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SOME ECOSYSTEM SERVICE ASPECTS OF YOUNG STREET TREE PLANTINGS

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ACADEMIC DISSERTATION

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ABSTRACT

In urban forestry, the assessment and valuation of ecosystem services provided by urban trees are increasingly important both for the rationale of planting new trees and for retaining and managing existing tree populations. To support the field of practical urban forestry, research is needed on the net effects of ecosystem services and costs. The aim of this thesis was to analyse the ecosystem service potential of young street tree plantings. To this end, transplanting recovery, tree growth and carbon and water exchange were studied on two case study streets, one planted with *Tilia* × *vulgaris* Hayne and the other with *Alnus glutinosa* (L.) Gaertn. f. *pyramidalis* 'Sakari'. The relationships between tree growth, tree and soil water and carbon exchange, environmental variables and tree properties were examined.

Transplanting recovery of *Tilia* trees was delayed due to excess soil water, while *Alnus* trees recovered within the first few years. *Alnus* shoot growth responded positively and *Tilia* negatively to an increase in soil water content. Branch leaf area in relation to branch basal area varied, showing effects of transplanting and subsequent adaptation of the trees to the new growing sites. The studied trees accumulated carbon in their woody biomass during the first decade after transplanting, but the sequestration was small relative to carbon loss from the man-made tree soils. Several additional decades of tree growth were estimated to be needed to attain net carbon sequestration in these street tree plantings if peat originating C and/or renewable C lost from tree soils was counted as C loss. Biomass equations developed in traditional forests predicted total aboveground street tree biomass fairly well, but performed unsatisfactorily in estimating specific aboveground biomass compartments. The biomass distribution and litter production of street trees also require further study to gain insights into the role of tree litter in urban biogeochemical cycles.

The annual variation in tree water use of the studied trees was high, but within one year, a Penman-Monteith-based evapotranspiration model with added stomatal conductance and leaf area dynamics description, together with soil water status, explained the variation in tree transpiration quite well. Using a single parameterization over all four years examined did not produce reliable tree water use estimates however. Scaling tree transpiration to different canopy cover percentages implied that especially the columnar *Alnus* trees could transpire a considerable proportion of annual rainfall with attainable canopy cover, potentially contributing to stormwater management.

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LIST OF ORIGINAL PUBLICATIONS

The thesis is based on the following two articles and one manuscript.

I Riikonen, A., Lindén, L., Pulkkinen, M., Nikinmaa, E. 2011. Post-transplant crown allometry and shoot growth of two species of street trees. *Urban Forestry & Urban Greening* 10: 87-94. doi:10.1016/j.ufug.2010.09.001

II Riikonen, A., Pumpanen, J., Mäki, M., Nikinmaa, E. High carbon losses from established growing sites delay the carbon sequestration benefits of street tree plantings - a case study in Helsinki, Finland. Manuscript.

III Riikonen, A., Järvi, L., Nikinmaa, E. 2016. Environmental and crown related factors affecting street tree transpiration in Helsinki, Finland. Accepted for publication in *Urban Ecosystems*. doi: 10.1007/s11252-016-0561-1

The publications are referred to in the text by their roman numerals.

Author's contribution

Anu Riikonen was responsible for the summary of this thesis. The original study ideas and research site setup used in all papers originated from Eero Nikinmaa. Anu Riikonen was mainly responsible for preparing the manuscripts for papers I, II and III and for analysis and interpretation of the data, with help and ideas from co-authors. Anu Riikonen planned, supervised and participated in data collection.

ABBREVIATIONS

a.s.l.	Above sea level
BE	Biomass equation
C	Carbon
CO ₂	Carbon dioxide
<i>D</i>	Water vapour pressure deficit
DBH	Diameter at breast height
ET ₀	Reference evapotranspiration calculated with a Penman-Monteith -based model
<i>I</i>	Global radiation
<i>k_c</i>	Crop coefficient
LAI	Leaf area index
LOI	Loss-on-ignition
PCA	Projected canopy area (on horizontal plane)
PM model	Evapotranspiration model based on the Penman-Monteith equation
SOM	Soil organic matter
SWC	Soil water content
<i>T</i>	Temperature

1 INTRODUCTION

1.1 URBAN FORESTRY AND URBAN TREE RESEARCH – CHALLENGES AND APPROACHES

Although the origins of urban forestry date much further back, there was a notable increase in interest and research in the area in the 1970s and 1980s (Koninendijk et al. 2006). Early on, the field of ecophysiology (the study of how the physiology of an organism relates to its environment) of urban trees derived and generalized much from what was known of other ecosystems (Kozłowski and Davies 1975, Federer 1976, Patterson 1977, Roberts 1977, Kozłowski 1985a, b). Slowly, the critical areas where the assumptions most needed testing were identified with the aid of either surveys (Foster and Blaine 1978, Gibbs and Palmer 1994, Randrup 1997) or more commonly in case studies investigating ailing urban tree plantings (e.g. Ruark et al. 1983, Berrang et al. 1985, Messenger 1986).

Development in urban forestry research led from early descriptive, explorative, often qualitative approaches towards analytical, explanatory and quantitative work. Controlled experiments were still rarely seen in the 1980s, and in 1988, Whitlow and Bassuk somewhat despairingly condense the state of the knowledge on ecophysiology of urban trees: “At worst, then, the field of ecophysiology as applied to urban trees is anecdote and conventional wisdom. At best, it is a body of unquantified empirical observation, supported all too infrequently by rigorous investigation and experimentation. The synthesis, then, should not be mistaken for full understanding or as implying that we have quantified the range of urban plant stresses or identified solutions.” In the early years, Whitlow and Bassuk (1987) also expressed a concern about the danger of dogmatism – or rather confirmation bias – in the growing field of urban tree ecophysiology. They noted that while it is often stated that the urban environment is stressful for trees, there are in fact papers, often disregarded or misinterpreted, which give evidence either entirely or partially to the contrary (e.g. Kjelgren and Clark 1993, MacDonald et al. 2004, Watson and Kelsey 2006); the urban environment is not so very different from all and any natural environments.

In addition to Whitlow and Bassuk (1987), also Kozłowski (1985b) stresses early on that much depends on the exact definition and delimitation of terms such as “stressful” and “difficult” used to describe urban conditions. Whitlow and Bassuk (1987, 1988) point out that much could be gained by applying the knowledge in related fields – forestry, ecology and plant physiology – to urban trees by seeing their situation as not necessarily unique, but rather as part of the wide range of environmental conditions that bring about different physiological responses in trees. This opens up avenues for studying e.g. the effects of climate change based on urban trees (Sicard et

al. 2016); urban areas expose trees to many of the effects expected to increase with changing climate, such as changed hydrology and increased temperatures (Arnfield 2003) and invasive pest species (Tubby and Webber 2010, Tomlinson et al. 2015).

After the need for more rigorous research approaches was recognized in the late 1980s, experiments in real-life urban environments (e.g. Costello et al. 1991, Hodge 1991, Kjelgren and Clark 1993) and studies in controlled experimental field or laboratory conditions (e.g. Watson and Sydnor 1987, Watson et al. 1993, Kjelgren and Montague 1998) became more common in the 1990s. Case studies and surveys are still quite common as well and can be used as a basis for material collection for more detailed studies. The sampling of existing tree populations (surveys) can pinpoint likely problems, although due to the diversity of species and sites, sample sizes needed tend to be quite large (Sun and Bassuk 1991). No doubt this has contributed to the relative popularity of studies based on selected, information-rich populations, species or sites (e.g. Peters et al. 2010, Scharenbroch and Catania 2012, Peper et al. 2014, Sanders and Grabosky 2014), moving from pure surveys towards case studies. Case studies, both qualitative and quantitative, investigate the chosen subjects at depth; the relative importance and mechanisms of factors of interest can be examined. By moving towards more controlled conditions, the issues and causal relations found to be important can be confirmed and quantified more accurately.

Overall, studies in increasingly controlled conditions have allowed for a more quantitative and causal approach to urban tree ecophysiology, but one of the problems remaining in many areas, such as studying tree water use, has been the need to use containerized or relatively small trees (Kjelgren and Montague 1998, Hagishima et al. 2007, Bartens et al. 2009). While environmental conditions cannot be controlled in an on-site research setup, careful monitoring of the environment allows for controlled research on large trees planted on-site. Case-based studies have many benefits in urban tree research, but also the inherent problem of generalization of findings to conditions outside the researched case. Case studies can facilitate understanding of cause and consequence, mechanisms and interactions; however, the generalization should be based on analysis rather than on statistical evidence (Yin 2003).

The bulk of existing studies regarding urban trees and their environment contains research mainly from two separate viewpoints; on one hand, the effect of the urban environment on tree growth and well-being – and solutions that bring about improved urban tree performance – has been investigated early on, but many interactions and species-level differences remain poorly understood. In more recent decades, the second viewpoint, the positive effect of trees from a utilitarian human perspective, i.e. ecosystem services (benefits provided to humans by ecosystem functions, Millennium Assessment, 2003) related to urban trees, has gained more attention (Koninendijk et al. 2006). Ecosystem ecology as a defined field is not much

older than the study of ecosystem services. The decisive role of biota in circulation of elements in the ecosystems in general became commonly accepted as late as in the latter half of the last century (Gorham 1991). Ecosystem ecology has developed to integrate biogeochemistry with the idea of a food chain in which energy and matter travel through the biota of the ecosystem. In urban forestry, understanding the controls and interactions of trees and ecosystem processes is an obvious step forward because it allows for an easy integration of the ecosystem service perspective.

The approaches originating from ecosystem ecology are just beginning to emerge in urban forestry. These promote the integration of the two previously presented approaches in urban tree research, the study of tree growth in response to its environment, and the ecosystem services derived from trees. This brings forth an explicit understanding that to gain the expected ecosystem services from tree plantings, certain conditions regarding e.g. site conditions and maintenance must be met (Hale et al. 2015). This can be achieved by management and maintenance inputs or by promoting tree resource acquisition from biogeochemical cycles. For example, available water resources affect tree growth; in a study of Danish street trees, more water led to larger trees (Bühler et al. 2006). A larger tree is more efficient in intercepting rainfall via a larger canopy and higher leaf area than a smaller tree (Xiao and McPherson 2011, 2016). Unless intensive tree management is a realistic option, the source of water for the trees should be the natural water cycle, either rainfall or runoff from another area. A conclusion can thus be drawn that to provide ecosystem services, trees need some ecosystem services to be available to begin with – in this case, trees need the water cycle to provide water-related ecosystem service.

As a parallel but less obvious example, soil organic matter is usually considered beneficial for soil properties (e.g. Reeves 1997, Layman et al. 2016), and better quality of soil resources can be expected to produce faster tree growth. Soil organic carbon content must be maintained by continuous C input, unless soil conditions are adverse to decomposition, e.g. waterlogged or very cold (Jenkinson and Rayner 1977, Oades 1988). Continuous litter input and turnover would maintain soil C content and act as a source of nutrients, which in turn could lead to improved carbon (C) sequestration by the trees via faster biomass accumulation. In effect, improved carbon cycle could lead to improved C sequestration of the tree planting. To put it simply, it appears that the better the initial state of ecosystem services, the more trees can add to them; a positive feedback loop is a possibility that should be used to advantage. This interaction between the tree, its environment and the ecosystem services produced can be integrated by considering urban ecosystem ecology – the pools and fluxes of energy and matter that meet and interact within the urban ecosystem.

1.2 ECOSYSTEM SERVICES, DISSERVICES AND COSTS OF URBAN TREES

The ecosystem services concept is derived from cost-benefit approaches (Westman 1977), mainly in the context of environmental protection (Ehrlich and Mooney 1983, Mooney and Ehrlich 1997). It is a fairly recent coinage (Ehrlich and Mooney 1983) and has served to highlight the benefits that humans receive from ecosystem functions. Ecosystem services were classified in the Millennium Ecosystem Assessment (Millennium Assessment, 2003) into four categories: provisioning, regulating, cultural and supporting. The regulating services category is the one most obvious and most commonly assessed and valued in the case of urban greening (e.g. McPherson and Simpson 2002, McPherson et al. 2005). The monetary valuation of ecosystem services has given rise to policies promoting the value and interest in urban greening (e.g. Pincetl 2010, Pincetl et al. 2013, Churkina et al. 2015). It has been noted, however, that not all ecosystem services accredited to urban greening are well founded in research, and the costs (Dwyer et al. 1992) and disservices (defined as various nuisances and losses produced by ecosystem functions, Lyytimäki and Sipilä 2009) related to ecosystem services are not always properly taken into account (Pataki et al. 2011). Ecosystem services research is often at risk of succumbing to the common “green is good” truism (Bentsen 2010), increasing the risk of choosing research approaches to produce the desired results. The calculated ecosystem service values are uncertain (e.g. Spangenberg and Settele 2010, Hou et al. 2013), partially related to valuation methods (Spash and Vatn 2006) in addition to uncertainty in quantifying the base benefits and thoroughly understanding the related ecosystem processes. Besides the common “green is good” preconception, the objectivity of the valuation is threatened by the high impact of the many value judgements and assumptions that must be made (e.g. Farber et al. 2002, Spash 2008).

However, as stated by Rodriguez-Labajos and Martinez-Alier (2013), “We side with the sceptics but we understand the logic of those who are keen to apply monetary valuation and payments for ecosystem services”. While the valuation of ecosystem services and disservices and their net effects is still seen as requiring improvement in many areas, the urban development and policymakers need decision tools today. Thus, several ecosystem service assessment protocols and tools, such as CITYGreen (Longcore and Wilson 2004) and i-Tree tools (McPherson 2010), are available for urban greening and city trees (Roy et al. 2012). Various benefits that trees can provide, such as rainfall interception (Xiao and McPherson 2002) to aid stormwater management, and energy savings caused by shading and wind protection (McPherson and Simpson 2003), are related directly to tree size, mainly to tree leaf area and/or canopy cover derived from coverage of individual tree crowns (Nowak 1996, Bolund and Hunhammar 1999, Nowak et al. 2006, Dobbs et al. 2011). Thus, large trees give more benefits than smaller trees of

the same taxa and vitality (Rowntree and Nowak, 1991, McPherson 1992, Nowak et al. 2002) and obviously even smaller species and nursery plants (Sydnor and Subburayaly 2011). However, the same may apply to ecosystem disservices; the balance is yet to be estimated (Dobbs et al. 2011, Roy et al. 2012).

In assessing the ecosystem services and disservices provided by urban trees, knowledge about the tree population, its species composition and its biomass is important. Urban tree inventories have originally been developed to aid in practical tree management (Smiley and Baker 1988). The inventories also became important in keeping track of the entire urban tree population (Sjöman et al. 2012, Östberg et al. 2013), serving as base data in ecosystem service estimations. Biomass equations (BEs) are then commonly used to derive other tree size parameters from the diameter at breast height (DBH) of the urban tree.

When discussing urban ecosystem services and disservices, it must be also noted that the definition of “urban” is not universally accepted (e.g. Koninendijk et al. 2006, Raciti et al. 2012b). Trees growing in areas with various levels of urbanization and human influence can be broadly defined as “urban trees”, forming the “urban forest”. The lowest common denominator of the term “urban forestry” is that it deals with forests in urban areas, but beyond that the definitions vary greatly both over time and across geography and cultural backgrounds, as reviewed by Koninendijk et al. (2006). In this work, the term “street tree” is used to refer to trees that have been intentionally planted or left within the infrastructure of urban area, on sites with predominantly sealed surfaces. “Urban tree”, commonly used in this work when referring to the literature, follows the common and more general definition in parallel with urban forestry, including all trees in an urban area, whether naturally regenerating or planted. Forest and forestry, when used without the prefix “urban”, refer to any forests and forestry outside the broadest definition of urban forest.

1.3 STREET TREE ESTABLISHMENT AND GROWTH

More and more trees are planted in intensively built city centres where the state of ecosystem services tends to already be at its weakest (Scalenghe and Marsan 2009, Kroll et al. 2012, Setälä et al. 2014). Especially soil sealing (separation of soils by layers and other bodies from other components of the ecosystem, Burghardt 2006), which is common in densely built areas (Schalenghe and Marsan 2008), hampers soil functions. The imperviousness of the soil surface affects mineralization of nutrients (Raciti et al. 2012a, Zhao et al. 2012, Zong-Qiang et al. 2014) and impedes gas exchange and the carbon (e.g. Pouyat et al. 2006, Raciti et al. 2012a) and water cycles (e.g. Boyd et al. 1993, Assouline and Mualem 2002, Perry and Navaz 2008, Valtanen et al. 2014). Trees growing in such areas can give essential

contributions to local ecosystem services, but also highlight the many conflicts between urban environments and the requirements for tree survival and growth (Randrup et al. 2001, Costello and Jones 2003, Koeser et al. 2013, Delshammar et al. 2015, Morgenroth et al. 2015, Mullaney et al. 2015). The results of these conflicts are often classifiable as ecosystem disservices or ecosystem service cost, e.g. sidewalk damage (Nicoll and Armstrong 1998, McPherson 2000), root intrusion to sewer pipes (Rolf and Stål 1994, Randrup 2000, Pohls et al. 2002, Östberg et al. 2012) and damage by falling trees or branches (Lopes et al. 2009). In parallel with the ecosystem services approach, poor performance, short life-span and high mortality of street trees are common problems from the tree management point of view (Gilbertson and Bradshaw 1990, Skiera and Moll 1992, Roman and Scatena 2011). Especially tree mortality has been linked to many site-specific factors, such as proximity to high traffic, land-use type and level of maintenance, and to tree-related factors, such as tree size and species (Nowak et al. 2004, Koeser et al. 2013).

Street trees are planted in increasingly large size (Pauleit et al. 2002); in Finland, the current commonly recommended planting size is 6-8 cm DBH and trees larger than > 10 cm in DBH may be used (Peurasuo et al. 2014). In comparison, the common size range for street trees in the late 1990s was, based on a questionnaire to Finnish municipalities, 3-6 cm DBH (Lindén 2000). Large transplant size may contribute to the often reported poor initial survival of planted trees (e.g. Gilbertson and Bradshaw 1990, Pauleit et al. 2002). Most of the tree root length and also a considerable proportion of root biomass are left behind in the nursery (Watson and Sydnor 1987, Gilman 1989, Gilman and Beeson 1996) when a tree is dug up. For large trees, root ball size tends not to increase in proportion to tree size due to nursery equipment, transplant weight and handling issues. Transplanting leads to transplanting stress (Rietveld 1989) or transplant shock (McKay 1997), leading to the reduction of aboveground growth for one to several years after planting; root loss and resulting difficulties in water uptake are believed to be a major cause of the phenomenon (Harris 2007, Struve 2009). In relation to tree size, the transplanted root system tends to be smaller for larger transplants, and therefore, planting larger trees may exacerbate the transplanting shock (Struve et al. 2000), although there are several confounding factors that tend to go hand in hand with tree size (Struve et al. 2000, Struve 2009).

Challenges in the life of a street tree do not end with successful transplanting. The mainstream of urban tree research assumes that limited rooting volume is a major issue for street trees and they suffer from lack of soil-derived resources (Krizek and Dubik 1987, Kopinga 1991, Kristoffersen 1998), an issue that keeps pace with tree size. Currently, the issue is seen as one of the main challenges in the field of street tree study, and the trade in tree establishment solutions is largely based on this idea (Bühler et al. 2009, Mullaney et al. 2015). The connection between street tree site planning and

ecological functions of tree soils are rarely explicitly understood, although there are many different tree establishment methods and practices, each with its own strengths and weaknesses in relation to ecosystem functions. The suitability of a given establishment method also depends on both the surrounding environment and management practices and resources. It follows that no solution is best suited for all geographical and cultural contexts.

In the 1990s, street trees in Finland were usually planted in boxes (e.g. Riikonen 1990, Jyränkö et al. 1994), first made of wood, then of corrugated metal fencing or concrete; the volume of these boxes was from one to about five cubic metres. Average soil volume was still around 2 m³ for street trees in the late 1990s (Lindén 2000). It was already known, however, that these volumes were often inadequate (Jyränkö et al. 1994, Rolf 1994). Calculations of soil volume needed to supply water for tree transpiration (e.g. Kopinga 1985, Vrecenak 1988, Lindsey and Bassuk 1991) gave volumes that became more and more difficult to attain in the heavily competed urban environment. It became apparent that using the space under the pavement was often necessary if increases in rootable soil volume were to be achieved. This space under the pavement is usually not usable for tree roots, because achieving load-bearing while avoiding the risk of frost heaving requires heavy compaction of the soil or, as is typical currently in Scandinavia, replacement of local soil with crushed rock.

Amsterdam tree soil (Couenberg 1994) is often stated to have been the first successful attempt in allowing root development under pavement. Amsterdam tree soil comprises sharply graded sand that is mixed with a small amount of organic matter and clay (Couenberg 1994). Its main disadvantage in northern Scandinavia is that it is not entirely frost heaving-free due to the fine overall particle size distribution (Ferguson 2005). In the early 1990s, the idea of structural soils (often also called skeletal soils) emerged in the literature (Rolf 1994, Grabosky and Bassuk 1995, Neal and Whitlow 1997, Kristoffersen 1999). According to Rolf (1994), structural soils were first used in the Netherlands in the 1980s, and in 1994 were also in use in Sweden, Germany, Norway and Denmark. Structural soils share the same basic idea with Amsterdam tree soil; they consist of a two-component material mix that allows for both load bearing and tree root growth. Instead of sand, rock forms a load-bearing and frost-heaving-free matrix with some empty space within. This empty space is protected from compaction and can be filled with material suitable for root growth. In addition to structural soils, various “suspended pavement” solutions were known from early on (e.g. Smiley et al. 2006), giving uncompacted soil volume for tree roots by bridging the road or footpath over the tree soil with e.g. large concrete slabs. These methods have not gained much popularity, likely because they have to be planned and built case by case.

Early reports on tree growth on structural soils relative to topsoil in experimental conditions gave generally similar (Kristoffersen 1999, Grabosky

et al. 2001, 2002) or weaker but still acceptable (Kristoffersen 1999, Smiley et al. 2006) tree growth. Overall, studies of tree growth on structural soils are somewhat conflicting. This is not surprising, as the category contains very different implementations of the base idea (e.g. Costello and Jones 2003, Bartens et al. 2009, Xiao and McPherson 2011), and tree species and maintenance in the trials have varied. Increasing evidence exists that different tree species perform best with different establishment methods (Bühler et al. 2007). Much of the research has aimed to produce results directly applicable to practice in certain regions and conditions, and thus, application of the results elsewhere is problematic. Recently, the study by Grabosky (2015) called for a common standard for reporting the properties of tested structural soils, which is a step in the right direction. Variation of experimental conditions, species and maintenance is, however, difficult to overcome in a research field so dependent on its practical application. The few existing studies with a more theoretical approach give valuable insights into root resource acquisition and water relations in structural soils (Loh et al. 2003, Grabosky et al. 2009).

In the wake of structural soil development, several solutions that combine the “suspended pavement” methods and load-bearing matrix methods have arisen. Constructing the load-bearing matrix from e.g. stackable, durable plastic elements or steel or concrete pillars allows for a larger fraction of fine soil than the stone matrix (Bartens et al. 2010). These methods have not been researched much or gained a real foothold in Scandinavia yet, perhaps because most share the same problem as the “suspended pavement” method; they must be planned and constructed case by case, increasing the cost of the process. Stone-based structural soils can be easily integrated in Scandinavian street construction, presenting few problematic vertical boundaries in soil load-bearing capacity. There have also been some trials of “humusless” tree soils, i.e. growing media that has little to no fine soil. This is based on the idea that soil O₂ availability is very critical for tree roots, and this type of soil is easy and relatively low cost to build (Schönfeld 2004). Results from this line of research are scant and somewhat mixed (Schönfeld 2004, Andreasson et al. 2014).

Overall, all of these street tree establishment methods when successfully applied provide an increase in rootable soil volume for trees compared with open tree pit size realizable on the same site. However, the soil surface sealing above the soil volume (e.g. asphalt or pavement) can be seen as diminishing many of the benefits that this additional soil volume could provide. Soil sealing prevents or hampers gas and water fluxes as well as carbon and nutrient cycles (Schalenghe and Marsan 2008), isolating the soil underneath. Therefore, it is not a surprise that comparison with a large open planting pit (at least 15 m³ with at least 12 m² unsealed surface) in Denmark shows that tree growth in structural soil or Amsterdam tree soil is not as good within the first 15 years (Bühler et al. 2007). In contrast, in a case studied in New York, better tree growth has been found 10 years after

planting on structural soil than on regular soil with lawn (tree lawn) (Grabosky and Bassuk 2008), but, unfortunately, the corresponding soil volumes or the level of maintenance were not reported. Permeable pavements, concrete and asphalt appear as possible solutions to some of the issues caused by soil sealing, allowing for varying degrees of water and gas exchange (Morgenroth and Buchan 2009, Morgenroth and Visser 2011). The problems in nutrient and carbon cycles caused by soil sealing (Raciti et al. 2012a) likely cannot be solved by these advances, as litter is either removed or blown away from paved sites and the transport of particulate matter through even permeable materials is uncertain (Nehls et al. 2006).

1.4 CARBON IN URBAN TREES AND SOILS

Over time, carbon (C) taken up in photosynthesis is accumulated in the biomass of trees, some stored for decades or centuries in the woody biomass, some released back to the carbon cycle much sooner as litter. Litter may end up in longer term storage in soil as soil organic matter or return to the atmosphere sooner or later. In terms of measured tree biomass, the long-lived compartments dominate, but in terms of annual investment of C, the short-lived, litter-producing leaves and fine roots become very important. In assessing C sequestration of urban trees, it is conventional to account only for the C in the current live biomass (e.g. Nowak and Crane 2002, Díaz-Porrás et al. 2014, but see also Zheng et al. 2013). The destiny of litter can be difficult to trace, and no validated estimation methods for root or root litter biomass are available for street trees.

Changing biomass partitioning is one of the mechanisms to allow tree acclimation and adaptation to its environment (King 1991, Berninger and Nikinmaa 1994, 1997, Ibrahim et al. 1997, King et al. 1999, Mäkelä 1999). Changes in biomass partitioning show at different paces in short-lived leaves, composed of very recently accumulated biomass, than in the woody structure, which grows over a longer time period. There are indications that scarcity of soil resources can increase the belowground biomass in relation to the aboveground compartment (Vanninen and Mäkelä 1999, Coleman et al. 2004, Helmisaari et al. 2007, but see also e.g. Ingestad and Ågren 1991). Heavy competition for light tends to produce slender, tall trunks with few and smaller branches (Ilomäki et al. 2003). It appears that the opposite is true as well; without competition for light, trees have larger branches and less biomass in the trunk (Ares and Brauer 2005, Zhou et al. 2014), but as this situation is not commonly encountered in native forest or traditional forestry, data is scarce.

Many models dynamically describing the formation of tree structure with biomass allocation and structure relationship rules have been developed and tested against measurements from forest stands with varying success (Lacointe 2000, Sievänen et al. 2000, Godin and Sinoquet 2005). Several

such models and also the origins of tree biomass equations are based on pipe model theory (Shinozaki et al. 1964 a, b). Pipe model theory states that there is a constant quantitative relationship between leaf mass above a certain point and sapwood area below it. The original form of the theory (Shinozaki et al. 1964a) makes no statement as to how this relationship is formed, allowing it to be the product of both water transport capacity needed and the demands of structural stability.

There is evidence that not all tree biomass relations conform to the pipe model theory (e.g. Mäkelä 2002, Gilman and Grabosky 2009, Gehring et al. 2015, Sattler and Comeau 2015), but neither do observations go directly against it; some form of constant relationship between biomass compartments is commonly found. Thus, derivation of tree biomass, leaf area, etc., from tree DBH with biomass equations (BEs) is a common practice, also in urban forestry (e.g. Nowak 1996). What is known concerning tree structure and biomass relations is, however, mostly drawn from measurements of trees growing in either managed or unmanaged forests, which may be a problem because tree structure adapts to its environment. Biomass and structure studies on urban trees are becoming available (Gilman and Grabosky 2009, McHale et al. 2009, Dahle et al. 2014), but as yet, they cover little of the tree species and growing site variation encountered in urban environments. Thus, the biomass and C storage of urban forests is commonly based on non-urban tree BEs (e.g. McPherson et al. 1994, Hutyra et al. 2011, Strohbach and Haase 2012) and usually does not account for root biomass. With such methods, the findings indicate that C storage in urban forest aboveground biomass per area is, for example, about half of that in natural forests in the same region in USA (Nowak and Crane 2002). Overall, however, such comparisons are difficult to make due to varying land uses represented and different definitions and methods employed in the studies (e.g. Jo 2002, Davies et al. 2011, Strohbach and Haase 2012).

Depending on the ecosystem in question, soil can also contain a considerable C stock; globally, soils are the largest terrestrial C pool (Schlesinger 1997), which applies to boreal forests as well (Liski et al. 2006). In a vegetated area, organic matter is constantly produced and incorporated into soil. Soil organic matter is composed of both live and dead organisms, and the composition of these constituents can be very complex. Organic compounds differ widely in their ease of decomposition, and fractions more resistant to decomposition tend to accumulate in the soil organic matter.

In a native environment, the soil organic matter is often considered to be in a dynamic steady state (Jenkinson et al. 1990), meaning that annual input and mineralization are approximately equal. In many instances, this assumption does not hold in reality; for example, Finnish upland forest soils are believed to be accumulating soil C (Lehtonen and Heikkinen 2015). Similarly, in urban areas, the assumption of dynamic steady state is often not valid. Land use change, which can be found at the origin of any urban soil,

brings about a change in the input and mineralization of organic matter in the ecosystem. The C stock and input to soil in the ecosystem often initially decreases drastically in a land use change from either agricultural or forest to urban (Chen et al. 2013), but the long-held assumption that this deprived state of soil C stock continues appears to be false (Churkina et al. 2010, Edmondson et al. 2012, 2015). Evidence is starting to accumulate on long-term effects (Pataki et al. 2006), implying that C stock development after this initial decrease is likely to be positive (Scharenbroch et al. 2005, Bae and Ryu 2015), depending, however, to some extent on the climatic region. At least in arid areas, resources are invested in urban plant irrigation and fertilization, which may bring about higher soil C stocks than usual in the surrounding areas (Golubiewski 2006, Pouyat et al. 2006). C stocks of urban soils have not yet been very strongly tied to urban trees, but recent research suggests that the selection of urban tree species may have an effect on soil C stock dynamics (Edmondson et al. 2014), and planted urban soils appear to have higher C stocks than sealed soils (Edmondson et al. 2012, Wei et al. 2014).

1.5 WATER AND URBAN TREES

For an urban tree, several stumbling blocks hinder the natural water cycle. One issue often identified is that soil volume and soil water storage accessible to tree roots tend to be limited (Loh et al. 2003, Buhler et al. 2007). Rainfall may not be able to enter soil because soil is sealed, as is seen in the relationship between runoff percentage and soil sealing (Pauleit and Duhme 2000, Matteo et al. 2006, Valtanen et al. 2014); also several studies show a connection between the area of open soil surface around trees and tree size at a given age (Grabosky and Gilman 2004, Sanders et al. 2013, Sanders and Grabosky 2014). From the early days of urban forestry as a field of study, it has been commonly assumed that urban trees often suffer from drought (see discussion in Whitlow and Bassuk 1987), but there is surprisingly little research that directly supports this supposition (Hodge and Boswell 1993, Bühler et al. 2006, Nielsen et al. 2007). Flooding or excess water in the soil, which could also be caused in part by the impaired water cycle in the urban environment, is even more rarely shown to be a problem for urban trees (Berrang et al. 1985, Smith et al. 2001).

Because of the urban heat island (Oke 1982), urban trees are likely to face higher water vapour pressure deficit (D) situations than are common in forests in the same climatic region. Both the supply and demand of water may thus be different in urban areas than in rural or forested areas. When discussing water relations of trees, the distinction between aboveground and belowground environment is somewhat artificial; water availability experienced by trees is a combination of supply and demand, the first of which is largely determined by underground conditions and the latter by

aboveground conditions, the two being separated by soil sealing to a much higher extent than in forests.

In well-watered conditions, plant transpiration is roughly similar to evaporation from a wet surface, driven by the energy supply for a water phase change from liquid to vapour form and a partial pressure gradient driving vaporised water away from the evaporating surface. A boundary layer of unmoving air forms between the evaporating surface and atmospheric air, through which the vapour must diffuse. In a forest or agricultural field, the canopy is usually considered uniform, i.e. the boundary layer is assumed to be one continuous horizontal layer between the canopy and atmosphere – a problematic assumption for urban trees, which are typically sparsely spread or isolated. Air mixing, i.e. wind, will keep the boundary layer thin, bringing about convective mixing of air mass, which speeds up the evaporation process. In plants, the vaporisation of water happens in the intercellular spaces, which are connected to the atmosphere via stomata. Transpiration is determined largely by the same factors as evaporation, but the plant is able to regulate the transfer of gases in and out of leaf intercellular space via regulation of stomatal openings. Stomata react to environmental factors such as light intensity, D , air CO_2 content and tension of water within the tree, signalling water availability. Also hormonal signals, especially abscisic acid, are involved in stomatal regulation (Johnson et al. 2001, Cutler et al. 2010).

The stomatal control can regulate the gas exchange between the plant and atmospheric air, but this happens at the expense of carbon uptake and growth; closing the stomata to prevent water loss will also cut off the source of CO_2 apart from what can be recycled from the plant's own metabolic processes. In addition to stomatal regulation, there are numerous long-term and short-term mechanisms of water loss control that plants have at their disposal, reviewed by e.g. Chaves et al. (2002) and Bréda et al. (2006). The transpiring leaf area of a tree is strong determinant of water use, and leaf abscission in reaction to drought is a way to downscale the need for water (Battaglia et al. 1998, Marron et al. 2003). Tree crown structure and even leaf petiole length, leaf size or leaf angle (Forseth and Teramura 1986, King 1997, Van Zanten et al. 2010) may have an effect on transpiration via leaf exposure, self-shading and regulation of the thickness of the boundary layer.

The interrelations of urban trees and water can be examined from the point of view of water availability, lack or excess, experienced by trees, but the disturbed urban water cycle implies also a need for stormwater management in the urban environment, bringing about the ecosystem services approach to urban tree water issues. The runoff from sealed surfaces and other poorly permeable areas in a city must be controlled to prevent floods, as flooding events are both costly (e.g. Dutta et al. 2001, Morita 2008, Muis et al. 2015) and potential health hazards (Ahern et al. 2005). Stormwater management is usually predominantly considered over the short term because flooding events follow high-intensity rain events, i.e. high amounts of rain in a short time. In this rain-event time scale, urban trees aid

in stormwater management mainly by intercepting rainfall in their canopies (e.g. Herbst et al. 2006, Xiao and McPherson 2011). Canopy rainfall interception can be considerable and can be modelled on a large scale with relatively simple canopy parameterization (Muzylo et al. 2009). The role of tree transpiration becomes visible mostly over a longer time scale, as it participates in emptying the soil water storage capacity for the next rain event. The linkage and importance of this in relation to short-term rainfall event level dynamics are relatively poorly known (Rauch et al. 2005).

1.6 CONNECTION BETWEEN WATER AND CARBON CYCLES IN URBAN TREES

Trees transpire water constantly while stomata are open, which in turn is a requirement for carbon dioxide uptake. This creates a strong link between water and carbon cycles in ecosystem. Atmospheric demand for water is dependent on air water content and temperature, which together determine water vapour pressure deficit (D) of air. Transpiration is also tightly linked to transport of substances within trees and evaporative cooling of the plant. Due to this linkage in water and carbon flows through the ecosystem, tree growth, and consequently, C sequestration, is expected to respond to limitations in soil water resources. The effects of drought are rare, yet fairly well known in temperate (Bréda et al. 2006) and boreal forest trees (e.g. Irvine et al. 1998, Holopainen et al. 2006, Kljun et al. 2006, Duursma et al. 2008), and drought tolerance is a basis for urban tree selection and breeding (e.g. Percival et al. 2006, Sjöman et al. 2015), although studies on street tree water status rarely show the problem.

In addition to drought, soil water conditions can be problematic at the other extreme as well. As soil water and soil air, supplying necessary oxygen to roots, occupy the same pore space in soils to a large extent, water-saturated soil causes problems for root functions. Impaired root function due to flooding is a long-known phenomenon in forest trees, as reviewed by e.g. Coutts and Philipson (1978) and more recently by Sauter (2013) and Kreuzwieser and Rennenberg (2014). Flooding is known to affect tree growth and survival (Kozłowski 1986, Wall and Heiskanen 2009). Similarly to the effects of drought on urban tree vitality and growth, also the effects of flooding on urban trees have been demonstrated rather rarely, although the phenomenon can be expected to occur in urban environments (Smith 1994, Ware 1994). Also the presence of plentiful sealed soil surfaces in urban environments implies potentially higher water inputs in unsealed areas, depending on local small-scale topography and surface runoff routes. The scarcity of quantifiable evidence regarding the effect of soil water status on urban trees is no doubt partly due to the fact that these matters may be considered self-evident, but it is also quite difficult to assess the soil water status for urban trees because the soil environment is highly heterogeneous

and often difficult to access. Thus, the concern voiced in the late 1980s (Whitlow and Bassuk 1988) appears valid today; some of the most critical beliefs concerning urban trees and their environment are still largely derived from practical experience and anecdotes.

Even in forestry, not much is known about the effects of short- and long-term soil water status on tree crown structure (Grier and Running 1977, White et al. 1998, King et al. 1999), although it is known that firstly, water availability is one of the most important environmental factors affecting plant growth, and secondly, trees respond to their environment with changes in biomass allocation. While hypotheses concerning the effects of water availability on tree structure can be supported by comparison of biomass allocation between tree species (Litton et al. 2007), such comparisons cannot determine whether the variation shows genetic (adaptation) or phenotype (acclimation) -level effects. The urban flora and fauna tend towards being more different from local native than other urban populations elsewhere (La Sorte et al. 2007); also some urban tree clones are very widely used. This clearly opens an avenue for understanding the tree biomass relations and allocation in relation to environmental factors, as the same urban tree species and even the same clones can be studied in different climates around the world.

2 AIMS OF THE STUDY

The aim of this study was to analyse the potential of young street tree plantings to produce ecosystem services. To this end, tree growth and tree carbon and water exchange on selected urban streets were investigated, and the relationships between these factors, environmental variables and tree properties were examined.

Within this general framework, specific aims of the study were as follows:

- To evaluate tree transplanting recovery in relation to tree structure and soil water conditions and the development of crown structure (Study I). It was hypothesized that:
 - Crown allometry reflects site conditions and tree transplant recovery status.
 - Soil moisture has a clear influence on tree growth in boreal urban conditions.
- To assess the carbon sequestration effects of urban tree plantings with artificial tree soil by measuring soil and tree carbon content dynamics during the first nine years from tree planting, and to predict tree biomass C sequestration about one hundred years in the future (Study II). The hypotheses were:
 - Existing biomass equations are sufficient for the prediction of tree biomass.
 - Major changes in soil organic carbon content will occur after establishment.
 - Under common local practices, the C loss from artificial tree soil under pavement offsets the beginning of net C sequestration of street tree planting beyond the current life expectancy of street trees.
- To assess the potential effects of tree transpiration on stormwater management by measuring street tree water use and to determine the most significant drivers of water use (Study III). It was hypothesized that:
 - The annual variation in tree water use can be predicted with variation in microclimatic factors, soil water availability and tree leaf area.
 - On an annual time scale, street tree transpiration can significantly contribute to urban stormwater management.

Observations of tree structure, biomass relations and growth were used to investigate transplanting recovery, carbon dynamics and also the interrelation of tree structure, water availability and use. Soil carbon stock changes were evaluated with repeated soil sampling from three different tree soils. Tree water use was measured and analysed in relation to environmental factors as well as tree-related factors such as tree structure. The analysis of tree biomass relations and water use from the case study sites

was utilized in predicting the development of the C stock and net sequestration of street tree plantings over a longer period of time and the assessment of the potential contribution of tree transpiration to storm water management.

3 MATERIALS AND METHODS

Materials and methods are described briefly here. More detailed information can be found in the original publications (Studies I-III).

3.1 STUDY SITES

The studies were conducted on two streets built in 2001-2003. Both are regular streets, but they were planned and equipped to allow for continuous data gathering for research purposes. Both streets were built with three different structural soil growing media, two of them based on commercially available tree soils and one mixed specifically for the research site. These soils were used for at least three trees on both streets. The streets are located in Viikki district, Helsinki, in southern Finland (N 60 15', E25 03'). The climate is hemiboreal and the region is low-lying, with the streets only 2-6 m above sea level, and with predominantly clay soils.

One of the streets, referred to as the Tilia site according to the tree species planted as street trees on the site, is some 250 m in length, running roughly north-south on the southern side of the University of Helsinki Viikki campus. It was built across agricultural fields in the early 2000s, but by 2014, two of the lots on the eastern side of the street had been built; one has a parking lot next to the street and the other a 2- to 3-storey office building. The western side of the street is bordered by a park, parking lots and older 1-storey buildings. The asphalt-surfaced street serves as a low-traffic thoroughfare in the campus area and slopes from north to south, with the northern end about 6 m a.s.l. and the southern end at 2 m a.s.l.



Figure 1. The Tilia site in early September 2012, ten years after planting, seen from the south end and looking towards the north. The studied trees can be seen on the left side of the street, between parked cars. The left side of the street borders a park, and on the right side, an apartment building parking lot was built in 2009. Office buildings can be seen on the far right.

The second street, referred to as the Alnus site, is located some 500 m east from the main campus area and is surrounded by residential 2- to 3-storey buildings close to the street, forming a shallow street canyon in contrast to the more open Tilia site. The street is a cul-de-sac serving only the local residents, paved with concrete unit pavers of 28 x 16 cm with a 2 mm sand joint. It is approximately 200 m long, running also roughly in the north-south direction and slopes only slightly towards the south, from 4 m a.s.l. to 3.5 m a.s.l.



Figure 2. The Alnus site photographed from boom lift in August 2010; view is from north towards south. Planting pockets with their granite unit pavers were arranged along the street paved with concrete unit pavers.

3.1.1 STRUCTURAL SOILS

On both sites, three different structural soils were used, as one of the initial aims of the sites was to find a suitable structural soil, made of local materials, for Finnish conditions. One of the tested soils was a commercially available mix (soil 1), on soil 2, the fine soil fraction was commercially produced landscaping soil, and the third mix was made-to-purpose for the study to be coarser relative to the other two mixes. In tree soil 1, fine soil was composed of peat, sand and clay. In soil 2, fine soil was derived from composted sewage sludge mixed with peat, sand and pine bark. In soil 3, the components of fine soil were fine gravel, sand, clay and leaf compost (no peat added in the composting process). The structural soils at the sites were built either as planting pockets for 2-4 trees (Alnus site) or as a continuous strip (Tilia site). The planting strip and pockets were 3 m wide and 1 m deep, and the variation in planting distance was such that *Tilia* trees at the Tilia site had 45-50 m³ of structural soil allotted per tree, and *Alnus* trees planted at the Alnus site had 15-30 m³ of structural soil allotted for each tree. The different structural tree soils each contained ca. 2/3 stones ranging from 30 to 120 mm in size and 1/3 fine soil by volume. Soils were pre-mixed and installed in three layers, each compacted with a vibratory soil tamper.

During soil construction root sampling wells made of 100 mm diameter PVC pipe with holes drilled along the length of the pipe were installed through the entire structural soil layer. The first well was located in the corner of the tree grate, 80 cm away from the trunk, and the other 3 were 50 cm away from the first and each other, in direct line with the tree radius. The wells were filled with the fine soil fraction of the respective structural soils.

Cast-iron tree gratings 2.25 m² in size were used around the trees, and the streets outside the gratings were paved over with sand-jointed unit block pavers. The planting pockets around the *Alnus* tree gratings were paved with 160 x 160 mm granite unit pavers with 10-20 mm drainage gravel filled joints. The *Tilia* trees had 10 x 10 cm granite unit pavers with 10 mm sand joints covering an area of 4 m² around the tree grate, and beyond this, concrete blocks (28 x 16 cm) were installed with 2 mm sand joints over the planting strip. An average soil volume of 50 m³ per tree at the *Tilia* site and 22.5 m³ at the *Alnus* site was used for the groups of trees on each street for which data was usually collected.

3.1.2 TREES

In late 2002, 15 *Tilia* × *vulgaris* Hayne trees were planted as balled and burlapped, DBH 8.2-10.8 cm, originating from Harviala Oy's nursery in Janakkala, near Hämeenlinna in southern Finland. As is common in Finland, the trees were unspecified clones, grafted to seedling *T. cordata* Mill. rootstock. Of these 15 trees, 5 did not open their buds in the following spring and were replaced in late 2003, and one of the replacements died and was replaced yet again in late 2004. Examination of the root balls of the dead trees showed very poor root system quality (example given in Figure 3), which is assumed to be the main reason for these early tree deaths. No trees were lost later than within one year of transplanting.

Alnus site had 22 *Alnus glutinosa* (L.) Gaertn. f. *pyramidalis* 'Sakari' trees planted also in late 2002. The trees originated from Puutarha Tahvoset Oy, located in Pohjankuru in southern Finland. Trees were balled and burlapped and the tree DBH at planting was 6.7-11.0 cm, grafted to seedling *A. glutinosa* rootstock.

At the *Tilia* site, 3 *Tilia* trees were planted in tree soils 1 and 2 each, and the rest of the trees in soil 3. At the *Alnus* site, soil 1 had 5, soil 2 had 6 and soil 3 had 11 trees planted. At the *Tilia* site, only 4 and at the *Alnus* site, 6 of the trees on soil 3 were included in the datasets in Studies I, II and III, except for the assessment of leaf abscission, which included all trees on each site (Study III). These additional trees on soil 3 on each site were planted as material for another urban tree study, which was unfortunately discontinued early on. On both sites, similarly sized trees were chosen among the nursery trees to be planted next to three monitoring points to be built and instrumented for sap flow measurements (the intensive monitoring trees).



Figure 3. The dug-up root system, showing the edges of the initial root ball, of one of the originally planted *Tilia* trees not opening their buds in the first year after planting. The reason for unsuccessful transplanting appears quite clear; while being in the soil for some 8 months after lifting from nursery might have caused some fine roots to disintegrate, the root stumps remaining appear to be mainly unbranched and cut off sharply at the edge of the original root ball. The size and quality of the root ball appears not in proportion with a DBH 8-11 cm tree.

The trees were irrigated weekly in May-August for two years after transplanting according to standard practice (Rakennustietosäätiö 2010). Early on, no pruning was performed, aside from removing broken and dead

branches and shoots growing from the rootstocks. Regular pruning was started in autumn 2008, and thereafter, conducted approximately annually for the *Tilia* trees and every 3-4 years for *Alnus* trees. *Tilia* trees, which had started with a clear trunk of 160-180 cm, were crown lifted, first with reduction pruning of large lower branches and proceeding to lower branch removal. *Alnus* trees required no crown lifting due to the narrow crown shape, thus needing little pruning.

3.2 METHODS

Studies I-III used data from continuous automated measurements from the study sites and additional weather and micrometeorological data from nearby weather and eddy covariance stations. The studies also used manually collected data on tree growth and properties and soil properties from the study sites.

At the study sites, a monitoring point with a datalogger was established on each soil on each street (Figure 4), at six points altogether. At each monitoring point, tree sap flow, micrometeorology and soil environment were continuously measured. From each monitoring point, all data were recorded to the datalogger at 1- to 30-min intervals, with a 2-min logging interval predominating over the growing season and a 10-min interval in

winter. All measured data were sent via gsm connection to mass data storage, automatically during the growing season and manually in the wintertime.

3.2.1 DATA COLLECTION

3.2.1.1 Aboveground environment

At the study sites, each monitoring point had photosynthetically active radiation and temperature measured on top of an 8-m-tall pole. In addition, several variables were used in Study III: rainfall (p), wind speed (u), water vapour pressure deficit (D), global (I) and net all-wave radiation (R_n). These were either measured at the nearby (4 km SW) SMEAR III eddy covariance measuring station ($60^{\circ}12'$, $24^{\circ}57'$, 51 m a.s.l.), or derived from combined SMEAR III and tree site measurements. Additionally,

precipitation measured at Helsinki-Vantaa Airport ($60^{\circ}19'$, $24^{\circ}57'$) by the Finnish Meteorological Institute was used in Study I.



Figure 4. One of the measuring points at the Tilia site, seen in autumn 2011. Datalogger box on the left along with the temperature and PAR radiation measuring pole. The tree adjacent to the datalogger box was instrumented with sap flow sensors; the black plastic radiation shield can be seen attached to the inside of the trunk guard.

3.2.1.2 Belowground environment and soil sampling

At the study sites, soil temperature and soil volumetric water content (SWC) of the fine soil fraction of the structural soil were measured at each monitoring point, in six locations altogether, temperature at depths of 10, 30 and 60 cm and SWC at depths of 10 and 30 cm (Studies I, II, III). In addition to the continuous measurements of soil water content and temperature, manual soil sampling was performed at three-year intervals from construction: in 2005, 2008 and 2011 (Study II). In September-

October, sampling pits were dug on each site and soil, and soil and root samples were collected for analysis. At each sampling time, two sampling pits were dug on each soil at both sites, 12 pits altogether. These were additionally divided into 30-cm layers in the vertical direction. The fine soil of the structural soil was collected with an industrial vacuum cleaner, mixed thoroughly and sampled.

The soil organic matter content of the samples was analysed as loss-on-ignition (LOI) and dry weight, and loose bulk density was measured (Study II). Particle size distribution was analysed with dry sieving and a laser diffractometer (Study II) and water holding capacity with pressure plate apparatus (Study I). Separate subsamples were incubated to assess the temperature, moisture and LOI dependence of soil respiration. Visible roots were picked and eight subsamples were placed in incubation bottles that were filled with atmospheric air of known composition, sealed and placed in four different temperatures (5-25°C) for 24 hours. CO₂ production was then measured from bottle headspace air samples with a gas chromatograph.

3.2.1.3 Tree measurements

Three of the trees on both study sites were measured for sap flow, and from six to twelve trees on both sites were measured manually for growth and various crown characteristics. Additionally, a small number of older street tree plantings in Helsinki were measured for DBH (Study II).

The intensive monitoring tree at each measuring point was instrumented with sap flow sensors (Study III) to gain a measure of tree transpiration. The construction and installation of the sensors are as described in Hölttä et al. (2015). Daily mean transpiration for the years 2008, 2009, 2010 and 2011 was calculated (Study III). Daily means were considered preferable to annual sums because daily means could be attained without gap-filling the data.

Manual tree measurements aimed to estimate the development of tree crown properties, tree growth and biomass accumulation in different tree parts. Tree trunk circumference at breast height and crown dimensions were measured for all trees in the research sites annually. The annual total shoot extension of the trees was measured for each year based on 30 measured shoots per tree, after shoot growth cessation (Study I). The within-year change in leaf area was estimated based on shoot growth rate measurements of six trees per site each year (Study III). From each sampled tree, three first-order branches were chosen for measuring shoot extension growth 2-3 times per week during the period of active growth. After the shoot growth cessation, the number of leaves was counted every 1-2 weeks for five randomly chosen lower crown shoots of all trees on each site. Mean number of remaining leaves was then compared with the number of leaves at growth cessation. Combined, the shoot growth rate measurements and follow-up of

leaf loss in the autumn formed an index for intra-annual leaf area change (Eq. 4, Study III).

For the annual measurement of tree leaf area and biomass (Figure 5), the diameter of all first-degree branches and diameter of the trunk above each branch were measured, and the height of the branch on the trunk and the trunk diameter below the branch were recorded (Studies I and II). Leaf samples were collected from a sample of branches and branch leaf area measured. Both branch basal area and branch leaf area were log-transformed and a linear relationship was established between them (Study I). This was then used to estimate the leaf area for each tree.

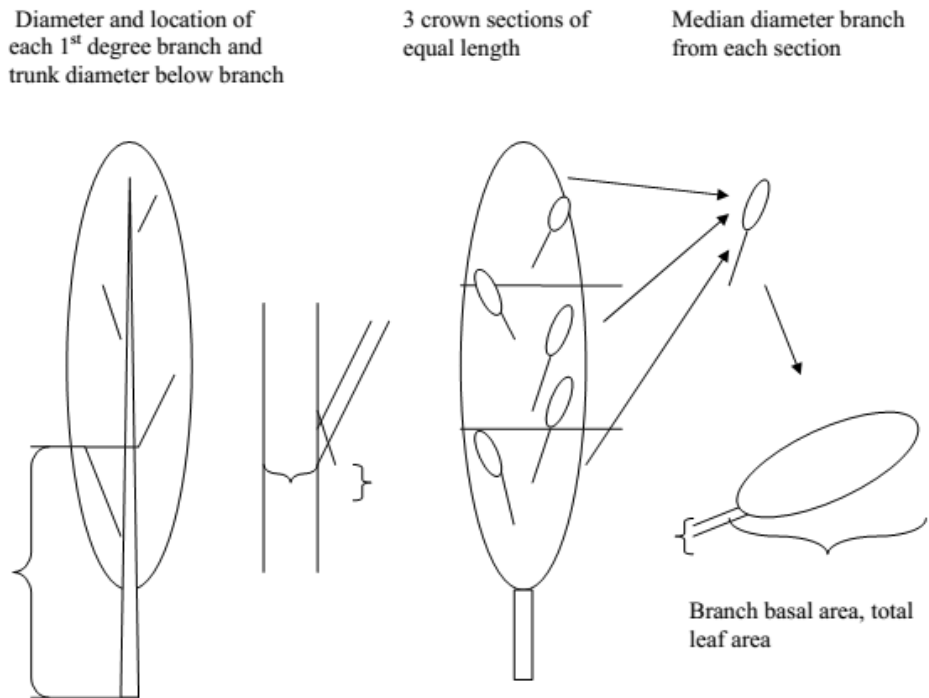


Figure 5. Tree biomass and leaf area measurements. First, the height and diameter of each live branch were recorded, together with trunk diameter below the branch. Then, the tree live crown was divided into three sections of equal length, and a median diameter branch was selected from each section for leaf area and biomass measurements.

For tree biomass estimation (Study II), the diameter of each first-degree branch was used to estimate its dry woody and leaf biomass based on diameter/dry mass relationship determined for a sample of pruned branches. To estimate the amount of carbon in leaf litter and pruned branches at a given time (Study II), each leaf cohort of the measured trees was subjected to decay function (Olson 1963) at an annual time step. Tree trunk volume was

calculated from the trunk diameter and branch height data collected as stacked cylinders, using specific gravity for *Tilia americana* L. (0.40) for *Tilia* and *Alnus rubra* Bong. (0.43) for *Alnus* (Alden, 1995).

To estimate tree C accumulation in relation to C lost from the tree growing media, tree biomass accumulation was predicted for ca. 100 years into the future using pre-existing biomass equations (Study II). Equations from the literature were used for diameter development over time for both species. Only one predictive equation was available for *Alnus* and two for *Tilia* (Study II, Table 1). Based on the predicted DBH, C in aboveground biomass, roots and separately in leaves, branches and stem was estimated with biomass equations (BEs). Also each leaf cohort was subjected to decay function as in the measurement-based estimates. A pruning program, imitating the regular pruning of street trees, was used to produce pruning biomass, which was then subjected to similarly decay over time. To assess the applicability of the DBH growth predictions (Study II), data were collected for 19 *Tilia* plantings of known planting year (Mustiala 2003). For *Alnus*, sufficient data for assessing DBH prediction could not be attained.

The number of trees measured at the study sites for leaf area, biomass and annual shoot extension varied annually depending on the type of data collected and the time and resources available for the measurements. The intensive monitoring trees (one tree per soil on each site) had the highest priority and were measured each year. The sampling aimed for representativeness and equal numbers of trees from each soil type. The trees measured in addition to the intensive monitoring trees, and their priority order within each soil type, were randomly selected at the time of the first measurements. Thereafter, the priority order always remained the same, but the number of measured trees on each soil and site varied from 2 to 3 at the *Tilia* site and from 2 to 4 at the *Alnus* site. This resulted in repeated measurements pattern where the repeat interval varied from one to several years for each tree.

3.2.2 DATA ANALYSES

Tree transplanting recovery, response to soil water conditions and differences between the three tree soils were studied by analysing tree shoot growth with a mixed model (Study I). Because the measurements were repeated for each tree, a spatial power covariance structure was used in the analysis. The sampled trees were treated as if the sampling had been completely random. The effects of the three different soils on tree shoot growth were compared with Tukey–Kramer adjustment for multiple comparisons.

The effects of soil water status and transplanting recovery were examined with regard to the branch basal area to leaf area relationship. To determine the parameters of the relationship (Study I), the data on leaf area and branch basal area were pooled by species and year, both were log-transformed and a

linear regression was fitted between them (Eq. 1, Study I). This was then transformed back to the original scale in order to predict leaf area for all branches (Eq. 2, Study I; used also in Studies II and III). To determine the branch basal area to branch biomass relationship (Study II), a non-linear regression was fitted to basal area and biomass measurements without any data transformation, as a good fit could be attained on the original scale.

To examine whether tree biomass measurements produced similar results as BEs from the literature, the biomasses of each biomass compartment measured for the studied trees were compared with biomasses estimated for the same trees with BEs with Tukey's two-tailed T-test (Study II).

In Study II, tree soil C loss was estimated with two methods, based on LOI change, and with an Arrhenius-type model parameterized with soil incubation experiments (Eqs. 4 and 5 in Study II). After the testing of year-, soil- and site-specific datasets, the smallest possible number of separate parameterizations that presented no heteroscedasticity problems within the dataset was used (separate parameterization for each soil and site). This model was applied to the tree sites, based on the measured soil temperature and water content at the sites.

To test the possibility to predict annual variation in tree transpiration based on microclimate, soil water and tree leaf area, the tree transpiration for the years 2008-2010 was modelled with a Penman-Monteith (PM, Penman 1948, Monteith 1965) transpiration model (Study III). The measured sap flow of the study trees was used as a reference and three model versions of increasing complexity were used. The PM model with only a fitting parameter c was the version M1, the previous with an added canopy conductance function (g_c , Eq. 3, Study III; three additional fitted parameters: a , b and g_{Smax}) was the second version M2, and the third version (M3) had also a submodel for leaf area, canopy surface area and within-year leaf area change (L , Eq. 4, Study III; parameter c was replaced by parameter β). These model versions with increasing complexity were used with the measured sap flux in an attempt to find the simplest model with acceptable accuracy. The models were fitted both on an annual level (annual model/parameterization) and over the 2008-2010 data for both sites separately (general parameterization) on half-hourly time step. Least-squares non-linear procedure NLIN of SAS 9.3 was used in parameter estimation.

For the best-fitting PM model version, the residuals between measured and modelled transpiration were compared with the environmental variables (T , I , D and SWC) with multiple regression with backwards selection (Study III). Additional analysis of the environmental factors affecting transpiration was performed with a regression model (Eq. 6, Study III), regressing sap flow to I and D , and the residuals of this regression to SWC.

Weekly sums of tree water use per m^2 of leaf area, so that same days were included for both species, were calculated from sap flow measurements and compared between the species with Kruskal-Wallis test (SAS 9.3, procedure NPAR1WAY) (Study III). To assess whether street tree transpiration could

contribute to urban stormwater management, the annual sum of measured tree transpiration was compared with total annual precipitation (Study III). Estimates of tree transpiration were made for different canopy cover percentages (proportion of land area covered by horizontal projections of tree crowns) by scaling the measured or PM modelled sap flow per PCA.

Soil water content differences between years and sites were analysed as monthly averages for April-October. Non-parametric analysis of variance for repeated measures was performed with Friedmann test (SAS 9.3, procedure FREQ), as if the three measuring locations at each site were chosen at random. The threshold for significant difference was adjusted as appropriate with Bonferroni correction.

The p-value required for a significant difference and/or effect was set at ≤ 0.05 for all statistical analyses.

3.3 ENVIRONMENTAL CONDITIONS DURING THE RESEARCH PERIOD

An overall view of the environmental conditions and weather during the research period is presented as background to help in the assessment of tree growth and function. Additional weather data are presented in Study III (*I, T* and *D* at research sites in 2008-2011, Study III, Figure 2).

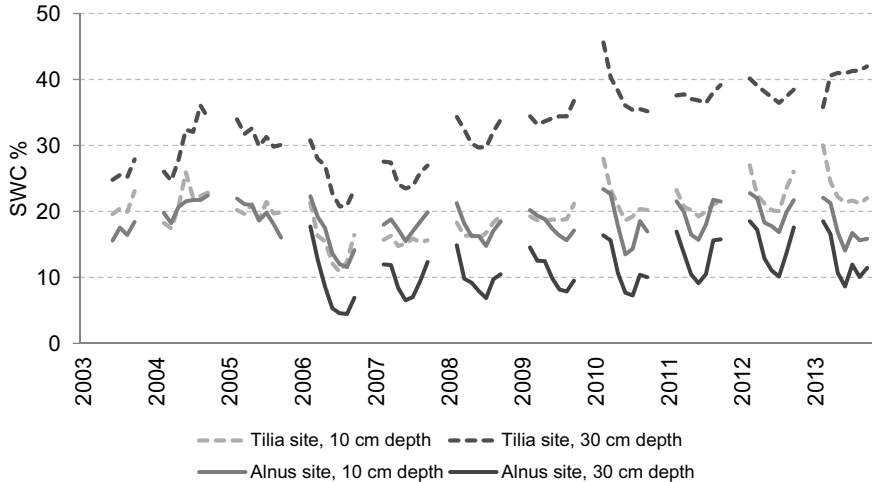


Figure 6. Mean monthly soil water content at Tilia and Alnus sites, at depths of 10 and 30 cm, from April to October in the years 2003-2013. The volumetric water content shown is the average water content in the fine soil fraction of all tested three soils. Measurements at 30 cm depth at the Alnus site began later, in 2006.

Table 1. June-August and annual rainfall (millimetres) in the Helsinki region: at the Helsinki-Vantaa Airport (FMI) located about 11 km NNW, and at the Kumpula SMEAR III station located 4 km SW from the tree sites.

Year	Helsinki-Vantaa Airport		Kumpula SMEAR III station	
	June-August	Annual total	June-August	Annual total
2003	140 (-70)	510 (-170)	n/a	n/a
2004	380 (+170)	840 (+160)	n/a	n/a
2005	280 (+70)	660 (-20)	n/a	n/a
2006	70 (-140)	560 (-120)	n/a	n/a
2007	170 (-40)	760 (+80)	n/a	n/a
2008	230 (+20)	850 (+170)	200	841
2009	240 (+30)	630 (-50)	180	600
2010	120 (-90)	460 (-220)	190	660
2011	180 (-30)	510 (-170)	270	840
30-year average	206	682	n/a	n/a

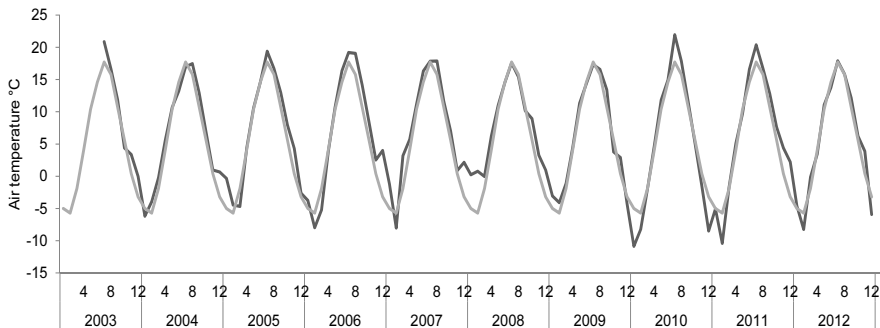


Figure 7. Average monthly air temperature at the research sites from 2003 to 2012 (dark line) and 30-year average monthly temperatures for the nearby Helsinki-Vantaa Airport for comparison (light grey line, FMI, Pirinen et al. 2012).

Rainfall and soil water content can both be used to describe the water availability experienced by the trees. Volumetric soil water contents (SWC) at both 10 cm and 30 cm depths were significantly different between the sites, with higher values at the *Tilia* site. The trees at the research sites were irrigated only in the first two years (2003-2004); year 2004 was also

unusually rainy (Table 1). In 2006, SWC descended to a level not seen before or since (Figure 6). At the Finnish Meteorological Institute (FMI) weather station nearest to the research sites, the Helsinki-Vantaa Airport, July 2004 was the rainiest (200.7 mm) and July 2006 the driest (4.2 mm) within the 30-year period from 1981 to 2010 (Pirinen et al. 2012). The two measuring stations closest to the tree sites, FMI Helsinki-Vantaa Airport and Kumpula SMEAR III, are separated by a distance of ca. 13 km from each other. The considerable differences in both June-August and annual rainfall between the two stations (Table 1) support the use of measured soil moisture as the primary factor in analysis of tree response to water availability, as the rainfall at the research sites was not measured, and it appears that the local variation in rainfall in the Helsinki region is quite high.

Table 2. July average soil and air temperatures at the tree sites and average RH (relative humidity) at the Kumpula SMEAR station compared with the 30-year average at the Helsinki-Vantaa Airport. Deviations from the 30-year average at the Helsinki-Vantaa Airport are given in parentheses.

Year	Tree site soil T °C, 30 cm depth	Tree site air T °C (deviation from 30-year average)	Kumpula air T °C, (deviation from 30-year average)	Kumpula RH % (deviation from 30-year average)
2003	19.4	20.9 (+3.2)	20.6 (+2.9)	70 (+4)
2004	17.3	17.0 (-0.7)	16.5 (-1.2)	75 (+9)
2005	19.8	19.4 (+1.7)	19.4 (+1.7)	64 (-2)
2006	20.4	19.2 (+1.5)	19.0 (+1.3)	55 (-11)
2007	18.4	17.8 (+0.1)	17.3 (-0.4)	72 (6)
2008	17.4	17.6 (-0.1)	17.5 (-0.2)	65 (-1)
2009	17.2	17.4 (-0.3)	16.9 (-0.8)	68 (+2)
2010	20.4	22.2 (+4.4)	21.9 (+4.2)	66 (0)
2011	19.3	21.1 (+3.4)	20.6 (+2.9)	69 (+3)
2012	17.1	17.9 (+0.2)	n/a	n/a

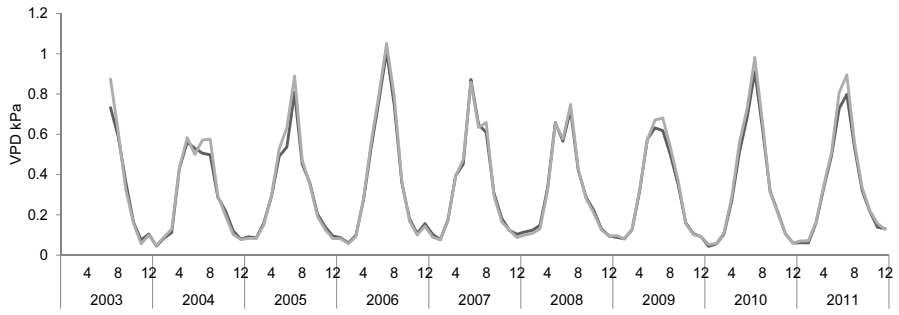


Figure 8. Average monthly D (kPa) at the Tilia (dark line) and Alnus (light line) sites in 2003-2011.

4 RESULTS

4.1 TREE TRANSPLANTING RECOVERY, SHOOT GROWTH AND CROWN ALLOMETRY

4.1.1 TREE TRANSPLANTING RECOVERY AND SHOOT GROWTH

The results up to the year 2008 showed an ongoing increasing trend in *Tilia* annual shoot extension growth (Study I; Figure 9), indicating that transplanting recovery had not yet concluded. Very little shoot growth was evident in 2003-2005, but in 2006 - 2008 improvement was seen primarily in trees growing on soil 2. Time since transplanting was a significant factor in annual *Tilia* shoot extension in 2004-2008, but the current or previous year soil water content had no discernible effect. When the years from 2003 to 2007 were analysed, leaving out the last year in the dataset, the transplanting effect could not be seen, but a negative influence of previous year's high soil water content on shoot growth was revealed. Based on the mixed model (Study I), soil 2 had the highest effect on shoot growth over the entire period, when soil water content was taken into account.

In *Alnus*, the annual shoot growth increased for 3 years after transplanting and thereafter remained at a relatively steady level, apart from lower shoot growth in 2006 (Study I; Figure 9). This indicated that *Alnus* transplanting recovery was mainly over already in 2005, although time since transplanting was significant factor in explaining shoot extension over the entire 2003-2008 period. Annual shoot growth was smaller on soil 1 than on the other two tested soils, which did not differ from each other. In *Alnus* tree shoot growth, soil water effects were significant for the current rather than the previous year; also contrary to *Tilia*, the effect of increasing soil water content was positive rather than negative.

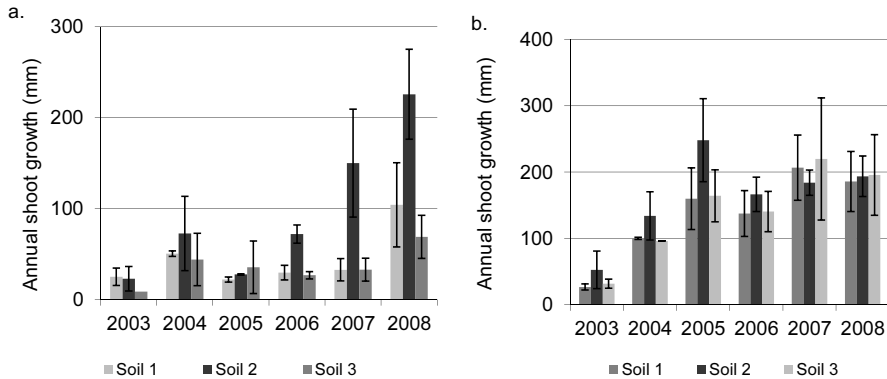


Figure 9. Annual shoot extension of *Tilia* (a) and *Alnus* (b) in the three tested soils during the period 2003-2008. Trees were planted in autumn 2002. The columns represent average annual shoot growth, which was measured from 2 to 4 trees and 30 shoots per tree, and the bars show standard deviation between trees.

4.1.2 TREE CROWN ALLOMETRY: BRANCH BASAL AREA – LEAF AREA RELATIONS

In *Tilia*, the relationship between branch basal area and leaf area was close to linear (Study I). The correlation between ln-transformed branch leaf area and branch basal area varied from 0.85 to 0.97. The relationship between branch basal area and leaf area seemed to tend towards linear, with significant deviations from $\alpha = 1$ seen only in two years. The first year of observations, 2004, was an exception; the relationship was somewhat exponential (Table 3). Another exception was the year 2005, when the relationship leaned slightly towards saturating (α was at its lowest, but not significantly different from 1).

For *Alnus* trees, the leaf area to branch basal area correlation was strengthened as years from transplanting passed (Table 3). Also soil water content had an effect on crown allometry in the early years (Study I, Table 3). Leaf area was smaller on small branches in the dry year of 2006 than in the previous and next years, leading to exponential branch and leaf area relationship. Also the correlation between branch leaf area and branch basal area was at its weakest in 2006. A slightly exponential relationship dominated until 2010, and the deviation of α from a value of 1 was detectable in the dataset in 2007-2010.

Table 3. Linear model ($\ln A_L = \beta + \alpha \ln A_B$), where A_L is the branch leaf area and A_B is the branch basal area, parameter estimates. Values of α marked with * differ significantly from 1 (F test at $p < 0.05$). Parameter values indicated with the same letter do not differ significantly (paired 2-way T-test at $p < 0.05$). Data for years 2004-2008 are also presented in Study I, Table 3.

Year	<i>Tilia</i>				<i>Alnus</i>			
	obs.	α	β	R ²	obs.	α	β	R ²
2004	15	1.43 ^a *	-2.79 ^a	0.92	21	0.90 ^a	-1.17 ^{ac}	0.78
2005	24	0.92 ^c	-2.07 ^b	0.85	26	1.11 ^{ab}	-1.29 ^{acd}	0.92
2006	26	1.04 ^{bc}	-2.12 ^b	0.93	24	1.32 ^b *	-1.98 ^b	0.77
2007	25	1.01 ^{bc}	-1.60 ^e	0.95	32	1.17 ^b *	-1.31 ^{acd}	0.97
2008	20	1.13 ^b	-1.97 ^{bd}	0.94	21	1.15 ^b *	-1.40 ^{ad}	0.99
2009	21	1.13 ^{bc}	-1.74 ^{cde}	0.92	19	1.12 ^{ab} *	-1.42 ^d	0.98
2010	21	1.11 ^{bc} *	-1.45 ^{cde}	0.97	19	1.11 ^{ab} *	-1.21 ^c	0.98
2011	18	1.05 ^{bc}	-2.09 ^{bd}	0.94	18	1.15 ^{ab}	-1.79 ^{abcd}	0.89

4.2 TREE BIOMASS AND SOIL CARBON DYNAMICS

4.2.1 TREE BIOMASS MEASUREMENTS, BIOMASS EQUATIONS AND CARBON ESTIMATES

The woody biomass of the studied *Tilia* trees contained approximately 26 kg C in 2011 (nine growing seasons since transplanting), 18 kg of which had been sequestered at the planting site (Study II, Table 3). About one-third of this was in the root compartment and the rest was divided approximately equally between the branch and stem compartments (Figure 10). The total cumulative C that the trees invested in their leaves from 2003 to 2011 was 12.5 kg. Transferring leaves annually to the leaf litter compartment and applying the decay model (Olson 1963) with a decay parameter of 0.24 (Hobbie et al. 2006) to each cohort gave an estimate of 6.3 kg for the remaining leaf litter C stock, accumulated over the years, as estimated for August 2011. Prunings had only 0.7 kg of C altogether, and 0.6 kg of this remained in 2011.

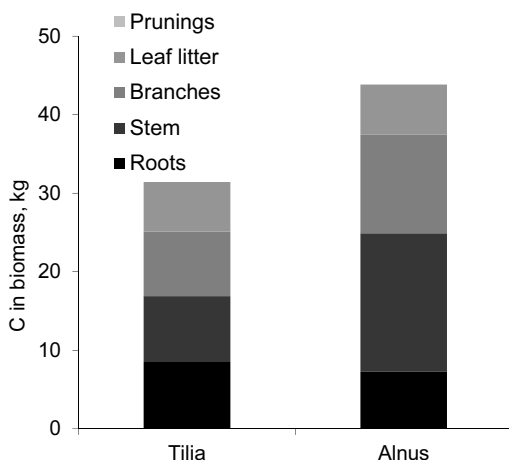


Figure 10. Average total tree C stock by compartments for the studied *Tilia* and *Alnus* trees, based on measurements in 2011 (Table 3, Study II).

For *Alnus*, the C stock in woody biomass per tree was measured to be 38 kg in 2011 (Study II), 32 kg of which had been added after transplanting. In woody biomass C stock, stem C dominated, with a contribution of approximately 50% of the total, while branches contributed about one-third. The relative contribution of leaves to the total C stock estimate was also smaller than in *Tilia*, both because the cumulative C investment in leaves (17 kg by 2011) was about 10% less relative to the woody biomass than in *Tilia*, and because the leaf decay estimated using a

decay parameter of 0.6 (Dilly and Munch 1996) was faster than for *Tilia*. *Alnus* trees needed little pruning, and therefore, both the cumulative and remaining C in prunings was negligible.

The best literature biomass equation (BE) estimates for total aboveground or woody aboveground biomass were often quite close to the measurements (Table 4). When inspecting the different components of aboveground biomass, the BEs performed less satisfactorily than for total aboveground biomass. There seemed to be a pattern such that the BEs underestimated leaf and branch biomass and overestimated trunk biomass.

Table 4. Biomass of the various tree compartments as measured (kg per tree \pm SD, n = 6-12) and predicted from the DBH with biomass equations (average of trees included in biomass measurements) for each taxon in 2011. The number and letter in superscript indicate the equation used (Table 1, Study II). ABW = aboveground woody biomass, AB = aboveground biomass, including leaves. * denotes that the biomass model prediction is significantly different from the measurement results.

	Stem	Branch	Leaf	ABW	AB
<i>Tilia</i> measured	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> predicted	36.1 * ^{3d+3e}	7.5 * ^{3d}	1.4 * ^{3b}	43.2 ¹ , 79.3 * ⁴	47.8 ² 42.0 ^{3a}
<i>Alnus</i> measured	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> predicted	61.0 * ^{5c} 69.3 * ^{6d} 60.7 * ^{7d}	12.8 * ^{5b} 5.3 * ^{6c} 7.8 * ^{7c}	1.0 * ^{6b} 1.8 * ^{7b}	48.0 * ^{5a} 73.7 ^{5b+c} 73.5 ^{6a-b} 69.1 ^{7a-b}	74.5 ^{6a} 70.9 ^{7a}

4.2.2 SOM DYNAMICS OF STRUCTURAL TREE SOIL

Calculated per tree soil volume (the current local standard structural soil volume of 25 m³ per tree, Rakennustietosäätiö 2010) and LOI given by the tree soil manufacturer (or estimated based on soil components), the three different structural soils of the tree sites contained 250-900 kg of C per tree at the time of construction (Study II). LOI measurements in 2011 indicated that more than half of the initial C had been lost on soils 2 and 3, and almost half also on soil 1, with the average C loss across all soils being 290 kg per 25 m³ (Figure 11) and the loss on the entirely peat-based soil 1 around 170 kg on average (6-7 kg m⁻³ since site construction). The C loss model built based on the results of soil sample incubation gave slightly smaller, but still considerable C loss estimates (Study II, Figure 3).

These carbon loss values were somewhat confounded by the observed change in fine soil particle size distribution. The coarse fraction (>2 mm) was more dominant in fine samples after structural soil construction than before, implying that fine soil may have been “diluted” by stone chips originating from the stone matrix. The proportion of the additional stone material in the fraction coarser than 2 mm was on average 25% by weight.

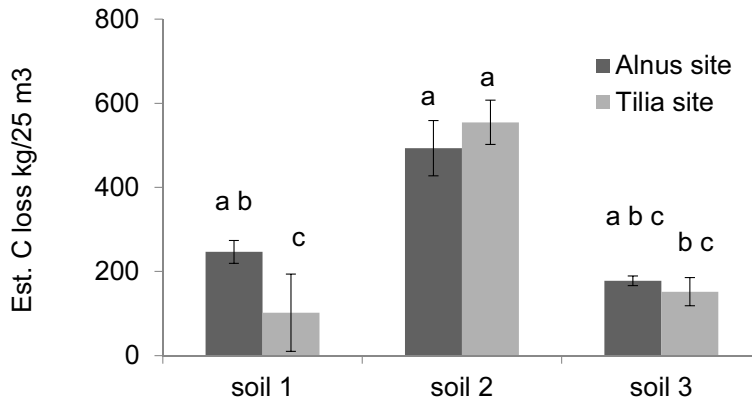


Figure 11. Cumulative C loss (\pm SD) estimated per 25 m³ of the three different tested tree soils from 2002 to 2011, based on the tree soil loss-on-ignition change. Mean values indicated with the same letter do not differ significantly.

4.2.3 STREET TREE CARBON SEQUESTRATION RELATIVE TO GROWING MEDIA C LOSS

The estimates of tree soil C loss and tree C sequestration since planting show that the street trees had sequestered only a fraction of the soil C loss by 2011 (Study II). In comparison with the average C loss from soil per tree (290 kg), the *Tilia* trees had sequestered ca. 5% of this during the period from planting to 2011. Accounting for C in leaf litter brings the tree C sequestration to 8% of the average soil C loss per tree. For *Alnus*, the situation was slightly better; 10% or 13% of the soil C loss had been compensated by tree sequestration by 2011, depending on whether one accounts for leaves or not. Considering only the peat-based soil 1, *Alnus* trees had sequestered about 30 kg of C compared with the 170 kg C loss (average over the two sites, Figure 7).

Predictions were made for tree DBH growth, and based on this, C sequestration was predicted for the first 100 years after transplanting (Figure 8). For *Tilia*, two predictive equations were found (Böckmann 1990, Larsen and Kristoffersen 2002); DBH growth prediction based on the German forestry yield table (Böckmann 1990) gave DBH values closer to the average DBH measured for the sample of Helsinki street trees at given age and a smaller bias (Study II), although neither prediction was significantly different from the measurements. For *Alnus*, the validity of the only available DBH growth model, based on the German forestry yield table for highest fertility site (Schober 1987), could not be assessed, as there were too few and too young *Alnus* street trees available for comparison.

The next step after predicting a DBH for the trees was biomass estimates, which were calculated based on the DBH. Also the C stock in litter and prunings was estimated, the latter based on an ideal pruning programme of the City of Helsinki. The resulting predictions show that the C sequestration

rate of urban trees is fairly low at the beginning; the average soil C loss would be compensated in about 60 years of *Tilia* growth (Study II, Figure 7). The lowest soil C loss in the study was predicted to be compensated in 30 years and highest in about 80 years. For *Alnus*, the prediction was even more uncertain than for *Tilia*, but the attained prediction was rather similar to the *Tilia* prediction (Study II, Figure 7).

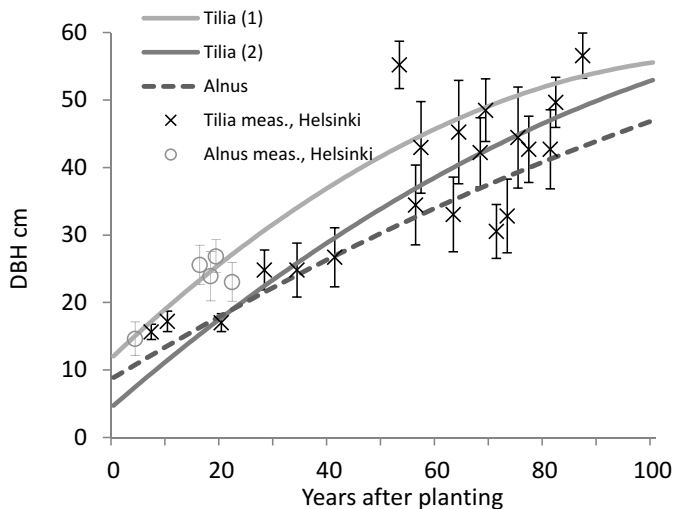


Figure 12. Measured and predicted DBHs for *Tilia* and *Alnus*. The DBH predictions, two for *Tilia* and one for *Alnus*, are based on the literature (see Study II, Table 1). The crosses represent the average DBHs measured from the street tree *Tilia* plantings in Helsinki (\pm standard deviation, SD) plotted against the number of years after planting. The average DBH (\pm SD) of the two known older *Alnus glutinosa* f. *pyramidalis* plantings in Helsinki, one at two time-points and one at three time-points, are indicated with circles.

4.3 TREE WATER USE, ITS DRIVERS AND POTENTIAL IN STORMWATER MANAGEMENT

4.3.1 TREE WATER USE

The water use per tree or per projected canopy area (PCA) was quite different for the two species (Table 5), but the leaf area-based transpiration was significantly higher for *Alnus* only in 2009 out of the four years analysed (Study III). Daily tree water use was at its highest for both species in 2010, about 40-50 l d⁻¹ for *Tilia* and 60-70 l d⁻¹ for *Alnus* (Table 5). On a leaf area basis, the highest average values measured for July-August over the 2008-2011 period were about 0.8 l d⁻¹ for *Tilia* and 1.1 l d⁻¹ for *Alnus*.

Table 5. Daily high-summer tree water use measured for *Tilia* and *Alnus* trees. j_s = Sap flow per trunk cross-sectional area, LA = leaf area, PCA= crown projection area. Mean water use per tree is given for all measurements available in each year in parentheses.

Tree species	<i>Tilia</i>	<i>Tilia</i>	<i>Tilia</i>	<i>Tilia</i>	<i>Alnus</i>	<i>Alnus</i>	<i>Alnus</i>	<i>Alnus</i>
Year	2008	2009	2010*	2011	2008	2009	2010	2011
Time period	9 July - 20 Aug	9 July - 20 Aug	9 July- 30 July	9 July - 20 Aug	3 July - 23 Aug	3 July - 23 Aug	3 July - 23 Aug	3 July - 23 Aug
Mean j_s (g cm ⁻² d ⁻¹)	200	330	340	140	260	510	310	270
Mean per LA (l m ⁻² d ⁻¹)	0.37	0.51	0.78	0.38	0.50	1.09	0.87	0.70
Mean per PCA ± SD (l m ⁻² d ⁻¹)	2.1 ± 0.6	3.4 ± 0.8	5.5 ± 1.9	2.4 ± 0.6	7.9 ± 3.0	13.3 ± 3.5	18.4 ± 3.2	9.6 ± 2.8
Mean water use per tree (l d ⁻¹)	19	32 (28)	49 (41)	22 (20)	29	68 (51)	72 (56)	52 (44)

4.3.2 MODELLING TREE WATER USE

When fitted annually (annual parameterization) to half-hourly sap flow measurements, the prediction by the simplest Penman-Monteith (PM) model version, M1 with only one fitted parameter, had an average R^2 of 0.70 and gave an overestimation of total water use by about 10% (Figure 13 and Figure 14). The attempt to fit the model over all of the examined years at each site (general parameterization) led to a much lower R^2 and gross overestimation of tree transpiration on both sites in 2008 and 2011 (Table 2, Study III). In the simple regression model (Study III, Eq. 6), the explanatory power followed the annual variation in PM M1 model fit (Table 6, Study III); if one fit relatively well in a given year and site, so did the other. Overall, however, the regression model explained some 10% less of the variation in tree transpiration than did the simplest version of the PM model.

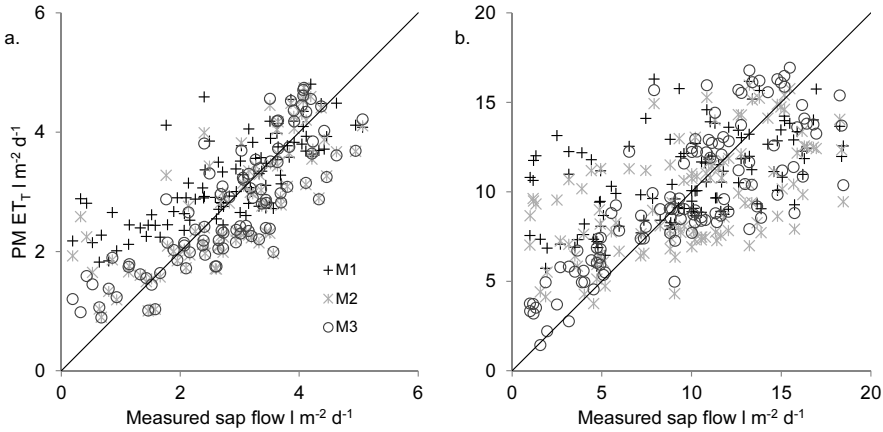


Figure 13. Measured daily sap flow and predicted transpiration ($PM ET_T$) with the three different versions of the PM model (Study II) (annual parameterization) for a) Tilia site and b) Alnus site in 2009. Solid line: 1:1 relationship. Note the difference in scale of the axes.

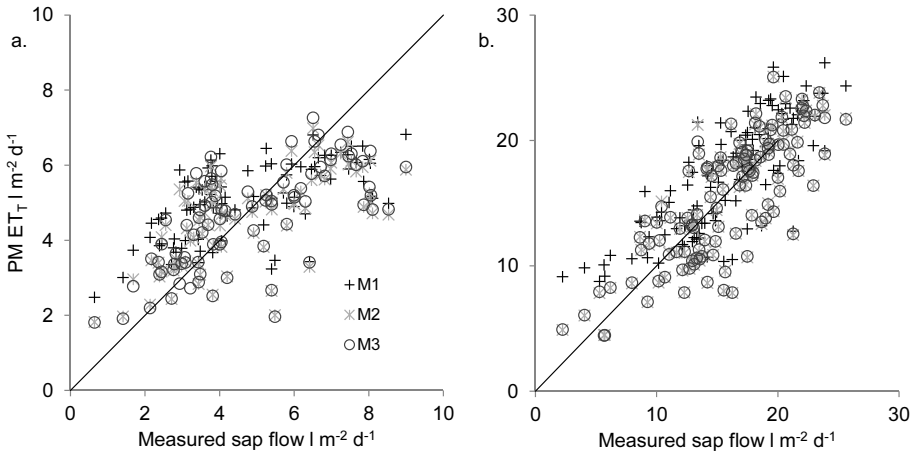


Figure 14. Measured daily sap flow and predicted transpiration ($PM ET_T$) with the three different versions of the PM model (Study II) (annual parameterization) for a) Tilia site and b) Alnus site in 2010. Solid line: 1:1 relationship. Note the difference in scale of the axes.

Changing the constant value for canopy conductance of M1 with light and D dependent model in M2 improved the fit of the model with both general and annual parameterization, more at the Alnus site than at the Tilia site. With the annual parameterization, the bias was rather small with model M2 (Study III, Table 2). Bias in the general parameterization was also somewhat smaller than with M1, but still discouraging. Despite the improvement in model fit, the value of the scaling parameter did not change much, although their interpretation changed; in version M1, the parameter included all

canopy properties, while in M2 canopy conductance was explicitly considered.

The M3 model had within-year leaf area change and crown surface area effects included in the model. This improved the fit of the annually fitted model to an average R^2 of 0.80 (Study III), but understandably had little effect in 2008 when hardly any leaf area change took place during the two-month period for which observations of sap flow were available. The M3 model assumed that transpiration saturates in relation to leaf area index at LAI of 6 (as in Granier et al. 2000), which was exceeded in all years and species, except in 2008 for *Tilia*. Therefore, the additions in the M3 model had little effect on the bias in the prediction of annual sums with general parameterization, in comparison with M2. However, the explanatory power of M3 with general parameterization improved somewhat relative to M2 (Study III, Table 3), mainly via the effect of canopy surface area, which accounted for some, but not all, intra-annual variation in tree transpiration.

4.3.3 ENVIRONMENTAL CONTROL OF TREE WATER USE

The PM model residual correlations were analysed both against variables included (directly or indirectly) as drivers in the PM model and against one variable that was not (SWC) (Study III). Comparing the residual between measured and modelled transpiration ($sf_{meas} - ET_T$) against environmental variables gave overall the strongest correlations on both sites in 2008 (Study III, Table 5), when the data covered only a period of two months in high summer and parameter values were notably different from the other three years. In later years, at the *Tilia* site, T (air temperature °C), I (solar radiation $W\ m^{-2}$) and D (Pa) were rarely significantly correlated with the residuals, indicating that the model described the effect of these variables on tree transpiration fairly realistically. In 2009, the only variable significantly correlated with the residuals was soil water content, which had a fairly weak effect, and in 2010 and 2011, only very weak residual correlations were found. At the *Alnus* site, the correlation with SWC was strong in 2008 and detectable in all analysed years, but also T and/or D displayed significant residual correlations in most years. Similarly, the regression model ($i+e \ln(D)+ f I$; Eq. 6, Study III) explained the observations from the *Tilia* site better than those from the *Alnus* site (Study III, Table 6). Regression model residuals correlated significantly with soil water content at the *Alnus* site in three years out of the four analysed, but at the *Tilia* site only in 2008.

4.3.4 TREE WATER USE AND STORMWATER

Comparing measured or modelled tree water removal (transpiration and canopy interception) in our study against annual precipitation in 2009-2011 with different scenarios of canopy cover, i.e. surface fractions of vegetation, gives an indication of the potential role of street trees in stormwater

management. With a canopy cover of 70%, it was estimated that 60-80% of annual rainfall could be transpired by the *Tilia* trees (Study III, Table 7). For *Alnus*, similar percentage of rainfall was transpired with much lower canopy cover; less than 30% canopy cover was estimated to be required and an amount of water corresponding to average annual rainfall could be transpired with less than 50% canopy cover. Taking an estimate of canopy interception into account (Figure 15) shows that especially for the columnar *Alnus* with its narrow crown, canopy interception adds relatively little to tree stormwater effects. While the different estimation methods (measurement-based, modelled with two model versions and corrected for soil water effects and including canopy interception, or not) shown in Figure 15 gave somewhat different results (see also Study III, Table 7), the differences are relatively small compared with the species difference seen in the study.

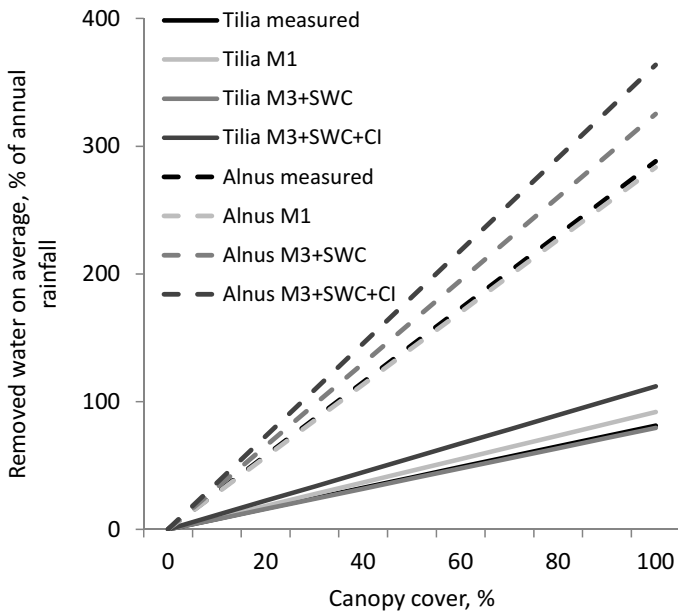


Figure 15. Measured and estimated tree water removal compared with annual rainfall as a function of canopy cover percentage. The estimates were calculated with the least (PM M1 general parameterization) and most accurate model (PM M3 annual parameterization), and the latter was corrected for SWC effect which was not included in the model ($ET_T M3 \times SWC_{high}/SWC_{meas}$). In the fourth estimate, also the estimated canopy interception (37.5% in summer, 27% in winter) was added to the tree water use, because although not measured in this study, tree canopy interception is part of tree stormwater effects.

5 DISCUSSION

5.1 TRANSPLANTING RECOVERY, SHOOT GROWTH AND CROWN ALLOMETRY

5.1.1 TREE TRANSPLANTING RECOVERY AND SHOOT GROWTH

The effects of transplanting on tree growth are fairly well known, although varying tree species and season of planting may confuse the situation somewhat (Richardson-Calfee et al. 2004, Solfeld and Hansen 2004). The results (Study I) regarding transplanting shock longevity and severity for *Alnus* are in line with other transplanting studies (Barton and Walsh 2000, Solfeld and Hansen 2004, Levinsson 2013). *Tilia* recovery was slow, however, and annual shoot length remained low relative to the literature throughout the 2003-2008 period. In a comparative study of transplanting seasons, eight-year-old *Tilia* 'Pallida' trees, clearly smaller than the ones in this study, were transplanted (Solfeld and Hansen 2004). The shoot growth of these trees was, depending on the season of transplanting, 26-30 cm in the first year, 22-59 cm in the second and 33-80 cm in the third year after transplanting. The shoot growth of control trees, left in the nursery, was 53-66 cm. The researchers concluded that trees had reached the pre-transplanting growth rate by the third year. The *Tilia* trees in our study had shoot growth of 12 cm in their last year in the nursery. *Tilia* trees on soil 2 reached and exceeded 12 cm annual shoot length in 2007, and trees on soil 1 came close enough in 2008 to be considered as having surpassed the transplanting shock.

The annual shoot growth of the studied *Tilia* trees remained fairly low, as, in addition to the much higher shoot lengths recorded by Solfeld and Hansen (2004), for example, von der Heide-Spravka and Watson found that average *Tilia cordata* 'Greenspire' shoot extension in Morton Arboretum, Illinois, in 1987-1988 was from 20 to 31 cm. Previous research reporting urban *Tilia* shoot growth after establishment phase contains measurements from varying environments, establishment methods and trees of different ages, but the effects of these factors appear not to be investigated systematically, making it difficult to get an overall picture. In his structural soil study, Kristoffersen (1999) found young *Tilia* × *vulgaris* 'Pallida' height growth in Denmark to vary in different treatments from 9 to 89 cm, with 78 cm growth on topsoil treatment. In the structural soil study of Grabosky et al. (2001), *Tilia cordata* 'Olympic' shoot growth was in the third year after transplanting 29 cm in agricultural control soil and 18 cm in structural soil. Also lower shoot lengths have been recorded; however, in an extensive urban tree survey conducted in 30 towns and cities all around Great Britain in 1989 (Hodge 1991), average shoot growth of the 214 *Tilia* trees surveyed varied from 10 to 12 cm in age

classes from 10 to 50+ years. Shoot growth in our experiment was more comparable to this field survey and trees on less favourable growing media and structural soils than to the research conducted in the more amenable arboretum, nursery, field or topsoil conditions. Also the difference in climate is such that less shoot extension growth may be expected in Helsinki with its cooler climate.

While *Tilia* spp. are common urban trees in e.g. Europe (Sjöman et al. 2012, Pauleit et al. 2002), and consequently, fairly commonly studied, *Alnus* is not a particularly important forestry or urban tree taxa and data are therefore much more scarce concerning its annual shoot growth; no data was found concerning the shoot growth of *A. glutinosa* f. *pyramidalis*. Neither was there any data available on our dataset on shoot growth at the nursery, but the increase in shoot growth after transplanting in *Alnus* was quite clear and also levelled out more clearly than in *Tilia* (Figure 9.), allowing some confidence in judging the transplanting shock length from our data alone. *Alnus glutinosa* is known to be a fast-growing tree relative to other native forest trees in Finland (Miettinen 1933). Young seedling height growth of *Alnus glutinosa*, originating from Finland, has been found to be around 50 cm per year in experimental conditions (DeWald and Steiner 1986).

The differences between tested soils indicated best initial tree growth on soil 2 for both tree species; the differences between the two other soils (soils 1 and 3) were smaller. *Alnus* growth was quite satisfactory on all tested soils with relatively small differences overall, but for *Tilia*, growth was clearly reduced on soils 1 and 3. The reasons behind the good growth on soil 2 were not attributable to any single soil property, as there were many differences between the three tested soils. The most likely reason, however, seemed to be the high original SOM and nutrient content of the waste water compost-based soil 2, and the good growth on that soil is in accord with the numerous studies showing beneficial effects of organic soil amendments (reviewed by Scharenbroch, 2009). Nutrient availability may have been enhanced by the fast decomposition of the plentiful organic matter seen on soil 2 (Figure 11; Figure 2 in Study II).

At the *Tilia* site, which was overly wet, trees reacted poorly to an increase in soil water content, and the effects were revealed not in the current, but in the next growing season. This was most likely due to the pre-determined shoot growth pattern that dominated for *Tilia* in the first years after transplanting. Pre-determined versus free shoot growth may be a factor that warrants attention in studies on tree response to their environment overall; for example, the needle (leaf) count of *Pinus sylvestris* shoots is known to depend on the conditions of the previous years, while shoot length is more determined by the conditions of the current year (Junttila and Heide 1981). There is at least one case (Le Dantec et al. 2000) in which the effect of drought on LAI has been found to be similarly delayed by the same mechanism in a French mixed forest. Lagging effects of environmental variables and extreme climatic events are also suspected to be among the

causes of interannual variation in ecosystem fluxes (e.g. Keenan et al. 2012, Verma et al. 2015).

At the *Alnus* site, all trees had free shoot growth (as opposed to pre-determined growth, for definitions of free and pre-determined shoot growth, see Pollard and Logan 1976) at least from year 2005 onwards, and soil water availability appeared to be a more significant determinant of shoot length than soil type. This may be related to the nitrogen-fixing filamentous bacteria with which *Alnus* trees live in symbiosis, as they could allow the trees a certain degree of independence from soil N resources. Soil water content increase had a positive effect on *Alnus* growth, and the effects were revealed already during the same growing season. Considering how commonly drought is stated to be one of the main problems for urban trees, surprisingly few studies have shown this conclusively, but similar positive response of street trees to increased soil water in limited growing media has been found in Denmark (Bühler et al 2006). Our study adds to this evidence and also shows adverse effects of excess soil water on *Tilia* shoot growth.

In conclusion, the results indicate that for *Alnus* transplanting recovery was quick and all tested structural soil types were satisfactory. Despite the fairly large (15-50 m³) structural soil volumes per tree, soil water content limited tree growth at least to some extent already during the first six years after transplanting. For *Tilia*, tree growth was satisfactory only on soil 2, possibly due to the delayed transplanting recovery, either alone or in combination with excessive soil moisture.

5.1.2 TREE CROWN ALLOMETRY: BRANCH BASAL AREA – LEAF AREA RELATIONS

Trees are long-lived and once established on a site, acclimate to their environment by adjusting their structure and function. The biomass distribution within a tree is a reflection of these structural and functional adjustments over time, in addition to ontogeny, the genetically determined features, and limits of plasticity (Nikinmaa 1992, Ericsson et al. 1996). Tree woody biomass is formed constantly, and a proportion of the woody biomass compartment may be nearly as old as the individual tree itself. New layers of wood are typically added each year outwards from the old layers. Non-woody compartments, such as leaves and fine roots but also often bark tissue, have shorter lifespans, from one to a few years (Kaufert 1937, Howard 1971, Lamppu and Huttunen 2001, Muukkonen and Lehtonen 2004, Pensa et al. 2007, Aro et al. 2013, Hansson et al. 2013b, Gričar et al. 2015). Theories and models concerning tree structure and function commonly state or logically lead to the hypothesis that tree growth allocation results in a situation where the resources are optimally allocated (Nikinmaa 1992, Mäkelä 1997, Lacoïnte 2000). This optimum situation would change according to environmental conditions, with some delay; more so with woody structures than shorter-lived compartments.

In our case study, transplanting had an effect on crown allometry (branch basal area versus leaf area relationship) in 2004-2008 in both examined species (Study I, Table 3). Correlation between leaf and branch basal area improved as transplanting recovery progressed, as may be expected if tree structure becomes more balanced after a disturbance (Mäkelä 1999). On a water-limited site, leaf area to sapwood area ratio has often been observed to be lower than on sites less limited by water resources (Grier and Running, 1977; White et al., 1998, but see also Meier and Leuschner, 2008); this might reflect the need to allocate more resources to woody matter to avoid loss of conductivity or less resources to transpiring leaves. C allocation to foliar biomass should be relatively high when soil conditions are not growth-limiting (Vanninen and Mäkelä 1999, Helmisaari et al. 2007), reflecting less need to allocate biomass to gather soil resources.

In *Tilia*, the observed relationship between branch basal area and leaf area was close to linear (Table 3). The first year of observations, 2004, was an exception; the relationship was somewhat exponential, reflecting the fact that small branches had little leaf area in relation to basal area and large branches had relatively more. This could be a consequence of transplanting, but also shoot growth varied more than in the later years (see SD in Figure 9), which could have contributed to the non-linearity. Transplanting effects tend to show directly as diminished leaf area during the transplanting shock (Harris et al. 1996, Barton and Walsh 2000, Solfjeld and Hansen 2004), but changes in total leaf area could not be detected here because pre-transplanting measurements were not made; the distribution of the leaf area reduction within the tree crown has not been investigated previously. Due to the delayed transplanting shock and non-linear recovery in *Tilia*, it may be expected, however, that aside from the transplanting shock itself, other possible factors affecting leaf and branch basal area relations were likely to be confounded and undetectable.

For *Alnus* trees, in addition to improving leaf area to branch basal area correlation as years from transplanting passed, also soil water content was observed to have an effect on crown allometry. Compared with other years, leaf area was low in 2006 on small branches, leading to an exponential branch and leaf area relationship. As the measurements were performed relatively late in the growing season (in August), this was likely due to preferential shedding of leaves from smaller branches during drought; premature shedding of approximately one-quarter of leaves was observed between July and mid-August in the *Alnus* trees in 2006 (Study I). The smaller leaf area of small branches in relation to basal area during drought may reflect either the more exposed position of smaller branches, located mainly in the upper crown, or vulnerability segmentation (Tyree et al. 1991, 1994).

It also appeared that temporarily poor shoot growth might be reflected in a saturating-type branch basal area to leaf area relationship; for *Alnus*, the only year with linearity parameter α value below one was the weakest growth

year in the dataset, 2004; for *Tilia*, the year 2005 stood out with a similar situation. Such an effect would be logical: the branch basal area reflects tree growth in the long term. The larger the branch, the longer the period it has accumulated sapwood. Leaf area, by contrast, is mostly the result of the current and previous years' success in gathering growth resources (Clements 1970, Pollard and Logan 1974, Le Dantec et al. 2000). A sudden decrease in shoot growth would thus lead first to linearity parameter value below one, but if poor growth prevailed for a longer period, the effect of each year would accumulate on woody biomass, and thus, the branch diameter would correspond better to the leaf area. This would result in a return towards linearity, assuming a linear relationship is the functional optimum of biomass distribution between a branch and its leaves.

In *Tilia*, the relationship between branch basal and leaf area seemed indeed to tend towards linearity after disturbances. This is a hypothesis (Mäkelä 1999) that has not yet been rigorously tested. The dataset collected in this study can offer no conclusive evidence, but the data for 2004-2008 in Study I do not contradict this hypothesis. Analysis of the data on *Tilia*, collected from the same trees with the same method for 2009-2010 supports the hypothesis further; the trend appears to continue towards linearity (Table 3.). For *Alnus*, however, the deviation of α from 1, while rather small, was quite dominant in the latter years. This may be due to recurring limiting water availability on the site (Study III) or be related to the columnar crown form and nearly vertical branch orientation of the *Alnus* trees (see Figure 2). The amount of sapwood required to support leaves depends partially on branch angle (Morgan and Cannell 1988). Also, as the branches grow thicker, they tend to attain more of the properties of the main stem such as a higher leaf area to sapwood area relationship (Nikinmaa et al. 2003, Berninger et al. 2005).

As a method of leaf area estimation, the development and application of branch basal area/leaf area relationship were not tested against other methods. A comparison of this method with a destructive measurement of leaf area was performed by Gilman and Grabosky (2009), showing that it was accurate and reliable; while leaf biomass varied somewhat with compass direction, leaf area did not. The development of the branch basal area to leaf area relationship is time-consuming, however, limiting its use somewhat. The method has found an application in determining pruning dose in tree pruning research (Gilman and Grabosky 2009, Gilman 2015), where good accuracy in leaf area determination on an individual branch basis is needed.

5.2 TREE BIOMASS AND SOIL CARBON DYNAMICS

5.2.1 TREE BIOMASS MEASUREMENTS, BIOMASS EQUATIONS AND CARBON ESTIMATES

5.2.1.1 *Tree biomass and biomass equations*

The measured total biomass C stock and its distribution within the studied trees differed between the taxa. The relative dominance of branch biomass in *Tilia* compared with *Alnus* may be due to the columnar crown form of the black alder trees (var. *pyramidalis*). Branches of the columnar *Alnus* are directed upwards, almost vertically, while the *Tilia* branches are more horizontal, which can also be seen in the larger projected canopy area of *Tilia* trees despite their smaller overall size (Table 1, Study III). Supporting branches on a near horizontal plane requires thicker branches than a more vertical branch angle (Morgan and Cannell 1988). High branch biomass may be linked to high leaf biomass; the LAI of the columnar *Alnus* trees (Table 1, Study III) far exceeds the LAI of native deciduous stands, usually about 8-10 at the highest (Dufrêne and Bréda 1995, Wulder et al. 1998, Le Dantec et al. 2000, Soudani et al. 2006). Comparing the LAI of open-grown single tree to a forest LAI is not straightforward, but the underestimation of tree leaf biomass by BEs indicates relatively high leaf biomass in the studied *Alnus* trees as well.

The unusual crown shape of the *Alnus* trees is likely at least partially responsible for BEs consistently giving rather poor estimates for *Alnus* leaf and branch biomass, but the same issue with BEs was seen in *Tilia* as well (Table 4). The main cause seems to be the open nature of the tree sites; the case study trees do not compete for light, a phenomenon which is managed in industrial forestry by adjusting stand density to increase biomass partition to timber, i.e. tree trunk. It is well known that tree crowns tend to be wider in open grown trees than in stands (Hann 1997, Bechtold 2004), but how this is reflected on biomass relations has received little interest to date. The research by Zhou et al. (2014) and, albeit less directly, also the study of Ares and Brauer (2005) support the finding (Study II) that for open grown trees, the biomass distribution tends to emphasize stem less than in forests. If the relatively high proportion of biomass in branches for open grown trees can be confirmed, it implies more leaves in open grown trees as well, and adds weight to the potential of leaf litter and prunings in adding to the total C sequestration of urban trees. Another factor that could contribute to the high branch and leaf biomass relative to trunk biomass in the studied trees is the somewhat young age and small size, as the proportion of the trunk tends to increase as trees grow (e.g. Mäkelä and Vanninen 1998, Litton et al. 2007). Overall, only two species of trees from two sites were studied, and conclusions concerning urban trees in general cannot be made, but combined

with findings in the literature, our results indicate that biomass distribution in street trees may differ from forest trees. Thus, using traditional non-urban BEs for street trees may lead to considerable errors in estimating specific aboveground biomass compartments. The total aboveground biomass predicted by BEs was surprisingly close to the measured, but it may in part be explained by the small size of the measured trees; the different BEs used diverge from each other much more when applied to larger DBHs.

The DBH has long been the most easily measured descriptor of tree size, although the advent of remote sensing may find other descriptors to be more easily obtained (Holopainen et al. 2013, Tanhuanpää et al. 2014). BEs, used to estimate biomass from DBH, vary in form, but the relationships described are most often log-normal, and consequently, most equations follow this form. Return to normal scale introduces a bias when the variance in the original dataset increases as the average increases (Baskerville 1972), as is typical in biomass measurements. Methods available to correct this bias were reviewed and compared by Clifford et al. (2013). Estimated from the data collected by Ter-Mikaelian and Korzukhin (1997), containing equations for more than 30 broadleaved tree species, the correction factor adds on average 1% in the whole tree aboveground biomass, and 10% in branch and leaf biomass. Of the BEs used in Study II, only one had an accompanying correction factor. Assuming the other equations produce underestimates in the magnitude previously mentioned, correcting for it does not improve the estimates notably compared with the measured values for each biomass compartment, as can easily be seen in Table 4.

Biomass measurement of aboveground tree compartments is quite easy when destructive sampling is possible. When it is not, as in Study II, numerous uncertainties are introduced into the estimates. Trunk biomass was calculated based on volume measurements and specific gravity for a related tree species (Alden 1995), as values for the species in question were not found. Additionally, a conversion factor for dry biomass to carbon (0.45) was used throughout instead of measurements of biomass carbon content. For branches, a relationship between leafless branch biomass and its basal diameter was developed based on sample branches from the studied trees. The samples were not, however, collected randomly; the branches in the measured population originated from formative and maintenance prunings. As a consequence, large branches (diameter >20 mm) were either underrepresented (*Tilia*) or absent (*Alnus*) in the datasets. Leaf biomass data was collected with a similar approach as the branch biomass data; the leaves from branches of various diameters were collected and a relationship between leaf area and branch basal area was formulated (Study I). The sampling of the branches for measurements was based on taking the median size branch for upper, middle and lower crown, which left out the largest and smallest branches. A method of this type for leaf area and biomass estimation has been typically used for fairly small trees (Nygren et al. 1993, Gilman and Grabosky 2009), but as such, appears to be reliable. The possible leaf and

branch biomass estimation problems in Study II arise mainly from the selection of sample branches – leaf area was calculated for larger branches than the largest present in the sample upon which the relationship was based; a problem that could be easily corrected. In the present study, however, the aim was to make comparisons over time on the same trees (Study I), and changes in the sampling method would have risked confounding the results.

5.2.1.2 Root biomass

Estimating root biomass of urban trees is problematic, both because little knowledge is available concerning e.g. root system structure and root distribution, and because it is difficult to propose a practical and representative sampling strategy. The measurement-based estimates of root biomass in Study II are a good example of problems faced in sampling urban tree roots and the resulting high variation and uncertainty. Understanding of urban tree roots and root systems leaned initially strongly on research on native forest trees, as exemplified by Perry (1982), but factors such as limited rooting volumes and soil sealing are very likely to render such information poorly applicable. Unfortunately, studies of urban tree roots, e.g. root system spread and depth, are rare (Gilman 1989, Krieter and Malkus 2000, Day et al. 2010, Watson and Hewitt 2012). The increasing availability of ground-penetrating radars capable of identifying large tree roots (Bassuk et al. 2011) offers hope for progress in this area. Understanding fine root (< 2 mm diameter, Gill and Jackson 2000) dynamics and e.g. root litter production in the urban environment are currently more distant goals.

In forest ecosystems, fine roots contribute typically less than 5% of total tree biomass, but due to the short lifespan of fine roots their expense in terms of annual primary production is much higher (Nadelhoffer et al. 1985, Vogt et al. 1995). Thus, in terms of biomass C stock, fine roots contribute relatively little at a given moment, although abundant resources are allocated to their generation, similarly to leaves. To understand urban soil C stock dynamics, the gaps in knowledge about urban root litter production and other root-related C inputs into the soil must be filled. In boreal forests, the fine root longevity of deciduous birch (*Betula* sp.) is known to be about two to three years (Aro et al. 2013, Hansson et al. 2013 b). This implies that if fine root longevity is similar for other deciduous trees and in urban soils, the impact of root litter on the C stock of the soil in the limited rooting volume over time should not be overlooked. In our study, root-related C outputs to the soil are added to the soil C stock, and thus, taken into account in the assessment of the tree and tree soil carbon dynamics. It would benefit the understanding of urban tree C sequestration to be able to measure fine root dynamics directly. As an example, if a tree with DBH of 15 cm and total biomass of approximately 70 kg (as measured for case study *Alnus* in 2011, Table 3, Study II) is allocated 5 m³ of soil, then the root litter input, estimated as 5%

of total root biomass per year (Hansson et al. 2013a) would add about 150 g of SOM per year per cubic metre of soil. Assuming urban tree soil depth of 1 m, this input is within the range measured for boreal forests (Leppälammikujansuu 2014).

There are some reasons to suspect that boreal urban tree fine roots may have shorter lifespans than their counterparts in forests, which might increase the fine root litter input into the urban soils relative to natural forests. Soil moisture conditions experienced by urban trees appear to vary rather strongly from dry to wet even in wet years (Nielsen et al. 2007), and it has been suggested that drought increases the fine root mortality (Meier and Leuschner 2008), although conflicting results have also been presented (Leuschner et al. 2004, Zeleznik et al. 2015). There is some evidence that water logging decreases root biomass in relation to total biomass, possibly also attributable to increased mortality (Poorter et al. 2012). Additionally, higher soil temperatures found in urban areas (Savva et al. 2010, Müller et al. 2014, Miyajima et al. 2015) may increase root turnover (Pregitzer et al. 2000, Leppälammikujansuu et al. 2014).

In attaining the root biomass values for the case study sites, in addition to the actual sampling with its own uncertainties and sources of error, assumptions had to be made concerning the spread of the root system, and thus, the biomass estimates attained are truly merely estimates. It was assumed, based on a meta-study by Day et al. (2010), that the tree root system radius would be around $38 \times \text{DBH}$. This exceeded the tree soil dimensions already in 2008, so the root system was assumed to cover the entire allotted tree soil area. Root samples were collected from sampling pits that were located 2-3 m away from the nearest trees; the root density found in these was assumed to represent the average density within the entire rooted area. With these assumptions, root biomass values were gained that were fairly close to literature estimates of average root contribution to total biomass (Cairns et al. 1997, Chojnacky et al. 2014), but between-tree variation was great (as demonstrated by the SD values in Table 3 of Study II). Consequently, the reliability of the root biomass estimates is the poorest among the tree biomass compartments that were measured in the study, and it is clear that root carbon dynamics of urban trees are in urgent need of further investigation.

5.2.1.3 Litter and prunings

While the C stock in the tree woody root, branch and stem biomass are indisputably valid components of urban tree C sequestration, the considerable proportion of C invested annually in leaves and fine roots and then discarded as litter is more difficult to categorize; especially the final destiny of leaf litter in urban areas is uncertain. Carbon in urban tree leaf litter is unlikely to enter the soil C pool due to soil sealing (Zong-Qiang et al. 2014). In Study II, the potential C stock remaining in leaf litter was

calculated with a litter decay equation (Olson 1963) and decay factors for each species (Dilly and Munch 1996, Hobbie et al. 2006). The resulting estimates of remaining leaf litter C over all leaf cohorts produced by 2011 implies that for *Tilia* with its higher relative C input in leaves annually and more decay-resistant leaves, the litter C stock could potentially be fairly important, at least while the trees are still young. In older trees, this might matter less; according to most BEs, the leaf biomass of deciduous trees is not as strongly exponential as a function of DBH than the whole tree aboveground biomass, thus, the proportion of leaves tends to decrease as trees grow. Based on crown geometry this is likely true also for street trees, which appear to have different aboveground biomass distribution overall compared with non-urban forest trees, but no empirical evidence is yet available.

It must also be noted that the potential remaining prunings and litter C stocks in 2011 (Study II) are also mere estimates; the applicability of the literature-derived litter decay factors in the urban situation is uncertain. Studies of urban versus rural originating leaf litter decomposition (Carreiro et al. 1999, Pouyat and Carreiro 2003, Pavao-Zuckerman and Coleman 2005, Nikula et al. 2010, Dorendorf et al. 2015) and studies transplanting rural litter in urban and urban litter in rural environments (Nikula et al. 2010, Pouyat et al. 1997) have shown contradictory results. These imply that the factors determining the rate of decay are not yet identified satisfactorily. Perhaps more importantly, however, leaf litter is likely removed from the tree and tree soil system, and it can be questioned whether it should be accounted for at all. In Study II, possible effects were calculated; the results imply that urban leaf litter has the potential to influence the C stock and sequestration of urban trees. Whether this potential can be realized depends on site design and the handling of urban greening waste. The perpetual soil quality issues affecting urban trees could to some extent be alleviated if the litter cycle at the tree site could be enabled, e.g. by planting shrubs or perennials under trees to retain leaf litter at the tree site.

In the biomass measurements, the amount of biomass removed in prunings was surprisingly low even for *Tilia*, which was pruned more heavily. This implies that prunings had little effect on tree aboveground C sequestration, independent of their final destiny. The decay rate for branches was derived from boreal forest floor twig and branch litter (Perruchould et al. 1999), but tree prunings are most commonly chipped and used as mulch. In a study by Duryea et al. (1999), the decomposition rate of utility pruning mulch was somewhat faster, resulting in a decay factor of approximately 0.4 in the first year. The carbon in fruit, cone and branch litter was not accounted for, but the majority of branch litter would be included in prunings because dead branches are routinely removed when present at pruning time.

5.2.2 SOM DYNAMICS OF STRUCTURAL TREE SOIL

5.2.2.1 *Soil carbon loss estimates*

Rather high carbon losses were seen in the tree soils of the case study sites in the first years after construction (Study II). Compost-derived soils 2 and 3 lost the most carbon, while soil 1 had entirely peat-based organic matter and lost the least in relation to initial C content. There was some uncertainty with regard to the sampling of structural soil pre- and post-construction. A notable change towards coarser particle size distribution was observed in the fine soil fraction post-construction compared with the original status. An addition of gravel to the fine soil fraction was found and assumed to originate from the stone matrix. The relative volume of fine soil compared with stone matrix cannot be measured after construction with current methods (e.g. Bühler et al. 2016), so the addition to the fine soil volume could not be ascertained or quantified. It seems possible that additional coarse material had “diluted” the fine soil LOI. The LOI used in the LOI loss calculation was determined for particles smaller than 2 mm in diameter and not corrected to the total sample weight, in an attempt to partially compensate for this source of error. Assuming similar addition to the materials finer than 2 mm as observed in the coarser fraction, the LOI loss estimates would be 20% lower, i.e. about 230 kg across all soil types by the year 2011.

While the incubation model did support the LOI loss-based estimates, the problems with the incubation model were in part similar to the LOI loss method issues. The same uncertainties concerning soil sample particle size shift and unknown exact proportion of fine soil in the structural soil apply, as the calculation of total loss was based on the fine soil proportion of the total volume. The sample incubation was performed with loose soil samples of ca. 15 g spread out on the bottom of 120 ml glass bottle, giving better gas exchange between the soil sample and free air than would be expected in the field. In the incubation, an increase in soil water content always increased the CO₂ output, which may be unrealistic in field conditions. Additionally, the LOI measurement-based C loss estimates and especially the parameters estimated for the incubation-based model show considerable site differences within the same soil type. These may to some extent reflect the reality, as site differences, especially the different tree species and long-term soil water status difference between sites, may have led to differences in soil fauna and flora (Williams and Rice 2007, Bomberg and Timonen 2009) as well as different SOM decomposition rates. To some extent, the differences may also be an artefact produced by the rather limited soil sampling, and consequently, the averages over both sites and all examined soil types are primarily used as a basis for discussion. Combining data from different sites already in the incubation model parameterization was not considered a viable option, as it would have brought about heteroscedasticity problems with the model residuals (Study II, Table 2).

The estimates concerning the soil C loss acquired with direct LOI change measurements and the incubation model also diverge somewhat, but they support each other well enough that they can be taken as an indicator of the magnitude of initial C loss from C-rich artificial growing media in the urban environment. If the origin of the C in the growing media is not short-term biogenic cycle, the concern is certainly more serious, but the status of peat in this respect is not universally agreed on (IPCC 2006). Indications of high soil CO₂ efflux from or induced by organic soil amendments were found in urban Boston (Decina et al. 2016), showing that improved understanding of soil-derived urban CO₂ fluxes is needed. In Finland, urban soil CO₂ fluxes have been found to be rather similar to those from forests (Vesala et al. 2008, Järvi et al. 2012); the study by Decina et al. (2016) indicates that management is critical for soil CO₂ emissions, which could well explain the difference.

5.2.2.2 Urban soil and growing media carbon stocks

The amount of SOM used in urban greening soils is traditionally fairly high in Finland (Rakennustietosäätiö 2010) compared with, for instance, the usual C content of mineral forest (Liski et al. 2006) or agricultural soils (Heikkinen et al. 2013). Intentional C input into soil is typical in both agriculture and horticulture, and also in landscaping as a part of the latter. The ultimate aim is enhanced plant growth, brought about by the improvement that an organic matter increase causes in the soil properties (Reeves 1997) and/or by the nutrients released from the organic matter as it mineralizes. In undisturbed, well-drained boreal forests, highest C content is typically found on the topmost organic soil layers (Deluca and Boisvenue 2012). In urban soils, it has been noted that C-rich materials end up in deeper layers than in native soil formation (Pouyat et al. 2002, Lorenz and Kandeler 2005, Bae and Ryu 2015). Also the Finnish urban greening soil recommendations stating e.g. 60 cm deep beds for large perennials and 100 cm deep beds for trees (Rakennustietosäätiö 2010) will lead to additional C in deeper soil layers. Research is needed both for optimizing urban soil C storage, avoiding any possible methane emissions, and for understanding how the important properties of soil, such as water holding capacity, may change over time as a consequence of soil C content change. On the other hand, the long dominant undervaluation or omission of soil C stocks from C stock estimates in urban environments (Edmodson et al. 2012, 2015) is more easily understood considering that the apparent organic layer is often entirely missing in most heavily urbanized soils.

5.2.2.3 The peat question

While soil 2 differed from the others with its much higher C content and loss, it also had the highest initial C content and the C loss varied less between soil types when examined as a proportion to the original C content of each soil type. The SOM in soil 2 was partially compost, partially peat-derived, but the proportions of the C sources were not studied and could not be determined after site construction. The slightly slower C loss from the entirely peat-based soil 1 is not surprising as such (e.g. Prasad and O'Shea 1997; Prasad and Maher 2003). If C originating from peat remains in the growing media longer than C originating from compost, does it support the use of peat in urban tree growing media? The stability of soil properties might well be better with the more slowly decomposing peat than with compost-originating SOM source alternatives. On the other hand, peat may be considered a non-renewable C source, on par with fossil fuels. Like coal, natural gas and petroleum, peat is a biogenic, yet largely non-renewable C source on the time scale relevant to human interests.

When left undisturbed, a boreal peatland, from which the peat used in Finnish urban greening growing media originates, has a lower rate of peat decomposition than formation in the long term, and thus, it acts as a C sink (Turunen et al. 2002, Vasander and Kettunen 2006). Compared with other C pools in the biosphere, peatlands are among the largest and most stable (Blodau 2002). While peat used in energy production is clearly classified as causing net C emissions (IPCC 2006), the ultimate destiny of peat-originating C in growing media is not as easy to determine, and thus, choosing the correct C accounting principle for peat in growing media remains an open question (Defra 2009). Depending on the calculation methods and assumptions made (e.g. Defra 2009), the estimated life-cycle C cost of peat varies greatly. In most calculation methods, CO₂ released from short-cycle, renewable biogenic sources (e.g. compost) does not count towards climate change, and full end-of-life cost is applied to peat (Ahlgren et al. 2015); i.e., none of the C originating from peat in the growing media is thought to remain in soil C stock. These two assumptions may emphasize the negative effects of peat use, but whether they reflect the reality and to what degree are difficult to judge at present. Based on the current knowledge and the finding that compost-based growing media produces similar tree growth as peat-based media (Study I), it appears a reasonable application of the precautionary principle to recommend leaving peat out of street tree growing media.

Research on replacing peat in e.g. nursery container growing media with other organic but renewable materials, often industrial by-products such as coconut coir, and various recycled materials such as composts, has been intensive in the last decades (e.g. Alexander et al. 2008, Schmilewski 2008, Caron et al. 2015). Unfortunately, peat is a common compost feed stock additive (Himanen and Hänninen 2011), so compost-based products may contain some fraction of peat-derived C. However, the existing interest in

peat replacements in nursery and greenhouse growing media gives a head start in research for peat alternatives for urban tree growing media development as well. Similarly to urban tree growing media, the stability of the structure and properties of the growing media over the production cycle is important in greenhouse and container nursery production. Thus, a fair amount of information on the behaviour of different compost-based products is available (reviewed by Carlile et al. 2015). Compost maturity has a large effect on its stability, meaning that with less mature compost faster mineralization can be expected (Bernal et al. 1998, Griffin and Hutchinson 2007, Rigby et al. 2016). However, compost mineralization is greatly complicated by the complex ecosystem of decomposers, varying composition of compost products and environmental conditions such as water content and temperature, making it difficult to predict the situation in the landscape from a stability and plant nutrition point of view. Newly available, relatively stable, renewable SOM materials, such as biochar (Scharenbroch et al. 2013, Andreasson et al. 2014) and clay-stabilized growing media materials (Bolan et al. 2012), may become viable options in future planting soils.

5.2.2.4 Long-term development of tree planting soil C stock

With respect to Study II and more generally, the future development of artificial tree planting soil C stock remains uncertain; little research on urban tree soil carbon dynamics is available (Beyer et al. 1995, Pouyat et al. 2002, Jenerette et al. 2006, Lorenz and Lal 2009), especially over the long term. Possible CH₄ and N₂O emissions from urban soils are not yet known (Beyer et al. 1995, Livesley et al. 2010), particularly in relation to soil sealing. Thus, whether tree soils would make a good long-term deposit to store biogenic carbon cannot yet be determined, but the cycle of urban zoning and re-zoning and the life expectancy of urban infrastructure (Facanha and Horvath 2006, Aktas and Bilec 2012, Buyle et al. 2013, Helsingin kaupunki 2013) may be too short to allow such considerations. Increasing attention to soil resource use and recycling can be seen in the Finnish legislation (e.g. Valtioneuvoston asetus 179/12) and may lead to re-use of tree soil materials in the future, with additional impacts on the life expectancy of urban soil carbon. Potentially, unless re-zoning and infrastructure renovations interfere, the lifespan of urban tree soil is as long as or longer than that of the urban tree, as it is common practice to plant replacement trees in the existing tree soil.

In more general terms, relatively little is known about urban soil C stocks development in the long term, and what is known is rarely applicable to boreal environments or to a small scale such as individual tree soil. There is some evidence, however, that as time elapsed from soil disturbance of various types increases, so does the urban soil C content (Scharenbroch et al. 2005, Park et al. 2010, Howard and Olszewska 2011). This, together with the observed slowing down of soil C loss in the case study sites gives some hope

for urban tree planting C sequestration. In an arid urban ecosystem, it has been observed that after the land use change, C accumulated into the soil exceeded native soil C levels, likely because of irrigation and fertilization (Golubniewski 2006; see also Jenerette et al. 2006). In boreal environment, urban vegetation is typically given little to no additional resources, but unintentional resource additions (Kaye et al. 2006, Pickett and Cadenasso 2009) are likely. Moreover, it can be inferred from agricultural experience that the lack of tillage in urban planting soils is likely to be a positive factor for the development of soil C stocks (Lal 2004). The selection of tree species appears also to be of importance in determining urban tree soil C stock development (Edmonson et al. 2014).

5.2.3 STREET TREE CARBON SEQUESTRATION RELATIVE TO GROWING MEDIA C LOSS

In the case study (Study II), the tree carbon sequestration was calculated from biomass measurements in the first 10 years after planting and compared with tree soil C loss over the same period. Despite the possible errors in estimation, the magnitude of the difference between soil C loss and tree C sequestration indicates clearly that within the first decade the street tree plantings in the case study acted as a carbon source rather than a sink. While the tree C sequestration over such a period after transplanting is likely to be relatively low elsewhere as well, the C loss originating from the tree soil in the case study might not be relevant in many cases. The attribution of soil C loss to the tree planting is less disputable in the case of peat-based growing media. The waste-derived fraction of C in the compost-based soils can be considered renewable and not relevant in C accounting. The use of peat as an additive (bulking agent) in the composting process confuses the matter somewhat, but alternative, renewable feed stock additives have been studied (e.g. Barrington et al. 2002, Manios 2004, Dias et al. 2010) and may become more attractive to Finnish compost producers as environmental awareness increases. In areas with lower land use pressure and when massive soil construction is not necessary, trees may be planted without additional C expenditure in the tree soil. If a tree planting is likely to be short-lived due to e.g. a high likelihood of mechanical damage and growing media containing peat is used, there appears to be a distinct possibility that the tree planting will not bring about any net C sequestration benefits.

As the soil C loss at the case study sites was still much higher than tree C capture after the first decade, tree C sequestration was also predicted for the first 100 years after planting. This approaches the maximum lifespan of the oldest *Tilia* street trees in the oldest planted streets currently remaining in Helsinki (Peurasuo et al. 2014). These trees are, of course, only a fraction of the originally planted trees and appear to represent the lifespan that *Tilia* street trees can attain in the current conditions rather than the average lifespan, which can be estimated (based on Pauleit et al. 2002) as comparable

to the more general figures of 20-30 years reported in the literature (e.g. Roman and Scatena 2011).

Predicting tree DBH growth, and consequently, tree C sequestration, is rather difficult (Peper et al. 2014), as can be seen in the measured Helsinki *Tilia* trees (Figure 8). Tree growth rate and resulting tree size at a given age are the main determinants of urban tree C sequestration of a given tree species; thus, to bring the estimation of urban tree C sequestration from the current tree inventory-based “snapshots” in time to a more dynamic approach depends on being able to predict tree growth. Urban tree mortality is another confounding factor in attempting to predict urban tree C sequestration, and any model predicting urban tree C sequestration on a population level should strive to include tree mortality and replacement C cost.

The examined sample of *Tilia* plantings in Helsinki (Study II) was relatively small and plantings were not included randomly, rather the plantings were selected based on the availability of information about planting year. Another selection criterion was sufficient number of trees remaining from those originally planted. This was assessed visually based on tree size and evenness of tree size in rows. The problems with this visual identification of replacements unfortunately link back to the size versus age relation that was under assessment in the collected data. Thus, it is possible that the dataset is biased towards a bigger tree size. On the other hand, the collected data are not likely to underestimate tree size as a function of age, which would be the more critical bias for the aims of the study. The data also shows quite clearly that the bigger the trees are, the more variation in tree size as a function of age can be expected; it was mainly due to the variation in measured trees that neither model used to predict *Tilia* DBH (Böckmann 1990, Larsen and Kristoffersen 2002) produced results clearly different from measurements (Study II). Variation in DBH reflects strongly in biomass estimated from DBH with BE, because BEs are exponential rather than linear functions of DBH.

In comparing measured tree leaf and branch biomass with BEs, it was found that BEs produced underestimates of both of these biomass compartments, and no equation produced satisfactory results. Consequently, the estimates of new leaf litter and pruning biomass subjected to decay equations each year may be too low as well, leading to underestimation of the potential role of prunings and leaf litter in tree C sequestration. The effects of pruning itself on tree biomass relations are not fully understood (Zeng 2003), and considering the constant need to prune street trees it indicates an area warranting further research, both to develop more effective long-term pruning strategies and to improve street tree biomass estimation. Some aspects of pruning in relation to biomass partitioning have been investigated in relatively young trees (e.g. Ares and Brauer 2005, Gilman et al. 2006), but applicability of these results to mature trees in the landscape remains an open question.

5.2.3.1 Additional C expenses of street tree planting

In the estimates of tree and tree soil carbon stock dynamics in Study II, several tree planting-related C costs were ignored, e.g. soil and site construction and transport of trees and growing media. These can be expected to push the compensation point even further into the future. Strohbach et al. (2012) estimated the CO₂ costs of a planting of 461 trees in Leipzig; the CO₂ emissions from the construction phase per planted tree were equal to approximately 6 kg C. Due to the large size of the planting and the relatively light infrastructure surrounding the studied trees, it is likely that the C cost of planting street trees in Scandinavia is higher rather than lower relative to the Leipzig case. For example, street trees commonly have a tree grate of cast iron, weighing up to 200 kg, installed around each tree; iron production causes emissions of 0.3-1.2 tonnes of C per tonne of iron (e.g. Kim and Worrel 2002).

Some establishment methods include additional concrete structures, such as planting boxes or curbstones, which further increase the C cost of tree planting due to the relatively high C footprint of cement manufacturing (Li et al. 2011). Thus, in addition to soil C loss, choices in the tree establishment set-up and equipment can easily offset the C sequestration benefits of short-lived trees entirely. Tree maintenance runs the risk of negating a considerable fraction of tree C sequestration benefits as well (McPherson et al. 2015). Thus, it is fortunate that in the current ecosystem service assessment schemes for urban trees the value of sequestered C tends to be relatively small compared with other benefits (e.g. McPherson and Simpson 2002, Roy et al. 2012).

5.2.3.2 The road forward

Compared with the predictions of the C sequestration compensation point, the literature on street tree life expectancy is discouraging. Early estimates of a life expectancy of 7-13 years (Moll 1989, Skiera and Moll 1992) were even more dire than the newer estimate of 19-28 years (Roman and Scatena 2011). The average life expectancy may not be the best estimate of the tree planting lifespan in relation to C sequestration, as young, small or newly planted trees may be more likely to die than older ones (Pauleit et al. 2002). Tree C sequestration efficiency is expected to improve with tree age in young and middle-aged forest trees (Pregitzer and Euskirchen 2004), and similar development may be present in street trees despite a different competition regime. It can also be expected that C sequestration improves after transplanting recovery (Study II). Thus, the loss of newly planted trees is less critical for C sequestration than the loss of trees already established, assuming initial C costs, such as those for tree grate and soil, are not repeated with each replanting. It seems obvious, however, that improvements in tree life expectancy and survival of older trees would aid

the urban tree C sequestration the most, as has been concluded also based on urban tree stock studies (e.g. Díaz-Porras et al. 2014).

5.3 TREE WATER USE, ITS DRIVERS AND POTENTIAL IN STORMWATER MANAGEMENT

5.3.1 TREE WATER USE

Measurement of tree water use is challenging; reasonably reliable methods for such measurements, such as ecosystem level evaporation estimation by eddy covariance measurements (Baldocchi et al. 1996, Scott 2010) or measuring single tree transpiration with a heat dissipation method (Granier 1985, 1987) or other thermal method (Tatarinov et al. 2005), have been widely available only in the last few decades. Early estimates, aiming for overall urban tree water use values to use in irrigation and container size selection have been predominantly based on relatively small, potted or lysimeter trees (Levitt et al. 1995, Beeson 2004, Montague and Kjelgren 2004), on porometer measurements (Kjelgren and Clark 1993, Kjelgren and Montague 1998, Kjelgren et al. 2001), which are difficult to scale over the entire tree crown and longer periods of time, or on measurements from other environments, such as forests or orchards (Levin and Assaf 1973, Miyamoto 1983). For example, aside from nursery tree irrigation (Beeson 2004, Beeson 2012, Tawegoum et al. 2015), energy balance-based models have been applied to urban tree water use in relatively few cases (Vrecenak 1988, Montague and Kjelgren 2004, Ngao et al. 2015). Overall, it can be stated that the tree water use values used currently in tree site planning are still rough estimates and do not consider e.g. the microclimate of the site, tree phenology or actual tree size.

The controls of tree transpiration are in general fairly well understood. Among tree-related factors, the effect of leaf area is perhaps the best understood in a forestry context. Stomatal regulation has been intensively studied, also in many deciduous trees (reviewed by e.g. Whitehead 1998), connecting the tree and environment-related factors that control tree transpiration; numerous stomatal conductance models exist as well (recent overview is available by e.g. Damour et al. 2010). To some extent, the known principles can be directly applied to urban environments, but problems may arise when the ranges of variation do not entirely overlap in urban and reference conditions. Especially the relatively sparse tree placement in urban environments and the dominance of non-evaporating surfaces are challenging when hoping to apply knowledge from e.g. forest tree transpiration to urban trees. Fortunately, some studies concerning the effect of urban surface materials on tree water use are available; the effect of asphalt relative to a vegetated surface can be in either direction – to increase water use through higher temperature leading to high D or to decrease it via

the high D triggering stomatal closure (Kjelgren and Montague 1998, Montague et al. 2000, Kjelgren et al. 2004). The effect of tree spacing on transpiration has also been investigated empirically, and the magnitude of the spacing effect within the range of variation seen in street tree spacing can be narrowed down somewhat (Beeson 2004, Hagishima et al. 2007, Ngao et al. 2015). The relative importance of D -related stomatal control versus soil water limitation in different tree species merits further study in urban trees. The application of continuous *in situ* tree transpiration measurements has made it much easier to relate tree transpiration response to environmental conditions (Bartens et al. 2009, McCarthy and Pataki 2010, Peters et al. 2010, Chen et al. 2011, Litvak et al. 2011, Wang et al. 2011): this approach has resulted in the greatest progress made recently in urban tree water relations research.

When the case study street tree sap flow (Study III), assumed to represent tree transpiration, is compared with forest tree transpiration in boreal environments, street tree water use appears to be similar or somewhat higher (Herbst et al. 1999, Arain et al. 2003, Amiro et al. 2006, Bernier et al. 2006, Daley et al. 2007, Oishi et al. 2008, Launiainen 2010). *Alnus* tends to have fairly high stomatal conductance (Eschenbach and Kappen 1999), which is probably the main reason for the *Alnus* transpiration being rather high relative to *Tilia* (Study III) and to previously cited literature on forest tree transpiration. However, no literature on measurements from urban environments was found for either species. In Minnesota, Peters et al. (2010) measured transpiration rates of *Tilia americana*, and these were somewhat lower than those for *Tilia × vulgaris* in Study III. *Tilia americana* is considered to be more shade-tolerant than *Tilia × vulgaris* (Burns and Honkala 1990), and the late successional species tend to have lower stomatal conductivity as well (Bazzas 1979). Ngao et al. (2015), on the other hand, measured higher transpiration for *Tilia tomentosa* in France than that for *Tilia × vulgaris* in Study III, but the tree transpiration ratio to potential evaporation was similar.

It appears that columnar crown form is effective in increasing tree transpiration on PCA basis (Study III), at least in the studied case, in an open urban environment with little to no competition for light between individual trees. The species differences in daily tree transpiration found in Study III were to a large extent due to the differences in leaf area, and together with varying crown surface area and stomatal conductance parameters, were shown to account for most of the species differences (discussed in the following section), but not all between-year differences could be explained satisfactorily. In comparing the annual water use values within species, the j_s (sap flow per trunk cross-sectional area, Table 5) is more valuable in the detection of potential errors and biases of the sap flow method than water use per PCA or LA, as measuring or estimating these contain additional sources of error. For LA these are discussed in Study I; with regard to PCA, especially in *Alnus* there seemed to be much variation in how much shoots

leaned out of the columnar crown shape, perhaps depending on the length of shoot extension growth, which affected the measured PCA. Even in j_s , however, large differences between years were seen; at both sites, 2008 and 2011 showed lower water use than the intervening years. This pattern in annual differences in j_s on both sites is unlikely to be due to systematic error in measurements; instrumentation was independently run and maintained on each site, albeit similarly constructed. Not much research has been done on the factors explaining interannual variation in tree water use or gas exchange, but there are some indications that while variation in LAI has a notable role, plant response to the same environmental factor levels may vary between years (Richardson et al. 2007).

The heat dissipation method used for measuring the water flow in tree trunks in this study has been much examined in the literature. Both its theoretical physical basis (Tatarinov et al. 2005, Wullschleger et al. 2011) and empirical comparisons with measurements conducted using other methods (e.g. Braun and Schmid 1999a, Wilson et al. 2001, Dragoni et al. 2005, Hölttä et al. 2015) show some problems. The measurement reading has typically a time lag in comparison with changes in sap flow (Granier 1985, Braun and Schmid 1999b), which can lead to attenuation of the sap flux signal, underestimating peak flows (Hölttä et al. 2015). This lag can be related to factors affecting thermal properties of the wood, which in turn are related to e.g. wood water content (Chen 2012, Vergeynst et al. 2014). It may be also a true observation of water flow in the trunk to some extent, as the plant biomass above the sensor has a considerable stock of water, and its transpiration does not necessarily cause instantaneous replacement water flow past the sensors (Meinzer et al. 2003, Burgess and Dawson 2008, Phillips et al. 2009). However, time lag related to tree capacitance can be expected to be rather small (Perämäki et al. 2001, 2005). This phenomenon may cause water flow to be measured at night, which then can be incorrectly identified as transpiration, although the water tension in the tree may be merely returning to balance with soil water tension while tree stomata are closed (Phillips et al. 2003, Wang et al. 2012). Actual transpiration at night is a fairly common phenomenon as well, as shown by e.g. Daley and Phillips (2006). Both time lag and possible non-zero flow at T_{max} would cause the tree water use to be underestimated, which is for the stormwater management-related aims of the study a less critical issue than the possibility of considerable error in the opposite direction.

5.3.2 MODELLING TREE WATER USE

While the PM M1 model, fitted to measured sap flux, gave an average R^2 of 0.70 on an annual basis, each addition to it improved the situation, with R^2 of 0.80 reached in M3 at the annual level (Study III). Adding a canopy conductance model in M2 to replace the standard g_c value in M1, as has commonly been done in orchard research when applying the PM model (Lu

et al. 2003, Pereira et al. 2006), improved the fit of the model with both general and annual parameterization, more at the *Alnus* site than at the *Tilia* site. The value of the scaling parameter c was similar for both M1 and M2. Considering canopy architecture of a single tree in M3 led to considerable improvement in the application of the PM model, and also the scaling parameter, defined as β for M3, responded to this change in model structure. The improvement in the model performance from M2 to M3 is understandable, as the PM model was originally developed for uniform crop canopies. In contrast to our results, however, in a study including several tree and vine species in various orchard settings, canopy-related variables apart from LAI had no effect on transpiration (Pereira et al. 2006). Different tree spacing (Beeson 2004, Hagishima et al. 2007) and absence or presence of shading and non-transpiring surfaces (Kjelgren and Montague 2008) may have some role in the disparity. Also, as was discussed earlier in relation to measured tree transpiration, for an isolated tree that is not a part of a uniform canopy layer, the inclusion of more explicit crown architecture than mere horizontal projection area may be justified.

The degree of variation that could not be explained by tree or environment-related variation remained in the parameter β of the M3 model. Examining the variation in annually estimated β (Study III, Table 3) reveals that interannual differences were still considerable and the pattern was similar as observed in j_s in both species – high level of transpiration in 2009 and 2010, and low level in 2008 and 2011. Despite the lower than average sap flow in 2008 and 2011, the three g_c function parameters (Eq. 3, Study III; g_{smax} , a and b) were very different for these two years. The parameter g_{smax} describes the maximal canopy conductance. The parameter a governs the I response and b the D response of transpiration. At both sites, the parameterization of the year 2008 gave usually either the highest or the lowest values for the parameters and the three other years were more similar to each other. Interannual differences in parameter values imply that tree response to environmental factors may vary between years, as has been found in some studies of e.g. ecosystem model parameterization based on eddy covariance data (Richardson et al. 2007, Groenendijk et al. 2011, Keenan et al. 2012), although the causes for the variation are not well known. Phenology, soil moisture-related issues, leaf maturity and senescence and carry-over effects of extreme climatic events are among the main suspects (Verma et al. 2005, Vermeulen et al. 2015).

Site and tree species effects could not be entirely separated in our research set-up, as each species was present only at the respective site, and shading, tree spacing and soil water status differed between sites (Section 4.1). The D response modifier was not significant at the *Tilia* site in all years, but the other parameters were significant for both sites and all years. The *Alnus* site was slightly drier overall (Figure 6), which may have contributed to the more pronounced sensitivity of *Alnus* to D in comparison with *Tilia*, in addition to species-related differences. Especially the values of a and g_{smax}

were different for the two sites; g_{Smax} was approximately three times as high for *Alnus*, and the value of parameter a indicated higher sensitivity to light, resulting in higher water use in lower light than was seen for *Tilia*. This may be at least partially attributed to species differences, as *Alnus* is known to use water less sparingly (Eschenbach and Kappen 1999). Together with c , which was slightly higher for the *Alnus* site than for the *Tilia* site, the parameters fitted annually were able to produce the difference in predicted transpiration rates on PCA basis seen in measured transpiration between the sites. Very little indication of any species or site difference remained in β in M3; in the all years models (general parameterization), the value of β was 0.11 for the *Tilia* site and 0.12 for the *Alnus* site.

In M1, the parameter c roughly equals the crop coefficient k_c (ratio of observed water use to reference evapotranspiration) used to assess irrigation needs of various crop plants (Doorenbos and Pruitt 1977). Crop coefficient assumes a large area of uniform canopy, which is inconvenient for many horticultural applications. In Study III, the calculation is done directly for individual tree canopies rather than for a larger area, and c parallels k_c in this respect. The definition of k_c also assumes optimum soil water conditions, which implies that c determined for *Alnus* is not valid k_c in this respect; for *Tilia*, this was rarely a problem. In the model version M2, the effects of g_c regulation were directly attributed to the g_c function, leaving the canopy structure and leaf area-related effects in the parameter c . The value of parameter c did not respond much to this change in the model structure, implying that crown structure, rather than stomatal function, was the main cause of the between-year differences in parameter values. In the next step, in model M3, the effects of leaf and crown surface area were further removed from parameter c to be considered explicitly; therefore, parameter c is redefined as β in model M3.

Crop coefficient – and the scaling between plant water use and reference evaporation in general – is of some interest in urban forestry. For most urban tree soil volume calculation methods, tree water use is estimated from ET_0 or pan evaporation with an “evaporation ratio” or a “pan factor”¹, by Lindsey and Bassuk (1991) and DeGaetano (2000), respectively. These scaling factors are related to k_c , but exact definitions are somewhat confused. Both methods estimate tree water use as a product of projected canopy area, leaf area index, pan evaporation and the aforementioned scaling factor “evaporation ratio” or “pan factor”. As both LAI and PCA are included in the calculation, the scaling factor should be given on a leaf area basis; if the scaling factor were given on a land area basis (as ET_0 and k_c), then LAI should not be included in the equation. The scaling factor in Lindsey and Bassuk (1991) is given on a leaf area basis. However, some of the literature

¹ The use of term “pan factor” is unusual in DeGaetano’s (2000) paper; it usually refers to the conversion factor used to calculate ET_0 from pan evaporation. When using the term as referred to by DeGaetano, it is given in double quotes.

value comparisons in Lindsey and Bassuk (1991) refer to k_c values, which at least in two cases are (correctly as per definition of k_c) given on a land area basis (Doorenbos and Pruitt 1977, Levin and Assaf 1973). The confusion is also repeated in DeGaetano's (2000) calculation method, which cites both the given value for "evaporation ratio" of Lindsey and Bassuk (1991) and k_c values derived from the literature as possible "pan factors" for the model. Thus, it can be concluded that the PM M1 parameter c (Study III, Table 2) must be transformed to a leaf area basis (c/LAI) to be comparable with the aforementioned "evaporation ratio" or "pan factor". Further assuming the conversion factor between ET_0 and pan evaporation is e.g. 0.7 (Doorenbos and Pruitt 1977), the scaling factor from our data is about 1/4 of the original model parameter given (0.20). Because the summertime climate in Helsinki is slightly cooler than in Ithaca (NY, USA), lower values may be expected, and limited soil water might have contributed at the Alnus site. Larger tree size and relatively high LAI in our data may explain much of the remaining difference, as the trees examined for estimating the "evaporation ratio" (Lindsey 1990, ref. Lindsey and Bassuk 1992) were only up to 1.4 m² in canopy area.

5.3.3 ENVIRONMENTAL CONTROL OF TREE WATER USE

The PM model residual correlations (Study III, Table 5) were analysed both against variables included directly or indirectly as drivers in the PM model and against one variable that was not in the model (SWC). Comparing the residuals against environmental variables included in the model can reveal issues with model structure and give insights into the microclimate experienced by street trees. Residual correlations were overall strongest on both sites in 2008, when the data covered only a period of two months in high summer and parameter values were notably different from the other three years (Study III, Table 3). In later years, at the Tilia site, I and D were rarely significantly correlated with the residuals. In 2009, the only variable significantly correlated with the residuals was soil water content, which had a fairly weak effect, and in 2010 and 2011, only very weak correlations were found. At the Alnus site, the correlation with SWC was strong in 2008 and detectable in all analysed years, but also T and/or D had significant residual correlations in most years. The environmental factors included in the PM base model and the g_c model – solar radiation and water vapour pressure deficit – are the directly driving forces of transpiration. They are also interrelated with each other and air temperature because radiation increases temperature, and temperature increases D , assuming no change in partial pressure of water vapour. Temperature, while not directly included in the PM model, is included in the model through several derived variables (Eq. 1 in Study III), and D .

In Study III, the residual correlations seen in 2008 were likely partially related to the short data period available for model fitting, although fitting

the model only for July-August for other years included in the dataset did not produce as strong residual correlations (data not shown). The repeated occurrence of D and T -related residuals at the *Alnus* site implies that the T and/or D response was not entirely realistically captured by the PM M3 model. The PM model includes vegetation structure and physiology-related resistance terms that are responsive to environmental conditions. The fact that neither T nor SWC were directly included in the model could have caused the problem with the T and D response, as high T and D tend to co-occur with low SWC. Stomatal reaction can be related to either high D or low SWC, and stomatal closure has been observed in both situations in urban settings (e.g. Kjelgren and Montague 1998, Chen et al. 2011). High temperatures may also directly reduce the rate of photosynthesis (Smith and Dukes 2013), which could lead directly to reduced transpiration (Nikinmaa et al. 2013) without a direct effect of D .

SWC was not included in the fitted PM model versions. Residual correlations thus imply that it would improve the model to include the SWC effects. Correlations were quite weak, except for the *Alnus* site in 2008, when SWC alone could explain most of the remaining variation. While the period available for model fitting was short in 2008, the correlation was clear. A weaker but notable residual correlation was present also in 2011. Somewhat surprisingly in 2010, when SWC also was quite low at times, the correlation was much weaker. Thus, it seems that in the examined period, water availability had a strong impact on tree water use in two cases of the eight examined; however, the effect was detectable in an additional three cases. The two strongest cases were detectable by the simple regression analysis as well, but regression analysis was unable to identify all of the cases detected by PM residual analysis. This shows that the PM model residual analysis, while more complicated, is also a more powerful tool in examining the relationship between urban trees and their environment.

5.3.4 TREE WATER USE AND STORMWATER

Tree water removal (transpiration or transpiration + canopy interception) was compared with annual rainfall assuming different percentages of canopy cover (Study III). The potential effect of tree transpiration in water removal is clearly considerable, and very high water removal relative to annual rainfall appeared attainable with reasonable canopy cover, especially with *Alnus* trees. In urban areas, stormwater-related problems occur where land use intensity is high, and open or vegetation-covered ground is not feasible due to heavy trampling and high human use intensity. In a study reporting stormwater runoff from three urban catchments in Helsinki (Valtanen et al. 2014), runoff/precipitation ratios indicated that more than half of the warm season rainfall was lost from the catchment with highest land use intensity (total impervious area 89%) as runoff. It can be assumed that the remaining proportion was mainly lost as evapotranspiration. This would indicate that

the actual summertime stormwater loading of intensive land use areas could be of a magnitude of half of the rainfall, which brings the estimate of *Alnus* canopy cover needed to transpire the stormwater to roughly 20% in this case. Unfortunately, this estimate leaves much room for error, as non-vegetation land cover transpiration in urban areas is difficult to describe accurately on a small scale. Studies on transpiration from permeable pavements (Starke et al. 2011) emphasize, in addition to the role of surface albedo, also such difficult-to-measure variables as soil water storage and capillary rise in layers beneath the surface. Considering other possible land cover types, such as lawn, tree transpiration in relation to ET_0 can be seen directly from the value of parameter c in M1 (Study III, Table 2). It indicates that for *Alnus* transpiration is similar or slightly higher than that of the well-watered grass reference.

Scaling up the measured tree transpiration to different canopy cover is problematic, however, because transpiration per tree increases as canopy cover decreases (Oke 1987). It was found in a plant spacing experiment (Hagishima et al. 2007) that transpiration from scattered vegetation (per canopy area) could be several times higher than from plants placed close together. In Study III, it was not possible to take tree spacing into account because there was next to no variation in tree spacing; street trees tend to be planted in even rows. In the modelling exercise, taking crown surface area into account decreased the residual variation on both annual and interannual levels, which implies that the vertical dimension contributes much to total transpiration, and thus, tighter spacing could notably diminish the tree transpiration. Also the possible effects of shading buildings surrounding the street canyon on *Alnus* transpiration remain unquantified, but can be expected to lower the tree transpiration to some degree in a similar manner as closer tree spacing would, thus causing no risk of overestimating the potential tree transpiration in Study III.

While the rainfall interception by tree crowns is usually included in tree ecosystem service estimations (Bolun and Hunhammar 1999, McPherson and Simpson 2002, McPherson et al. 2005, McPherson 2007), the water use of urban trees is rarely considered in the context of stormwater management (Scharenbroch et al. 2016). This is counterintuitive as urban trees are commonly believed to suffer from lack of water; yet leading stormwater out of urban areas is both expensive and detrimental to receiving water bodies (Valtanen et al. 2014). To some extent, the problem is related to differing time scales; in Scandinavia, stormwater management issues tend to relate to rain events of high intensity and not necessarily long duration (e.g. Semadeni-Davies and Bengtsson 1999), while tree transpiration tends to become important on a longer time scale (e.g. Scharenbroch et al. 2016). Comparing measured or modelled tree water removal in our study against annual precipitation with and without taking canopy interception into account (Figure 15) shows that, especially for the columnar *Alnus* with its narrow crown, canopy interception adds relatively little to tree-related

rainfall removal. Canopy interception effect on stormwater generation benefits only the area covered by the canopy, while transpiration water can be – in theory – collected from a larger area. Whether this potential is realized in urban environment is doubtful, as both unsealed soil area and tree root extension tend to be limited.

The degree of soil sealing is a dominant determinant of tree environment in urban areas. This can be easily seen with the water cycle, but also with all resource inputs and outputs related to urban trees; carbon and nutrients are cycled between aboveground and belowground parts of the ecosystem as well. The potential that tree transpiration clearly has in management of the urban water balance can be realized by various establishment methods that allow for the use of rainwater as a water source for trees. Permeable surfacing is one such method (Morgenroth and Visser 2011, Viswanathan et al. 2011), bioswales are another example (Xiao and McPherson 2011, McLaughlin 2012). More solutions are being applied in pilot projects and also are in wider use in the field (Grabosky 2015), although research publications tend to lag behind in this practically oriented field.

5.4 REMARKS ON THE POSSIBILITIES AND LIMITATIONS OF THE CASE STUDY APPROACH

Studies I, II and III were performed at the same research sites, two real-life streets built following the usual local street and tree site planning and construction practices. The close connection between research and practice, typical for the field of urban tree research, can be both a blessing and a curse; on one hand, research can produce knowledge that is easily applicable in the field, on the other hand, it may lead to overly practical research set-ups from which results are difficult to compare (Grabosky 2015), not to mention generalize. Studies I, II and III are case studies, where a limited number of trees, representing only two species, and tree soils at the same sites were measured repeatedly. Focusing data collection on the case study sites allowed for efficient use of resources and automation of some measurements. Another clear benefit was the opportunity to see long-term effects of the environment on tree development and to shed light on long-term, ecosystem-level phenomena such as tree carbon sequestration and site water balance. The connection and feedbacks between the tree environment and the ecosystem services it produces benefit from a long-term integrated view, which is much more easily achieved with few well-documented sites than with e.g. sampling of large street tree populations.

Problems of the case-study set-up are largely related to analysing the data and generalizing the results; because the range of tree species and site conditions is limited, generalizing must be based more on analysis and understanding of phenomena, interactions and mechanisms than on statistical evidence. Thus, the results as such can only be applied in Finland,

where tree establishment practices and conditions are reasonably similar, but the understanding gained on how street trees interact with their environment/ecosystem and produce services can be applied more widely. For example, while the conclusion that the case study trees and soils were a carbon source rather than a carbon sink ten years after planting (Study II) does not give an answer as to whether this is true for the average street tree and soil in Finland, in boreal regions or globally, it brings forth the possibility and also provides tools to avoid this situation as well as tools to assess any given tree and soil case. Similarly, while observations of biomass distribution of only two tree species on one site reveal little of the applicability of forestry-derived BEs to urban trees in general, a possible deviation of the urban tree biomass distribution from that of the forest trees, and its likely causes were identified and can be challenged in future studies. Another problem of a long-term case study is the repeated measurements of the same few individuals, which fortunately can be solved with statistical methods to some extent. Some risk remains, however, that a given phenomenon seen in the data is caused by inherent variation in the studied trees rather than the factors and effectors analysed. On the other hand, repeated measurements can bring about high resource use efficiency, as taking the subject effects into account in analysis tends to reduce random variation, thus allowing for significant effects with smaller sample sizes.

The sites were constructed to serve as normal streets with street tree plantings, subject to typical street and tree maintenance procedures. This allowed for reasonable set-up and maintenance costs. Building such streets and maintaining them in e.g. an experimental field for a decade would have been prohibitively expensive, while in the realized research set-up, the costs of the street and site construction and maintenance were part of construction and maintenance of regular city infrastructure. When hoping to bring forth useful solutions in tree establishment, the set-up built on a real-life street has the additional benefit of serving as a practical example, a pilot case, which can be used directly as a source of information by the actors in the practical field of urban greening.

In conclusion, long-term case studies have a place and distinct value in urban tree research. For example, in Study III, the applicability of PM models in predicting street tree transpiration would have been perceived quite differently if only one year of data had been available. However, also larger population studies and more representative sampling are needed to gain a firm understanding of the relative importance of e.g. tree biomass distribution differences between non-urban forest stands and street trees. Long-term case studies can help in the formation of new hypotheses, which then can be validated with more representative data collected from larger populations and analysed with more robust statistical methods.

Overall, urban tree research is related to slow phenomena such as tree growth, and long-term studies are difficult to achieve, especially at the pace necessitated by changing demands for the urban tree environment and

services. Grabosky (2015) suggests that a common testing and reporting scheme for structural soil studies would help to gain a general picture and comparable results. To take the matter further, related fields, especially forest ecology, rely greatly on process-based modelling of tree function and development. Forestry and forest ecology research have produced detailed understanding of how trees react to their environment, and a fair degree of convergence in environmental responses can be expected between tree taxa (Meinzer 2003). Process-based models predict tree growth and function from environmental responses and can predict e.g. changes in forest growth as a result of various climate change scenarios, albeit with many uncertainties (see recent review by Lindner et al. 2014). In the field of urban tree research, the difficulty in attaining detailed information about the heterogeneous urban environment and the variety of tree species present has no doubt hindered the application of modelling approaches; however, as pointed out already in the 1980s, variation in the urban environment does not fall entirely outside the natural range (Whitlow and Bassuk 1987). This implies that process-based models developed in a forestry context may serve in elucidating urban tree responses to their environment. With the growing interest in urban climate models and increasingly high-resolution spatial and land-cover data available (Karsisto et al. 2015, Nordbo et al. 2015), developing models to describe tree function and growth in urban environments may be feasible in the future.

5.5 ECOSYSTEM SERVICE PROVISION BY YOUNG STREET TREES – PRACTICAL IMPLICATIONS

In the Million Tree Initiative and other tree planting campaigns promoted in our cities, many hopes and expectations of ecosystem service provision are pinned on the newly planted urban trees. To begin with, these trees must successfully survive transplanting and conquer transplanting shock, often a considerable challenge in itself, especially if initial maintenance level of the trees is low. Thereafter, how these expectations are realized is largely determined by tree growth and resulting tree size. These, in turn, depend on the growing environment that we create and its provision of growth resources for the trees. The majority of ecosystem services accounted for in evaluation protocols depend on tree size; the bigger the tree, the more and better ecosystem services it can provide. Tree growth resources can be supplied by maintenance, such as irrigation and fertilization, or by urban biogeochemical cycles running parallel to water and nutrient cycles of natural ecosystems. When aiming for low operational cost in ecosystem service provision, the latter is clearly the better option, but presents a multitude of challenges. The majority of these challenges are most easily tackled at the planning and establishment stages, avoiding subsequent expensive site renovations and retrofitting.

The interrelations of tree growth, water availability and stormwater management potential of trees form an interesting network, and the challenge lies in finding the optimal place and time for intervention in this network to attain the best overall results. Tree water use increases at the same pace as tree size, but when soil volume limits root exploration, increased resource acquisition from the soil is also limited. It would be beneficial if the amount of water available for trees could be scaled according to tree size in such a way that both flooding and drought could be prevented. This can be realized to some extent by arranging a frequent and large water input in tree soil combined with good drainage. In addition, if rainfall intervals tend to be longish, at least a moderate soil water retention capacity is needed. Leading stormwater to trees can fulfil the first condition, and the idea is rather well known (e.g. Bartens et al. 2009, Embrén et al. 2009), albeit its adoption is somewhat hampered by concerns over infrastructure damage and soil and groundwater pollution (to some extent, these concerns are common for all stormwater infiltration solutions). Stormwater is either collected from roofs or footpaths, or infiltrated through permeable asphalt or pavement, and led to well-draining structural soil with a subsurface drain at the bottom. The water retention capacity of structural soils tends to be low due to the large volume fraction of stones, although these can contribute to soil water retention as well (Grabosky et al. 2009).

Attaining adequate water retention and storage in structural tree soils without compromising oxygen availability and the high infiltration rate remains challenging, despite fairly extensive experience with the use of water retention agents in the fine soil fraction (e.g. hydrogel, Grabosky and Bassuk 1995; lava stone, Bartens et al. 2010). The testing of two structural soils used in the US, both designed for good water retention, gave plant available water storage of 7-11% of volume (Grabosky et al. 2009), which is likely in the upper range of what can be attained in structural soil by optimizing the soil mix; these numbers indicate that plant-available water fills at least 30% of the voids between matrix stones. Considering both water and carbon-related ecosystem services, perhaps the most attractive additive is biochar, already used in tree soils (Scharenbroch et al. 2013, Andreasson et al. 2014). Its renewable raw material and longevity in soil could help attain C sequestration in urban plantings. It may have additional potential benefits in cleaning stormwater passing through tree soils (Ahmad et al. 2014). It improves soil water retention and at least in certain situations retains plant-available nutrients; the amount of research available on other applications gives confidence in its positive effects (e.g. Jeffery et al. 2011). However, the effects and longevity of biochar depend greatly on its ingredients and pyrolysis process (e.g. Zimmerman 2010), and more research is certainly needed. Unlike with stormwater infiltration through tree soils, there are few concerns about potential harmful effects of biochar, and consequently, its use may be adopted more easily.

There appear to be few reasons to continue using peat in tree soils, as it can be replaced with other materials to provide the benefits of soil organic matter. Replacing peat as a compost bulking agent is being actively investigated (e.g. Barrington et al. 2002, Manios 2004, Dias et al. 2010). Overall, SOM additives in tree soils will likely continue to be valuable for soil properties, as it is difficult to envision any way to retain or improve local aboveground litter cycling and the related nutrient and carbon inputs to soil on sealed soils to any significant degree. Unnecessary soil sealing should obviously be avoided, and open soil can be retained under trees with e.g. ground cover plantings when permitted by site use. There are some other possible avenues to reduce tree planting-related C costs, the effectiveness of which could be assessed. Planting smaller nursery trees might cause lower related C expenses, as they are easier to transport and can be planted without heavy equipment. Whether the smaller tree size would offset this benefit by lower lifetime C uptake should be studied in more species and with different maintenance levels. The current prevailing view that smaller nursery trees attain the same or greater size than larger transplants within few years (Struve et al. 2000, Struve 2009, Gilman et al. 2010) is based on a limited number of studies with only a few tree species. Less C intensive tree-related infrastructure and materials could be chosen, recycling and re-using when possible. Trees planted in the most intensively built areas are obviously at disadvantage compared with more lightly built-up areas, but the trees in tightly constructed neighbourhoods can be very important providers of ecosystem services other than C sequestration.

Current tree canopy cover in cities can give some indication on how far canopy cover can realistically be increased in the hopes of improved stormwater management and other ecosystem services. Pincetl (2010) and Nowak et al. (2008) have provided average canopy cover values around 22% for US cities, but both between and within-city variation can be considerable. McPherson et al. (2008) estimated that in Los Angeles, a city where urban trees are largely dependent on irrigation, 27-33% canopy cover could be attained. Nordbo et al. (2015) analysed fine resolution lidar data to determine ground cover types for central Helsinki and for a smaller area in downtown Helsinki. These were found to have 16% and 12% of high vegetation cover and overall vegetation cover of 27 and 22%, respectively, much of the other vegetation being grass. Overall, the rather large scale of the examined areas in these assessments does not reveal how much canopy cover could realistically be attained in e.g. a residential downtown block, but values exceeding 20% seem possible. If trees with columnar crown form are used, attaining a high canopy cover is more challenging, as more individual trees, and corresponding tree sites, are needed.

While the columnar *Alnus* appears efficient in transpiring water based on this study, it is essential not to over-use any single taxa in urban environment to keep the tree diversity high (e.g. Raupp et al. 2006, Greene and Millward 2016). In most cities, however, *Alnus* is currently a relatively rare taxon (e.g.

Sjöman et al. 2012), and its use could be increased as long as diversity on a local scale is kept in mind. Many other tree species have columnar forms and varieties available, and it would be interesting to investigate the transpiration of these forms as well.

Among the ecosystem services provided by street trees, assessments of monetary value tend to show carbon sequestration as a minor benefit among the already modest proportion of non-aesthetic benefits (e.g. McPherson and Simpson 2002, McPherson et al. 2005, McPherson 2007, Peng et al. 2008, Soares et al. 2011). The aesthetic benefit, which can be assessed based on the increase of property value with the addition of adjacent trees, often produces the bulk of the monetary value of urban tree ecosystem services. Runoff prevention, based on crown interception only, is usually considered more valuable than CO₂ sequestration, typically comprising from a few percent up to 90% of the non-aesthetic benefits, depending on the assessed area and the local expenses in stormwater management. A comparison of study methods used shows that how monetary value is calculated for e.g. a tonne of sequestered carbon or a cubic metre of prevented stormwater affects the results strongly. Local data and valuation methods for services are thus needed as the basis for monetary valuation; in Finland, this work is still in its early stages (e.g. Punttila 2014). Based on the work presented here, it is not possible to assess the absolute or even relative value of examined benefits.

Overall, improving tree size and longevity appear the most important goals in increasing the ecosystem service benefits of urban trees. This may require more inputs, e.g. more tree maintenance, than is currently the norm, but these additional economic and environmental costs (McPherson et al. 2015) need to be re-assessed with the ecosystem services in mind; they may be justified if they can help old, large trees to remain and provide benefits in our cities. The value of large trees should be emphasized so that an effort is made to retain existing trees instead of cutting down and replanting; a tree economic valuation model based on trunk cross-sectional area seems a good start (Östberg and Sjögren 2016).

6 CONCLUSIONS

Street tree crown allometry was found to respond to disturbances, such as transplanting, and to at least one environmental condition, namely soil water content (Study I). Shoot growth was dependent on soil moisture, showing a decrease in both dry and overly wet conditions. This shows that, in addition to adequate soil volume provision, risk of drought and also flooding must be avoided in stormwater management solutions. The three types of structural soils tested in Study I were all reasonably well-suited for street tree growing media, assessed based on attained tree growth.

Within the first decade of life after planting according to local (Finnish) current standard practices, a street tree is much more likely to be a C source rather than a C sink due to high C loss from tree soils (Study II). While it is possible that the balance is inverted later in the tree's lifespan, this requires improvement in tree life expectancy. Also the source of organic matter in tree soils should be chosen with care. The *Tilia* trees studied were expected to take at least 30 years of growth at the street site to attain C neutrality with regard to tree sequestration versus soil C loss. While trees grew reasonably well on all three tested growing media, from the point of view of improving tree ecosystem services, relatively low and stable SOM in growing media is recommended, and the use of peat is discouraged.

Existing biomass equations are more suitable for predicting the total aboveground biomass of street trees than the biomass distribution between the trunk, branches and leaves. Further work is needed on biomass distribution within street trees and on whether the distribution is merely caused by the open growing site and pruning or whether some other features of urban environment significantly shape the biomass relations. The crown allometry measurements in Study I indicate that soil water content and the disturbance regime are likely to play some role. Also the effects of urban tree biomass distribution on above- and underground litter abundance and dynamics warrant further study in the context of urban carbon and nutrient cycles.

Annual street tree transpiration could be predicted with a reasonable accuracy from microclimatic factors and leaf area, but the addition of a more detailed description of crown dimensions improved the model notably. Accuracy could be further improved by including soil water availability in the model. Attempts at intra-annual parameterization of the tree transpiration model were less satisfactory. On an annual time scale, the case study trees contributed notably to local water balance with their transpiration. Calculations (Study III) indicated that *Alnus* trees could transpire the annual average rainfall with less than 50% canopy cover. While such canopy cover is often not realistic, the results show that leading e.g. roof water to street trees could markedly lessen the stormwater loading in drainage systems.

Additionally, while the canopy cover increases, it is expected that transpiration per PCA decreases as the sides of the narrow crown, contributing much to canopy surface area, become less exposed to atmosphere. Nevertheless, street trees could benefit from additional water resources and simultaneously lessen the stormwater loading if e.g. roof water could be led to tree soils.

Street tree transpiration, growth and litter production could be approached (in single tree to city block scale) with integrated models, taking advantage of the increasing availability and improved resolution of urban meteorological data and remote sensing methods to measure changes in tree dimensions. Even an unsuccessful attempt at developing and applying such a model will shed light on future paths. We need to identify the tree responses that are the least successfully transferred from other ecosystems to urban environment and focus our research on these. This type of modelling exercise would take what is currently known of tree environmental responses and use this information in a range of conditions increasingly likely to be met also outside cities as climate change proceeds, thus providing valuable model testing and validation.

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