and soil carbon to nitrogen
91 ratio (C:N), and ii) to disclose involved mechanisms causing differences in SOC between the two
92 management regimes.

93

94

95 **2 Materials and Methods**

96 **2.1 Study sites**

97 Public lawns in multi-family housing areas were investigated in three different cities,
98 Gothenburg, Malmö and Uppsala, and at three different sites in each city (Table 1). The nine
99 selected multi-family housing areas were established at approximately the same time during the
100 early 1950s. At each site, triplicate plots of two different lawn types were studied: utility lawn
101 and meadow-like lawn, with each plot comprising one complete lawn. The utility lawn was kept
102 short during the year and was mown on average every 18 days within the mowing period (8
103 times), which approximately corresponds with the growing period (May to mid-October in
104 Uppsala, April to late October in Gothenburg and Malmö). The meadow-like lawns were only
105 cut once, or twice in the single case of one lawn in Uppsala (Tuna Backar). Grass cuttings were
106 left on the surface on both lawn types. None of the lawns received any fertiliser. Grass species
107 composition did not differ greatly between the cities, with about 5-10 different grass species
108 abundant in utility lawns and meadow-like lawns. Utility lawns consisted of sparser, low-
109 growing species such as *Poa annua*, *Agrostis capillaris*, *Lolium* spp. and *Festuca rubra*, while
110 the most abundant grass species in meadow-like lawns were *Phleum pratense*, *Alopecurus*
111 *pratensis* and *Arrhenatherum* spp. (J. Wissman, pers. comm. 2015). The size of the individual
112 lawns was highly unequal with a range of 0.05-2.5 ha due to the heterogeneity of urban
113 landscapes. To obtain representative average values for the whole individual lawn, we conducted
114 all samplings described below adjusted to the size of the lawn, instead of using a “fixed grid”.

115 2.2 Estimation of aboveground net primary production and root biomass

116 Aboveground NPP in the utility lawns was estimated by repeated sampling of aboveground
117 biomass after the first mowing in spring by the local authority. Sampling was conducted on
118 average 12 ± 6 days after each mowing event. For the meadow-like lawns, biomass was collected
119 on several occasions even before the mowing to determine total growth at that specific time.
120 After the first cut, meadow-like lawns were treated as utility lawns. The plots were sampled at
121 four locations using a 50 cm x 50 cm square frame. Sampling locations were selected to be
122 representative of the total lawn area, so therefore sampling under trees or in proximity to other
123 vegetation was avoided. Repeated sampling was not conducted on the identical sampling
124 locations. The harvested biomass was dried at 70°C , weighed and multiplied by 4 to obtain the
125 biomass for 1 m^2 . The mean of the four replicates was divided by the number of days between
126 the last cutting and sampling to obtain daily growth rate. This growth rate was extrapolated to
127 cover all days between previous sampling and next mowing for which no growth rate was
128 determined. On average, this period accounted for 7 ± 6 days after each cutting event, and thus
129 data coverage (time for which the actual growth rate was measured) was more than $82 \pm 6\%$ for
130 the period between 1 January and the last sampling date, which was on average on 5 October ± 7
131 days. On the basis of these daily growth rates, we calculated cumulative growth until the last
132 sampling. Since this day varied slightly between plots and sites, we fitted a simple vegetation
133 model based solely on the plant response to air temperature, as developed by Yan and Hunt
134 (1999) to each growth curve in order to determine the regrowth after the last sampling until the
135 end of the vegetation period. The original equation is:

$$136 \quad r = R_{max} \left(\frac{T_{max}-T}{T_{max}-T_{opt}} \right) \left(\frac{T}{T_{opt}} \right)^{\frac{T_{opt}}{T_{max}-T_{opt}}}, \quad \text{Eq. 1}$$

137 where r is the daily rate of plant growth, T is the measured temperature at any day, T_{max} is the
138 maximum temperature (which was set to 30°C in this study), T_{opt} is the optimal temperature
139 (which was set to 25°C in this study) and R_{max} is the maximal growth rate at T_{opt} . Instead of using
140 R_{max} , which is used in eq. 1 to scale the temperature response function to actual observed
141 maximal plant growth at optimal temperature, we scaled the model by forcing the cumulated r
142 through the cumulated NPP value on the date of the last sampling, as illustrated in Figure 1 using
143 the example of the Björkekärr site in Gothenburg. The good fit indicates that: i) the growth

144 dynamics, and thus absolute growth, were well captured by the method used; and ii) the model
145 fits provide an unbiased and standardised extrapolation of aboveground NPP for the entire
146 growing period. Daily mean air temperature values for the closest weather stations of the
147 Swedish Meteorological Service (SMHI) to Malmö and Gothenburg were downloaded from
148 <http://www.smhi.se/klimatdata>. Daily average air temperature values for Uppsala were obtained
149 from the Ultuna climate station run by the Swedish University of Agricultural Sciences (SLU).

150 Root biomass was only determined once, and only in Uppsala. In each lawn, four cylindrical soil
151 cores of 7 cm diameter and 10 cm depth were taken at 0-10 cm soil depth. Aboveground plant
152 material was removed and soil cores were thoroughly rinsed and then put in a water bucket to
153 completely separate roots from soil. Roots were dried at 105°C weighed and analysed for carbon
154 (C) and nitrogen (N) content. Assuming a carbon content of 45% for plant biomass, we were able
155 to determine and subtract the adhering soil in the weighed root samples mathematically, as
156 described in Janzen et al. (2002).

157 **2.3 Soil sampling, analysis and SOC stock calculation**

158 Soils were sampled in autumn 2014 to a depth of 20 cm using an auger (2.2 cm diameter). In
159 each plot, 10 randomly distributed soil cores were taken and pooled to one composite sample.
160 Soils were dried at 40°C, sieved to 2 mm and visible roots were manually removed. Soil pH was
161 determined in water and samples with a pH value exceeding 6.7 were analysed for carbonates.
162 Soil texture was determined with the pipette method according to ISO 11277. As a slight
163 modification, wet sieving prior to sedimentation was done to 0.2 mm compared to 0.063 mm
164 prescribed in the ISO method. Total soil carbon and nitrogen were determined by dry
165 combustion of 1 g of soil using a LECO TruMac CN analyser (St. Joseph, MI, USA) and
166 carbonate carbon was determined using the same instrument after pretreatment overnight at
167 550°C. Organic soil carbon was calculated as the difference between total carbon and carbonate
168 carbon. Soil bulk density [g cm^{-3}] was determined by taking undisturbed cylindrical soil cores of
169 7 cm diameter and 10 cm height in an approximate soil depth of 5-15 cm, drying them at 105°C
170 and weighing them. Four samples were taken in each plot. To account for the fact that SOC
171 stocks under contrasting management regimes should be compared on the basis of equivalent soil
172 masses (Ellert and Bettany, 1995), we conducted a simple mass correction in which we first
173 calculated the soil mass (SM) [Mg ha^{-1}] of each plot using the equation:

174 $SM = BD \times D \times 100,$ Eq. 2

175 where BD is the soil bulk density [g cm^{-3}] and D is the sampling depth [cm]. The lower average
176 soil mass measured at each pair was then used as the reference soil mass (RSM) to which the
177 other treatment of each pair (three pairs per site) were adjusted.

178 SOC stocks [Mg ha^{-1}] were then calculated using the equation:

179 $SOC_{stock} = RSM \times \frac{C}{100}$ Eq. 3

180 where C is carbon concentration [%]. At one site in Gothenburg (Kyrkbyn), one pair of lawn
181 types had a large difference in soil texture, with 15% clay in the utility lawn and 30% in the
182 meadow-like lawn. The SOC concentration varied by a similar amount (2.46% compared with
183 4.58%), which was an outlying high difference when compared with that of all other pairs. We
184 attributed this to differences in soil texture and excluded this pair from the analysis. Apart from
185 slight differences in soil texture, the basic assumption was that the underlying pedology and
186 initial soil carbon stocks were similar for both lawn types, or at least not systematically biased.
187 Differences in soil texture between lawn types at each site was further not correlated to
188 differences in SOC concentration ($R^2=0.02$).

189 **2.4 Statistics**

190 All statistical analyses were performed with the R software version 3.1.2. We used linear mixed
191 effect models to analyse the effect of lawn management on aboveground NPP and SOC
192 concentration and stocks using the R Package nlme (Pinheiro et al., 2009). Management (utility
193 vs. meadow-like lawn) was used as the fixed effect, while city and site were used as nested
194 random effects (site nested in city). We used Tukey-type multiple comparison Post-Hoc analysis
195 (R Package multcomp) to test the management effect at each site for significance ($p<0.05$).
196 Average differences in SOC stocks between the different lawn types at each site (dependant
197 variable) were calculated and related to different explanatory variables (independent variables),
198 such as average clay content, differences in clay content between lawn types (absolute and
199 relative), soil pH, mean annual temperature (MAT), mean annual precipitation (MAP) and
200 differences in aboveground NPP. Generalised linear models with Gaussian error distribution
201 were used for multiple regression analysis. Model performance was evaluated using the Akaike

202 Information Criterion (AIC). The variable “clay content” had to be transformed to approximate
203 normal distribution. For both model approaches (mixed effect model and generalized linear
204 model) we used residual plots to study whether i) the regression function was linear, ii) the error
205 terms had constant variance, iii) the error terms were independent, iv) there were outliers or v)
206 the error terms were normally distributed. All values in the text and diagrams represent
207 mean±standard deviation.

208

209 **3 Results**

210 **3.1 Effect of lawn management on net primary production and soil carbon and nitrogen**

211 The intensively managed, i.e. frequently mown, utility lawns produced significantly ($p=0.003$)
212 more aboveground biomass (NPP) than the meadow-like lawns, which were cut only once a year
213 (Figure 2). At seven out of nine sites, NPP was higher in the utility lawns than in the meadow-
214 like lawns. The difference between the lawn types was most pronounced in Uppsala, where the
215 average NPP of the utility lawns (4.2 ± 0.9 Mg C ha⁻¹) was twice that of the meadow-like lawns
216 (2.1 ± 0.3). In contrast, two out of three sites in Gothenburg showed higher NPP on the meadow-
217 like lawns. Across all sites, the NPP of the utility lawns was 24% higher. Total root biomass, as
218 investigated at the three sites in Uppsala, was not significantly influenced by management
219 intensity and indicated a smaller ratio of belowground to aboveground NPP in meadow-like
220 lawns (Figure 3).

221 Concentrations of SOC were also positively affected by greater cutting frequency. Utility lawns
222 had significantly higher ($p=0.01$) SOC concentration than meadow-like lawns (Figure 4). Again,
223 the difference between the two lawn types was most pronounced in Uppsala, with an average
224 SOC concentration of $3.9\pm 0.6\%$ in the utility lawns and $2.9\pm 0.9\%$ in the meadow-like lawns. In
225 both Malmö and Gothenburg, we found one site with higher average SOC concentration in the
226 meadow-like lawns. The calculated SOC stocks are listed in Table 2. The average SOC stock
227 difference between the two differently managed lawn types was 7.8 Mg ha⁻¹ or 12%. The very
228 similar patterns observed for the variables NPP and SOC suggest that the SOC changes were
229 driven by NPP and thus carbon input. In fact, the difference in SOC stock between management
230 regimes at each site was significantly correlated to the difference in NPP (Figure 5). No other

231 parameter did add significant explanation to the model fit. Although clay content did not
232 improve the model fit of the generalized linear model, difference in SOC stock did also increase
233 with average clay content ($R^2=0.26$, $p=0.1$, not shown). This correlation is however strongly
234 driven by the sites in Uppsala, which showed the highest increase in both NPP and SOC. Thus,
235 also local management differences, which are however not available in detail, might have
236 influenced the observed treatment effect on SOC to some degree.

237 The soil C:N ratio of the meadow-like lawns (13.2 ± 1.2) was significantly higher ($p=0.007$) than
238 that of the utility lawns (12.6 ± 0.7), indicating that the soil organic matter under the utility lawns
239 was relatively enriched in nitrogen (Figure 6).

240

241 **4 Discussion**

242 **4.1 Effect of cutting frequency on aboveground productivity**

243 We showed that cutting frequency significantly altered the aboveground biomass production in
244 urban lawns. This can be explained by the fact that canopy CO_2 assimilation is a function of the
245 amount of assimilating plant matter (Wohlfahrt et al., 2008b). Wohlfahrt et al. (2008a) showed
246 that when the green area index (GAI) of an alpine grassland exceeded $4 \text{ m}^2 \text{ m}^{-2}$, the gross
247 primary production (GPP) decreased due to shading, but also due to plant phenology. Directly
248 after cutting (three cuts per season), their grassland had a GAI of $0.5\text{-}2 \text{ m}^2 \text{ m}^{-2}$, while directly
249 before cutting it had a $\text{GAI} > 6 \text{ m}^2 \text{ m}^{-2}$. The meadow-like lawns in our study were only cut once,
250 which indicates that the period in which the GAI of the canopy exceeded the optimum for CO_2
251 assimilation was very long. In contrast, the GAI of the utility lawn remained relatively close to
252 the optimum throughout the entire growing period. Furthermore, Klimeš and Klimešová (2002)
253 found that frequent mowing promoted the dominance of efficiently regrowing plant species,
254 which might provide an additional explanation for the higher NPP in our utility lawns. Our
255 results are also in agreement with Kaye et al. (2005), who found five-fold higher aboveground
256 NPP in an urban lawn than in a short-grass steppe. However, the urban lawn in that study was
257 fertilised and irrigated, while the urban lawns in our study were not. In a long-term field
258 experiment on cutting frequency effects on grass yield, Kramberger et al. (2015) found the
259 lowest yield in plots with the highest cutting frequency (2-week intervals) and the highest yield

260 in plots with moderate to low cutting frequency (8- to 12week intervals). This is in contrast to
261 our results from Uppsala and Malmö, but in line with the results from Gothenburg, where we
262 found higher aboveground biomass in the meadow-like lawns. However, we are unable to
263 explain the much higher NPP of the meadow-like lawns in Kyrkbyn.

264 4.2 Effect of cutting frequency on soil organic carbon in relation to similar management 265 contrasts

266 The higher aboveground NPP in the utility lawns had a significant positive effect on soil carbon.
267 This was expected, since the clippings were not removed and were thus able to contribute
268 directly to soil organic matter formation. For this reason, the results of our study are not directly
269 applicable to mown grasslands or leys, which are usually harvested. The responses of SOC to
270 management intensity in those systems are not well studied, but studies performed to date show
271 mixed results ranging from no effect (Kramberger et al., 2015) to significantly positive effects of
272 high cutting frequency (Ammann et al., 2007). In the latter case, the difference in SOC stocks
273 between intensively and extensively managed grassland was attributed to differences in N
274 fertilisation, which caused N deficiency and thus N mining in the extensive grassland, leading to
275 stronger mineralisation of stable organic matter. The effects of grazing intensity on SOC are
276 much better studied than the effects of mowing intensity. Both positive (Reeder et al., 2004;
277 Smoliak et al., 1972) and negative (Abril and Bucher, 2001; Su et al., 2005) effects of low
278 compared with high grazing intensity on SOC have been reported. However, many of the studies
279 reporting negative effects of intensive grazing refer to overgrazing in semiarid areas, which is
280 associated with strongly reduced vegetation cover and soil erosion. The actual effects seem to be
281 context-specific, as found in a global meta-analysis conducted by McSherry and Ritchie (2013).
282 The found positive correlation of difference in SOC and average clay content across sites has to
283 be interpreted with caveats, since a clear causality is not given. It is realistic, that more of the C
284 input is stabilised in clay-rich soils (Poeplau et al., 2015b). However, this correlation did not
285 hold within the three sites at each city, which indicates that the found correlation of clay and
286 difference in SOC, as well as of clay and difference in NPP across all sites might as well
287 resemble a random city effect.

288 Overall, our findings and those of previous studies (Christopher and Lal, 2007; Poeplau et al.,
289 2015a) confirm that plant input driven by NPP is the major driver for SOC dynamics. Root

290 carbon input is recognised as being of major importance for building up soil organic matter,
291 since a higher fraction of root-derived carbon is stabilised in the soil than in aboveground plant
292 material (Kätterer et al., 2011). In temperate grasslands, up to 70% of the total NPP is allocated
293 to roots and their exudates (Bolinder et al., 2007). However, in the present study, management
294 intensity did not significantly influence root biomass, indicating that root production was
295 relatively favoured in the meadow-like lawns. A similar finding has been reported in a study
296 which found higher root biomass under diverse swards than under conventional, intensively
297 managed ryegrass-clover pastures (McNally et al., 2015). Altered root production could therefore
298 not explain observed differences in SOC stocks in our study. However, the informative value of
299 the obtained root data is certainly limited, since root biomass was only determined in one city, to
300 a depth of 10 cm and at one point in time. It can thus not be assumed that the measured root
301 biomass measured is representative for root growth throughout the season (Ziter and
302 MacDougall, 2013). Furthermore, potential management effects on the depth distribution of
303 belowground biomass cannot be inferred.

304 The proportion of aboveground plant material stabilised in the soil has been estimated to be 13%
305 in a Swedish long-term agricultural field experiment (Andrén and Kätterer, 1997). Similar
306 values, i.e. around 10%, have been reported in other studies (Lehtinen et al., 2014; Poeplau et al.,
307 2015b). It can be assumed that lawn clippings undergo slightly lower stabilisation than straw in
308 agricultural systems, due to the lack of mixing of residues with stabilising mineral soil particles
309 (Wiesmeier et al., 2014). The mean annual difference in SOC sequestration between the two
310 lawn types we studied was $120 \text{ kg C ha}^{-1}\text{yr}^{-1}$. Assuming a constant stabilisation rate of 10%
311 across all sites, the calculated difference in SOC sequestration due only to different amounts of
312 recycled clippings would have been $69 \text{ kg C ha}^{-1}\text{yr}^{-1}$, which is only slightly more than half the
313 observed difference. Several studies report accelerated root turnover in more intensively
314 managed grassland (Klumpp et al., 2009; Leifeld et al., 2015). However, accelerated root
315 turnover could result in either more or less root-derived SOC, depending on the effect on total
316 root growth and exudations throughout the year, which is difficult to investigate (Johnen and
317 Sauerbeck, 1977).

318 Interestingly, the soil C:N ratio was significantly lower in the utility lawns than in the meadow-
319 like lawns, although neither system was fertilised and both were equally exposed to N

320 deposition. Furthermore, the proportion of N-fixing leguminous plants was higher in the utility
321 lawns than in the meadow-like lawns only in Gothenburg. This might indicate that nitrogen
322 cycling was more closed in the utility lawns. Potentially, more nitrogen is lost via leaching in the
323 meadow-like lawns, because N mineralisation and plant N demand occur asynchronously
324 (Dahlin et al., 2005). The peak in N mineralisation usually occurs around midsummer (Paz-
325 Ferreira et al., 2012), which might be too late for plant uptake when the grass is not mown and
326 would lead to N losses from the system. Another pathway of N loss is ammonia (NH₃)
327 volatilisation, which increases in later development stages of the plant due to ontogenetic
328 changes in plant N metabolism (Morgan and Parton, 1989). Whitehead and Lockyer (1989)
329 showed 10% N losses from decomposing grass herbage by NH₃ volatilisation. The consequences
330 of N deficiency for SOC dynamics are twofold: i) decreased NPP and thus decreased carbon
331 input (Christopher and Lal, 2007) and ii) increased heterotrophic respiration due to N mining in
332 more recalcitrant organic matter (Ammann et al., 2007). In an incubation experiment, Kirkby et
333 al. (2014) showed that more aboveground residues were stabilised in the soil when nitrogen was
334 added. Thus, negative effects of lawn management on soil N storage can feed back onto SOC,
335 which might also explain a certain proportion of the observed differences in SOC.

336 4.3 Implications for urban soil management

337 During the past decade, several studies have investigated biogeochemical cycles in urban soils,
338 since their relevance for the global carbon cycle and as a fundamental ecological asset in an
339 urbanising world is becoming increasingly evident (Lehmann and Stahr, 2007; Lorenz and Lal,
340 2009). Compared with data on agricultural land with similarly textured soils in the surroundings
341 of the study sites extracted from a national soil inventory database, we found on average 55%
342 (utility lawns) and 35% (meadow-like lawns) higher SOC stocks in the lawns we investigated.
343 Furthermore, it has been found in several studies that urban soils have higher carbon stocks than
344 native soils in adjacent rural areas, which can be attributed in particular to more optimised, but
345 also resource-consuming, management, including fertilisation and irrigation (Edmondson et al.,
346 2012; Kaye et al., 2005; Pouyat et al., 2009). However, in the present study we were able to
347 show that SOC storage in urban lawns can be increased at comparatively low cost under
348 temperate climate conditions by optimising NPP and leaving residues on the lawn. Losses of
349 carbon and nutrients are thereby minimised. Milesi et al. (2005) used the BIOME-BGC model to

350 compare different lawn management scenarios and found that applying 73 kg N and recycling
351 the clippings was more efficient for SOC sequestration (+40%) than applying 146 kg N and
352 removing the clippings. For the sites in Uppsala, Wesström (2015) calculated that the
353 management of utility lawns creates 54 kg ha⁻¹ yr⁻¹ more C emissions than the management of
354 meadow-like lawns. Subtracting this value from the annual difference in SOC sequestration that
355 we found (120 kg C ha⁻¹ yr⁻¹), the utility lawns in our study sequester a non-significant amount of
356 66 kg ha⁻¹ yr⁻¹ more carbon than the meadow-like lawns. However, for a full greenhouse gas
357 budget, the effects of lawn management on other trace gases, primarily nitrous oxide (N₂O), have
358 to be considered (Townsend-Small and Czimczik, 2010). In that case management of the
359 clippings will most likely play a key role, since coverage of the soil with organic material
360 increases soil moisture and the availability of labile carbon but decreases soil oxygen, all of
361 which favour N₂O formation (Larsson et al., 1998; Petersen et al., 2011).

362

363 **5 Conclusions**

364 This investigation of urban lawns in three Swedish cities showed that cutting frequency alone
365 can exert a significant influence on soil carbon, mainly by increasing net primary production and
366 thus carbon inputs. However, this is most likely only true when cuttings are left on the lawn,
367 since belowground production did not show any differential response to cutting frequency.
368 Moreover, the observed difference in soil carbon could not be fully explained by the expected
369 stabilisation of aboveground-derived carbon input differences, which might denote that either
370 root-derived carbon dynamics or nitrogen mining also play an important role. If clippings are left
371 on the lawn, nitrous oxide emissions might comprise a significant fraction of the greenhouse gas
372 budget of lawns and have to be accounted for to judge the climate mitigation potential of
373 contrasting lawn or grassland management strategies.

374

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377 Sciences and Spatial Planning (contract 225-2012-1369).

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529 List of tables:

530 Table 1: Site characterisation with year of establishment, clay, silt and sand content [%], soil pH
 531 for utility lawns (U) and meadow-like lawns (M) and mean annual temperature [MAT, °C] and
 532 mean annual precipitation [MAP, mm] (1961-1990) for all three Swedish cities studied

City	Site	Age	MAT	MAP	C:N	Clay		Silt		Sand		pH	
						U	M	U	M	U	M	U	M
Uppsala	Eriksberg	1949	5.5	527	12.8	36	46	43	44	21	10	~6	~6
	Sala backe	1950			12.5	45	45	47	51	8	4	~6	~6
	Tuna backar	1951			13.1	33	23	47	45	20	32	~6	~6
Malmö	Kirseberg	1950	8.4	540	12.7	12	10	49	46	39	45	7.2	7.2
	Sibbarp	1953			13.8	15	15	48	47	38	38	7.4	7.8
	Augustenborg	1952			13.9	13	10	49	45	38	45	7.4	7.7
Gothenburg	Guldheden	1950	7.4	714	14.1	16	14	45	44	39	42	5.5	5.4
	Kyrkbyn	1955			12.0	16	22	62	55	21	23	5.8	5.7
	Björkekärr	1950			12.8	14	16	49	58	37	27	5.5	5.7

533

534 ^xyear only approximate. *pH values for the Uppsala sites were not measured, and the values
 535 shown are estimates based on typical values for soils in Uppsala (e.g. Kätterer et al., 2011)

536

537 Table 2: Soil bulk density (BD) [g cm⁻³] and SOC stocks [Mg ha⁻¹] according to equation 3.
 538 Standard deviation is given in brackets

City	Site	Utility lawn		Meadow-like lawn		Utility lawn		Meadow-like lawn	
		BD		BD		SOC stock		SOC stock	
Uppsala	Eriksberg	1.13	(0.04)	1.13	(0.16)	74.8	(11.4)	63.1	(8.7)
	Sala Backe	1.14	(0.03)	1.1	(0.07)	96.2	(9.3)	69.8	(24.2)
	Tuna Backar	1.15	(0.07)	1.21	(0.06)	72.4	(13.3)	47.6	(19.5)
Malmö	Kirseberg	1.03	(0.07)	1.02	(0.08)	69.4	(4.5)	52.7	(0.95)
	Sibbarp	1.04	(0.06)	0.98	(0.06)	75	(8.3)	96.4	(3.5)
	Augustenborg	1.03	(0.06)	1.18	(0.15)	59.1	(9.4)	50.3	(22.4)
Gothenburg	Guldhelden	0.87	(0.14)	0.88	(0.21)	86.2	(2.3)	78.4	(21.3)
	Kyrkbyn	0.99	(0.09)	0.88	(0.06)	68.2	(8.1)	77.9	(7.8)
	Björkekärr	0.96	(0.1)	0.99	(0.08)	67	(14.4)	61.2	(3.1)

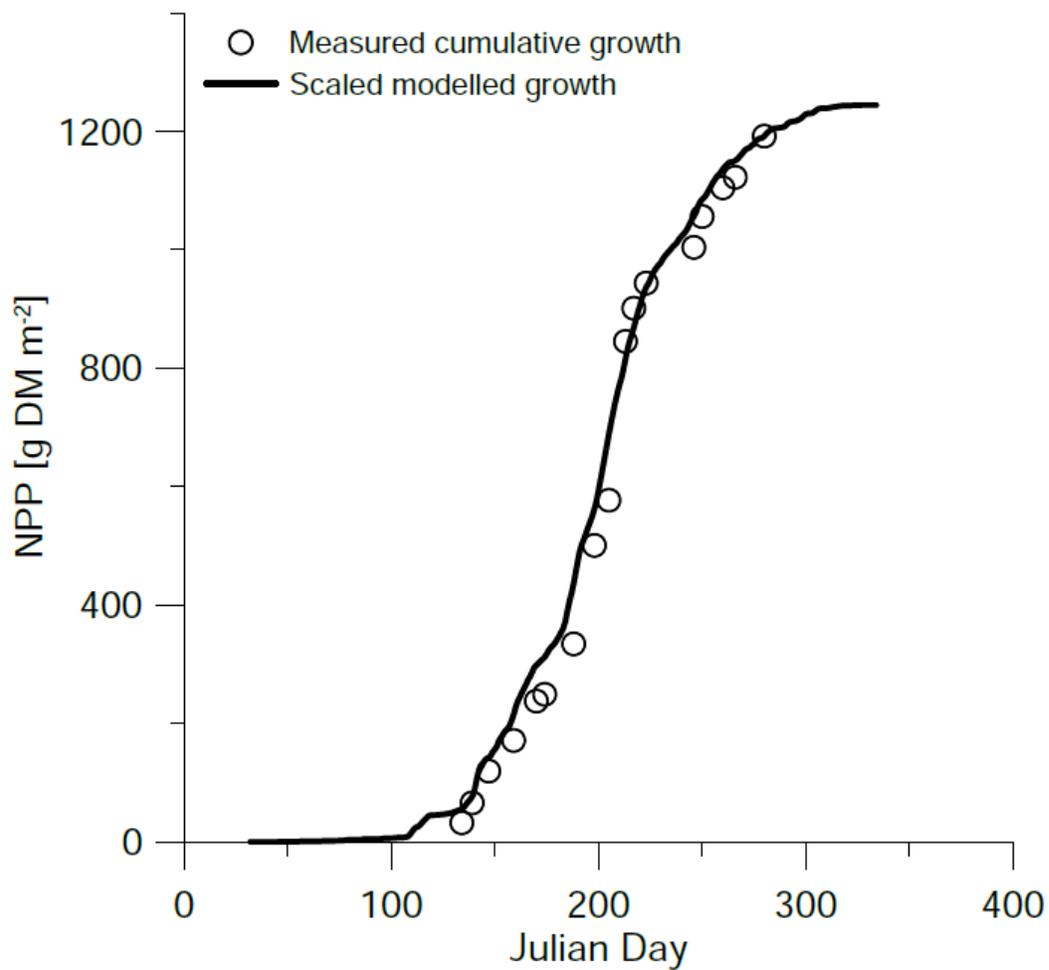
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541 Figure captions:

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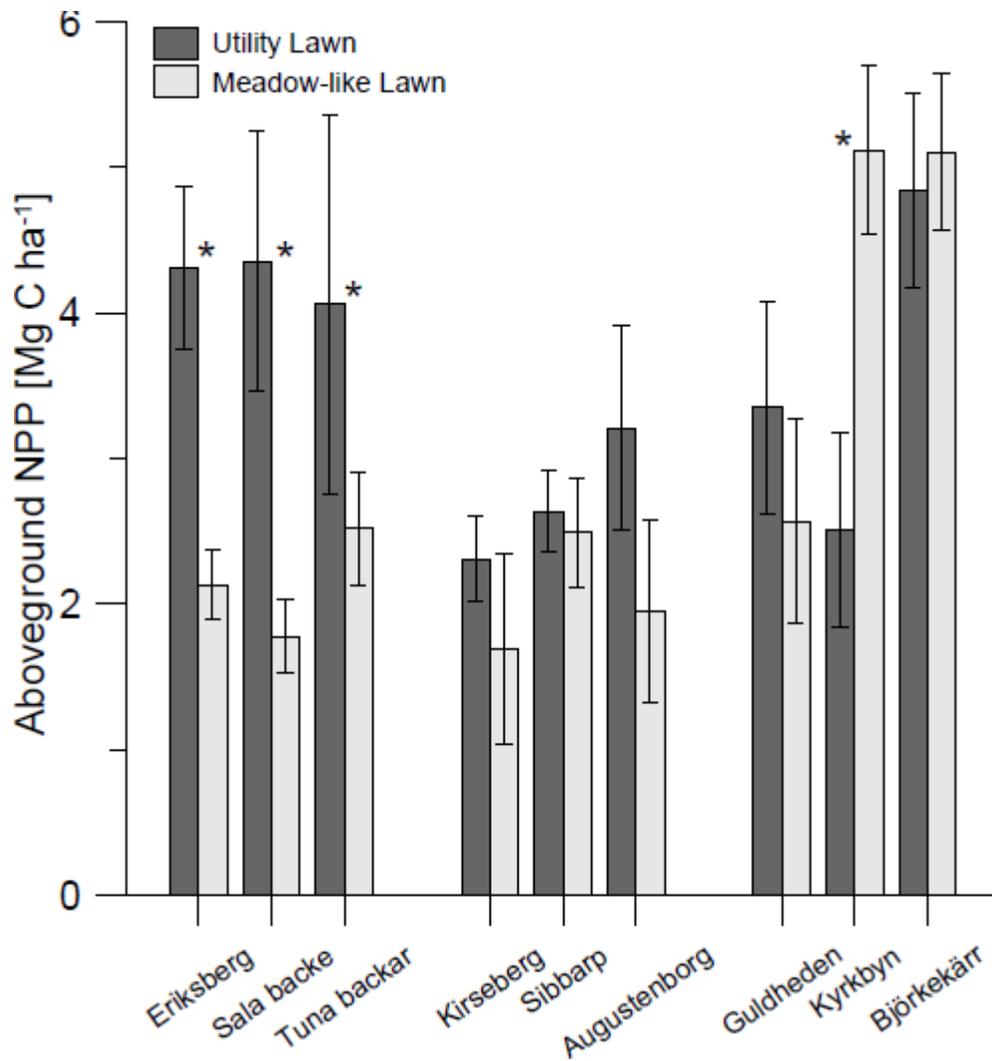
545 Figure 1: Example of the vegetation model (equation 1) fit to a calculated cumulative growth
546 curve for a utility lawn in Björkekärr, Gothenburg.

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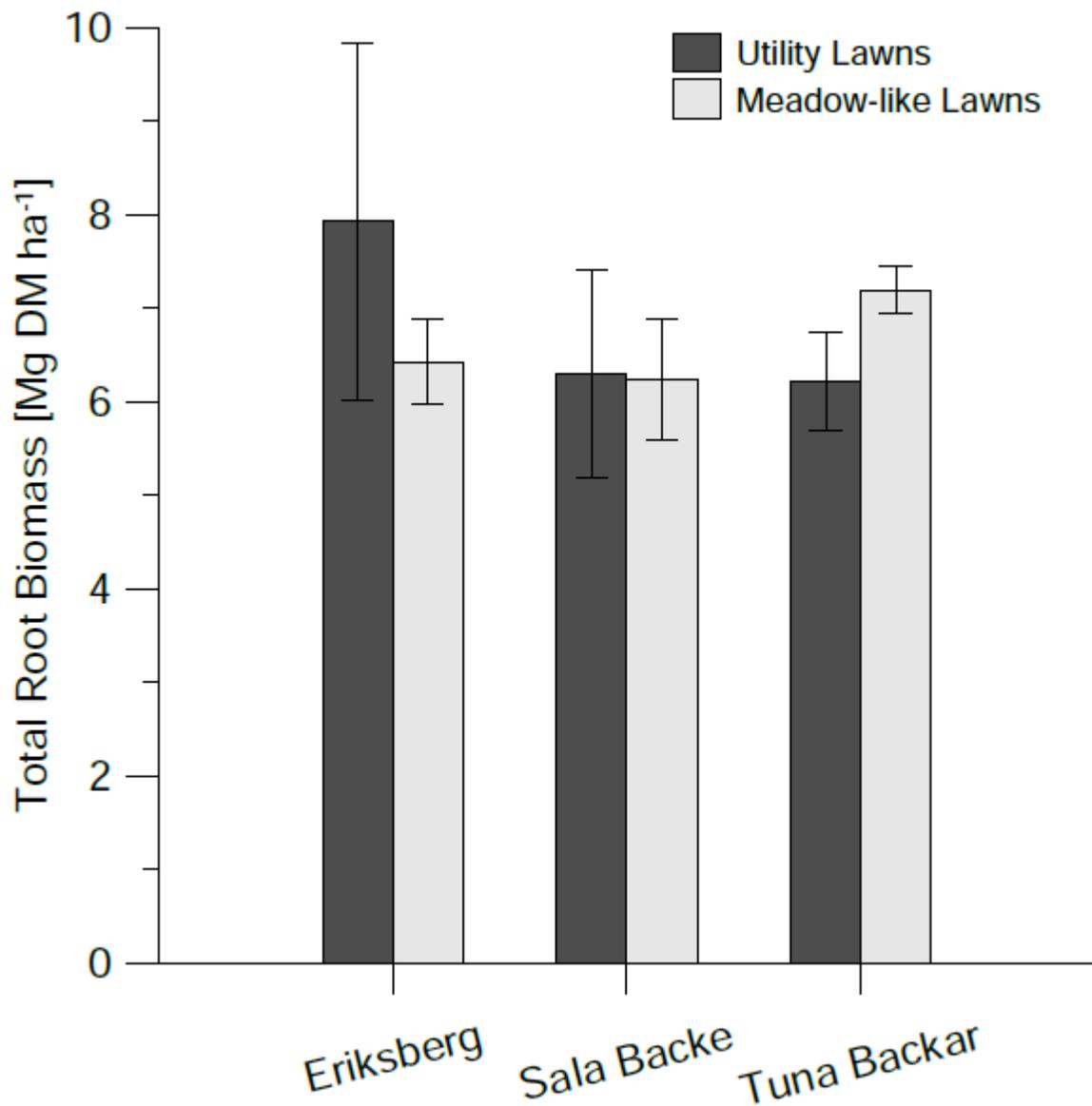
553 Figure 2: Bar plot showing estimated aboveground net primary production (NPP) of the two
 554 different lawn types at each site. Errors bars indicate standard deviation and stars indicate
 555 significant difference between treatments at the specific site ($p < 0.05$).

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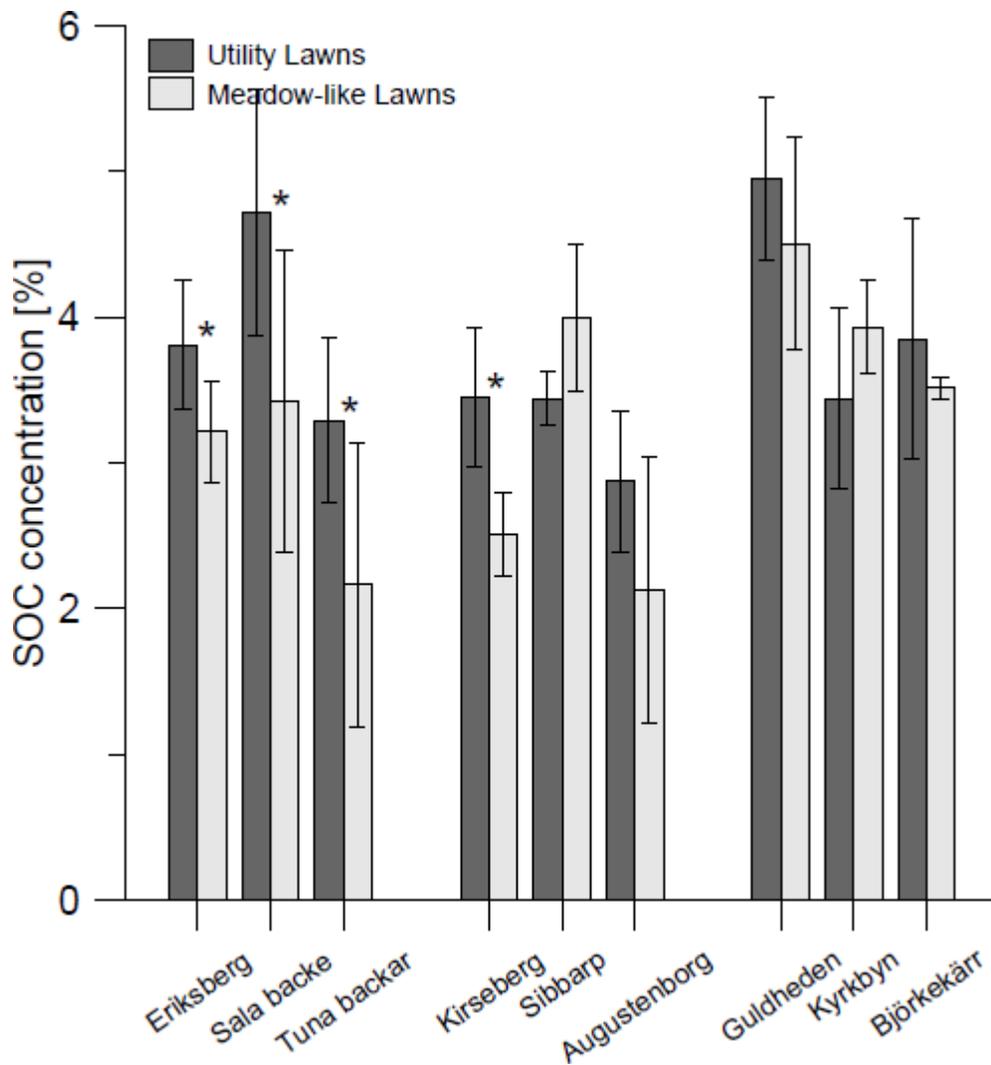
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561 Figure 3: Bar plot showing total root biomass at 0-10 cm depth for the two different lawn types
 562 at the study sites in Uppsala. Errors bars indicate standard deviation.

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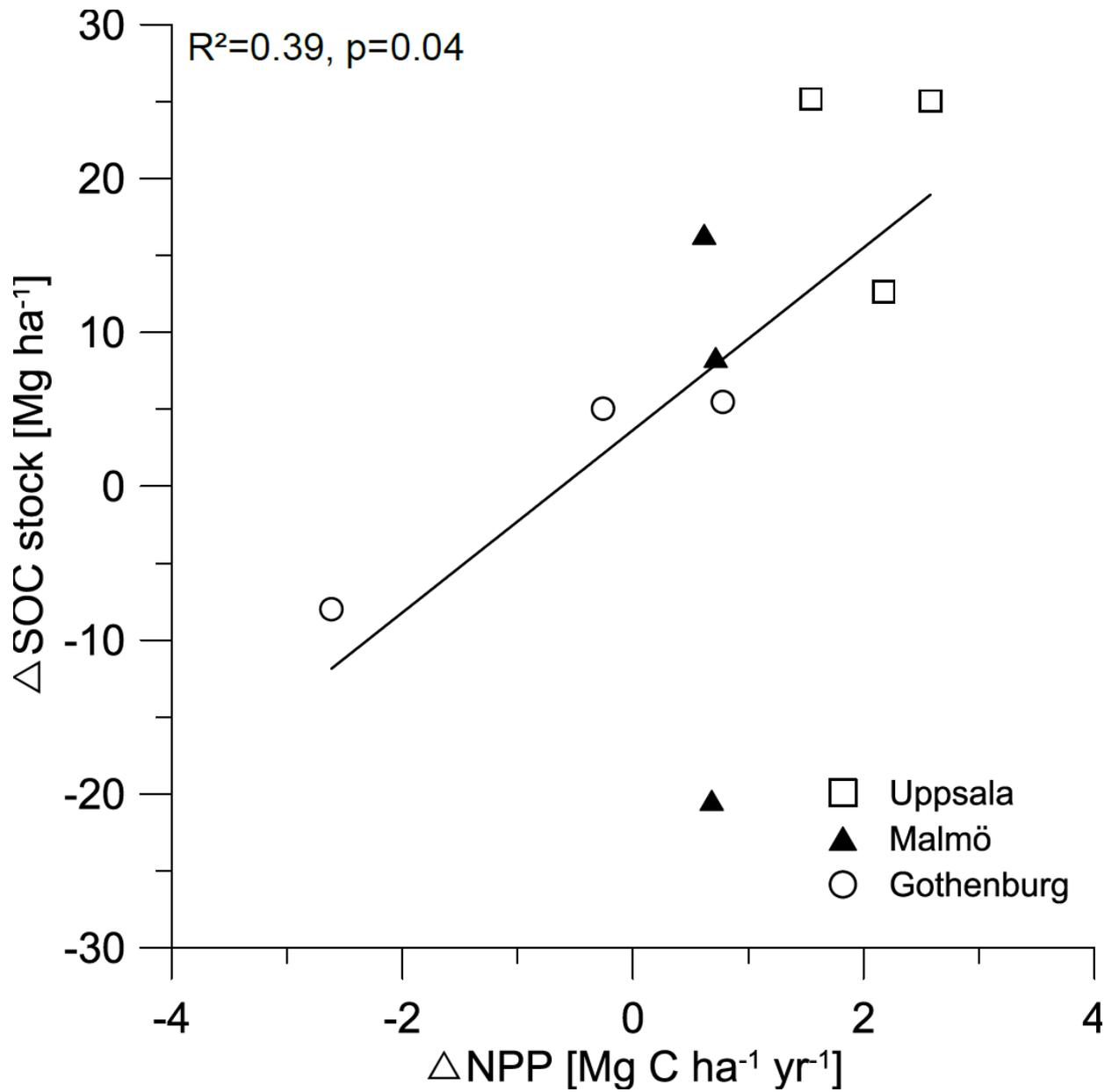
568 Figure 4: Bar plot showing measured soil organic carbon (SOC) concentration in the two
 569 different lawn types at each site. Error bars indicate standard deviation and stars indicate
 570 significant difference between treatments at the specific site ($p < 0.05$).

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577 Figure 5: Difference in soil organic carbon (SOC) stock between utility and meadow-like lawns
 578 as a function of difference in aboveground NPP for all sites.

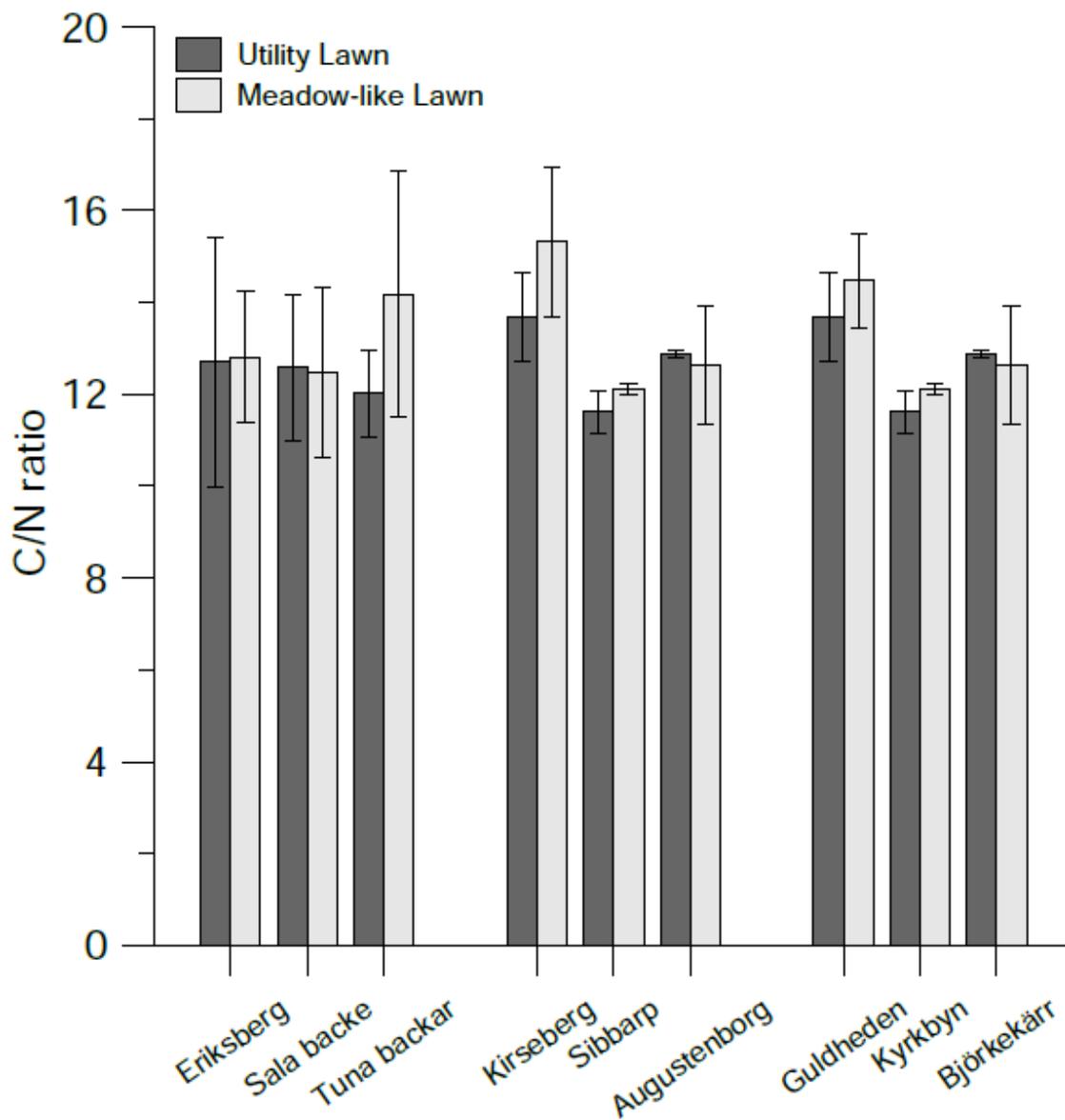
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586 Figure 6: Bar plot showing measured C:N ratio of the two different lawn types at each site. Error
587 bars indicate standard deviation.

