https://doi.org/10.35662/unine-thesis-2781

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Impacts of urban gardening on soil quality, soil fauna and soil multifunctionality

PhD thesis

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22/05/2019

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Titre:

"Impacts of urban gardening on soil quality, soil fauna and soil multifunctionality"

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Neuchâtel, le 15 juillet 2019

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Preface

This dissertation is part of the interdisciplinary project **BetterGardens** conducted at the Research Institute of Organic Agriculture FiBL and at the Swiss Federal Institute for Forest, Snow and Landscape Research WSL financed by the Swiss National Science Foundation in the frame of the Sinergia programm (CRSII1_154416). The overall project aim was an integrated analysis of soil quality, biodiversity and social value of urban gardens in Switzerland and took place from January 2015 to December 2018. This thesis is part of the subproject about soil quality, belowground diversity, and soil functions. It is conceptualised as a paper dissertation, meaning that all main chapters are either published or submitted to scientific journals. Therefore, all rights of the main chapters belong to the respective publishers.

Abstract

Urban gardens are important and at the same time popular greenspaces. As urban areas are expanding globally, urban gardens play an increasingly important role in contributing to essential ecosystem services, well-being of citizens, and biodiversity in a city. Gardening activities determine above- and belowground diversity of plants and insects. However, the effect of gardening activities on soil quality and soil functions have rarely been studied. For this purpose, a city-wide assessment of soil quality including key soil fauna species and soil functions was established on 170 plots in 85 urban gardens of the city of Zurich. They have been selected in accordance to an urbanisation gradient and to cover the spectrum of existing garden management practices, from intensively managed annual vegetable beds to extensively managed gardens dominated by perennial flowers and grass species. A management survey for the 85 participating gardeners was developed in collaboration with the subproject on aboveground ecosystem services, such as pollination and pest control, and the subproject studying the social value of urban gardens. This management survey was then used to assess drivers in soil quality, measured by a multitude of physical, chemical and biological soil quality indicators.

Taken together, soil quality was shaped by garden management activities and was mainly determined by three urban garden land-use types: vegetables (annual vegetation), flowers and berries (perennial vegetation dominated by forbs), and lawn (perennial vegetation dominated by grasses). In addition, heavy metal concentrations were linked to the proximity to traffic and industry, but not to other factors such as garden management or garden age.

Next, the soil function decomposition of organic material was investigated with litter bags of two different mesh sizes (1 and 4 mm) and litter types (*Zea mays* L. leaves and stems). In both litter types, we found the highest decomposition rates including both macro- and mesofauna species, but decomposition rates were higher in leaves, which are the better decomposable litter type. Garden management was again a main influencing factor affecting soil function decomposition and diversity of soil fauna species (earthworms, collembola, isopods and gastropods). We found a positive relationship of soil fauna species richness (4 taxonomic groups, 120 species and 81'007 individuals) as well as plant species richness with decomposition. This indicated that also in intensively managed urban greenspaces, such as urban gardens, biodiversity drives ecosystem services.

Furthermore, we assessed the impact of the three garden land-use types on soil fauna and multiple soil functions, related to food production and soil quality. The management of specific garden land-use types not only determined the diversity of plants aboveground, it had also implications on soil fauna and soil functions belowground. The strongest effects influencing soil multifunctionality were caused by the differences in soil characteristics. We found that across all urban garden soils, high soil biological quality had a positive effect on soil multifunctionality, whereas management intensity decreased plant and soil fauna diversity, which had a positive effect on soil multifunctionality. Moreover, we found that soil fauna species richness most often decreased with urbanisation, but soil fauna abundance increased. Decomposition rates were also found to be higher in more urbanised areas, while no significant effect was found with soil multifunctionality.

However, very little studies investigate current gardening practices. Therefore, we see a great potential of future investigations in urban garden ecosystems, for instance about other soil functions such as carbon or water storage potential, but also about effects of organic gardening practices on urban biodiversity. As an example, we have found that many gardeners still use pesticides in their gardens without knowing its detrimental effects on biodiversity and soil quality. Since the vast majority of urban gardeners are supportive of biodiversity, futures studies could develop and analyse ecological gardening practices on above- and belowground diversity and their effects on long-term soil quality, for example with a citizen science approach.

Overall, we highlighted that soil protective management practices, such as applying compost or mulch, and lower management intensity need to be integrated in urban planning strategies on a citywide scale but also at the local garden association level, in order to maintain important ecosystem services. We conclude that urban gardens have the potential to increase urban biodiversity and important ecosystem services, while at the same time being meeting points for people with different social backgrounds and increasing human well-being.

Keywords: urban gardening, soil quality indicators, urban ecosystem services, soil function decomposition, litter bags, tea bag index, soil mesofauna, soil macrofauna, earthworms, collembola, isopods, gastropods, soil biodiversity ecosystem functioning, soil multifunctionality

Résumé

Les jardins urbains sont des espaces verts populaires et importants. L'espace urbain est en pleine expansion à l'échelle mondiale, et les jardins urbains contribuent de plus en plus aux services écosystémiques, notamment au bien-être des citoyens et à la biodiversité en ville. Les pratiques horticoles déterminent la diversité des plantes et des insectes en surface et dans le sol. Toutefois, les effets de ces pratiques sur la qualité et les fonctions du sol ont été rarement étudiés. À cet effet, une évaluation de la diversité et de l'abondance des espèces faunistiques du sol a été réalisée sur 170 parcelles au sein de 85 jardins urbains à Zurich. Ces derniers ont été sélectionnés en fonction d'un gradient d'urbanisation et pour couvrir un spectre large des différentes pratiques horticoles, depuis les intensives cultures végétales annuelles intensives aux cultures pérennes extensives dominées par des fleurs et autres espèces herbacées. Une enquête a été élaborée concernant les pratiques des 85 jardiniers en collaboration avec les différents sous-projets : i) celui sur les services écosystémiques de support et de régulation, par exemple la pollinisation et le contrôle des parasites, et ii) celui sur la valeur sociale des jardins urbains. Cette enquête a ensuite servi à évaluer les déterminants de la qualité du sol, mesurés au travers d'une multitude d'indicateurs physiques, chimiques et biologiques. Au final, la qualité du sol est liée aux activités de jardinage et est principalement influencée par trois catégories d'utilisation : les légumes (plantation annuelle), les fleurs et les baies (végétation pérenne arbustive) et les pelouses (végétation pérenne herbacée). Les concentrations de métaux lourds sont associées à la proximité du trafic routier et des industries, mais pas à d'autres facteurs tels que les pratiques horticoles ou l'âge du jardin.

Ensuite, la décomposition des matières organiques a été étudiée en utilisant des sachets de litière de deux maillages différents (1 et 4 mm) et deux sortes de litière (Zea mays L., feuilles et tiges). S'agissant de la litière, les taux de décomposition les plus élevés concernent les organismes de la macro- et de la mésofaune; ils sont plus élevés pour les feuilles, plus faciles à décomposer. Ici encore, les pratiques horticoles représentent un des principaux facteurs influant sur la décomposition de la matière organique ainsi que sur la diversité de la faune du sol (vers de terre, collemboles, isopodes et gastéropodes).

Nous avons également trouvé une relation positive entre la richesse spécifique de la pédofaune (4 groupes taxonomiques, 120 espèces et 81'007 individus) et la richesse spécifique des espèces végétales en décomposition. Cela indique que, même dans les espaces verts urbains, la biodiversité stimule les services écosystémiques. En outre, nous avons évalué l'impact des trois types d'utilisation du jardin sur la faune et la multifonctionnalité des sols, en relation avec la production d'aliments et la fertilité du sol. Les pratiques horticoles n'ont pas seulement déterminé la diversité des plantes en surface, ils ont également eu des implications sur la pédofaune et les fonctions du sol. Les effets les plus importants sur la multifonctionnalité des sols sont liés à la variabilité des caractéristiques des sols. Nous avons constaté que, dans tous les sols de jardin, une qualité biologique élevée du sol a un effet positif sur sa multifonctionnalité, alors que l'intensité des pratiques diminue la diversité des plantes et celle la faune du sol.

De plus, la richesse spécifique de la pédofaune diminue avec le degré d'urbanisation, mais l'abondance augmente. Les taux de décomposition sont également plus élevés dans les zones les plus urbanisées, mais aucun effet significatif n'a été constaté en lien avec la multifonctionnalité des sols.

Cependant, peu d'études documentent l'effet des pratiques horticoles. Par conséquent, nous estimons qu'il existe un grand potentiel pour les futures recherches sur les écosystèmes de type jardins urbains, par exemple en ce qui concerne d'autres fonctions du sol comme stockage du carbone ou de l'eau, mais également les effets des pratiques culturales organiques sur la biodiversité en ville. Ainsi, de nombreux jardiniers utilisent encore des pesticides sans connaître les effets dommageables sur la biodiversité et la qualité du sol. À l'avenir, les études pourraient développer et analyser les pratiques de jardinage éco-

logique sur la diversité en surface et souterraine et leurs effets sur la qualité des sols à long terme, par exemple par le biais d'une approche scientifique citoyenne.

En général, nous avons souligné que pour maintenir des services écosystèmes importantes, les pratiques de protectrices du sol, comme l'application de compost ou de paillis, doivent être intégrées dans les stratégies des planifications d'espaces verts urbains, mais aussi au niveau des associations locales de jardins. Nous concluons que les jardins urbains ont le potentiel pour augmenter la biodiversité et des services écosystémiques urbaines, tout aussi bien qu'ils sont des points de rencontre de personnes et améliorant le bien-être humain.

De manière générale, pour favoriser et maintenir des services écosystémiques performants, les pratiques de protection du sol, comme l'application de compost ou de paillis, doivent être intégrées dans les stratégies de planification d'espaces verts urbains, mais aussi au niveau des associations locales de jardins. Nous concluons que les jardins urbains ont le potentiel d'augmenter la biodiversité et les services écosystémiques urbains, tout autant qu'ils constituent des points de rencontre de personnes, améliorant ainsi les contacts et le bien-être humain.

Mots-clés: pratiques horticoles, qualité du sols, services écosystémiques urbains, décomposition des matières organiques, litter bags, tea bag index, mesofauna, macrofauna, vers de terre, collembola, isopodes, gastéropodes, biodiversité du sols, multifonctionnalité des sols

1

General Introduction

1.1 Human impact on ecosystems

Since the middle of the 20th century, humans have changed ecosystems more rapidly than in any other period of time, largely to meet growing demands of food, water, fuel or fibre (MEA 2005). Land cover changes, modification of water and nutrient cycles or the massive inputs of fertilisers and plant protection agents have led to a massive and ongoing global biodiversity loss (Cardinale et al. 2012), including even common species (Ceballos et al. 2015, Hallmann et al. 2017). Currently it has been estimated that 40 % of the world's insect species will be extinct over the next few decades (Sánchez-Bayo and Wyckhuys 2019). This dramatic rates of decline will be accompanied by species replacement of habitat and dietary generalist species (Sánchez-Bayo and Wyckhuys 2019). Nevertheless, there is an increasing awareness of the detrimental anthropogenic impacts on ecosystems (Kremen and Merenlender 2018) and that human well-being depends on the functioning of ecosystems and the biodiversity within them (IPBES 2019). These functions can be described as services by ecosystems and are defined as "the benefits humans derive from ecosystems" (MEA 2005). Besides investigating the drivers of biodiversity, researchers have also started to investigate the consequences of biodiversity loss for ecosystem services (ES) (Cardinale et al. 2012). There is mounting evidence that biodiversity is important in maintaining and influencing ES (Hector and Bagchi 2007), such as food security and nutrition (Bommarco et al. 2013, Tscharntke et al. 2012, Rockström et al. 2017, IPBES 2019). Overall, there is a positive influence of biodiversity on ecosystem functioning (Duffy 2009, Cardinale et al. 2011) and its resilience (Hooper et al. 2005, Isbell et al. 2015). However, real world observations are still scarce (Gossner et al. 2016), especially in human dominated ecosystems (Isbell et al. 2017b, Schwarz et al. 2017) and are often based on single ES such as food production (Vogel et al. 2013, Caruso et al. 2018). However, the loss of biodiversity may drastically influence the rate of change in ecosystem processes (Cardinale et al. 2011), altering the regeneration and productivity of ecosystems (Wardle et al. 2011).

1.2 Urbanisation

Nowadays, more than half of the world's population lives in cities (cf. Figure 1.1), a proportion that is expected to rise to 66 % by 2050, with the highest annual rates of change for less developed regions (United Nations 2015a). The ongoing urbanisation process increases the level of anthropogenic pressure on planetary boundaries (Steffen et al. 2015). This also causes rising challenges for urban dwellers, such as environmental health risks or scarcity of resources and space (Moglia et al. 2018). Urbanisation is a major driver of environmental change from local to global scales (Figure 1.3, Grimm et al. (2008), Wigginton et al. (2016)). These include pollution of water, soil, air, change of land cover, and loss of biodiversity (Seto et al. 2012, Nagendra et al. 2018, Sánchez-Bayo and Wyckhuys 2019). The latter is affected both directly through the loss of habitats, fragmentation of natural patches and decreased connectivity (Alberti 2005), and indirectly through anthropogenic activities within urban areas (Güneralp et al. 2013). Therefore, urban green spaces are becoming increasingly important for the promotion of biodiversity in cities (Miller and Hobbs 2002, Goddard et al. 2010).

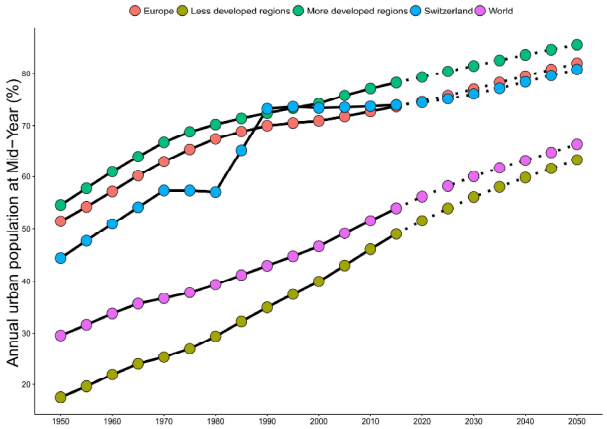


Figure 1.1 – Percentage of world population living in cities with data from the United Nations (2015a). More developed regions comprise Europe, Northern America, Australia/New Zealand and Japan. Less developed regions comprise all regions of Africa, Asia (excluding Japan), Latin America and the Caribbean plus Melanesia, Micronesia and Polynesia.

Worldwide, urban areas are growing faster than any other land-use type (Hansen et al. 2005) and cities grow even faster than the population increases in urban areas (Seto et al. 2011). For instance in Europe, urban area has expanded by 78 % since the 1950s, while the urban population has grown by 33 % (European Environment Agency 2006). Particularly, peri-urban areas are projected to grow up to 3.7 times compared to urban areas (Piorr et al. 2011). Furthermore, urban expansion often threatens biodiversity hotspot areas, (Figure 1.2, Seto et al. (2013)), regions with many endemic species, facing habitat loss and degradation (Myers et al. 2000).

1.3 Urban ecosystems as novel ecosystems

Urban areas consist of a wide range of ecosystems, including regions with high native biodiversity and rare as well as endangered species (Ives et al. 2016). Urban ecosystems are largely dominated by anthropogenic activities with cascading effects on urban biodiversity, such as changes in biogeochemistry (Grimm et al. 2008), local temperature (Voogt and Oke 2003) and hydrology (Walsh 2000). The consequences for biodiversity in urban areas depend on the taxonomic groups, the spatial scale, and also the intensity of urbanisation (McKinney 2008, Dale and Frank 2018). Urban ecosystems are also affected by broader global and regional climate changes, including changes in precipitation, warming or extreme weather events (McCarthy et al. 2010, Bender et al. 2010), interacting with urban biodiversity and ES (Youngsteadt et al. 2015).

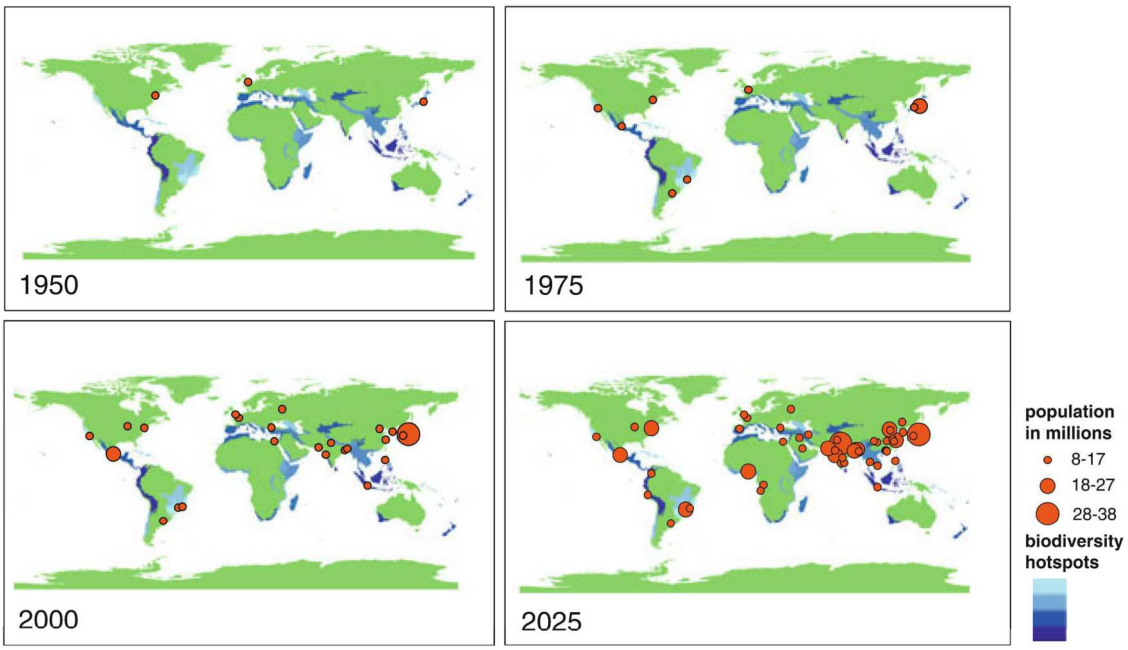


Figure 1.2 – Global projected increase of urban citizens in millions from 1950-2025 and biodiversity hotspots from Seto et al. (2013). The fast growth of urban areas (Seto et al. 2011) will increasingly require more land to build cities and supply urban consumption as population continues to increase (Seto et al. 2013). Together with urban land expansion on hotspots of biodiversity, this will have significant impacts on global ecosystem functioning.

1.3.1 Urban gardens

In many countries, gardens are a major component of urban greenspace and thus can provide considerable benefits for the biodiversity in cities (Goddard et al. 2010). Recent studies indicated that even small and fragmented greenspace elements, such as urban gardens, can provide critical habitat resources (Soanes et al. 2019). There are many studies that highlight the beneficial environmental and ecological impacts of urban gardens, such as improving local air quality (Kulak et al. 2013) reducing the urban heat island effect (Susca et al. 2011) and floods (Bolund and Hunhammar 1999). Moreover urban gardens store considerable amounts of N and C (Edmondson et al. 2012, Beniston and Lal 2012), and provide habitats for many species even in densely urbanised areas (Goddard et al. 2010). However, there are also disservices from urban gardens, mainly related to the use of pesticides and its effects on biodiversity (Wheeler et al. 2017), or human health issues in the case of contaminated soils (Clark et al. 2008, Cheng et al. 2015, Antisari et al. 2015, Brown et al. 2016). Other examples of ecosystem disservices from urban gardens are the amount of water used for plants or the introduction of invasive species (von Döhren and Haase 2015).

ES in urban areas are determined by the species composition, local abiotic factors (Youngsteadt et al. 2015), but also to a great amount by human decision making (Pincetl 2015). However, cities contain also artefacts driven by human decisions including unintended consequences by built infrastructures and by regulation, rules and laws (Pincetl 2015). The

management of residential areas, including urban gardens, has not only a regional impact on biodiversity or soil related functions, but also a continental-scale impact (Padullés Cubino et al. 2019). For example, it has been shown that the management of residential areas in the US has a large impact on water, energy, carbon and nutrient dynamics (Groffman et al. 2017). However, there is a potential for urban greenspace to increase human well-being and resource efficiency by maintaining or enhancing ES (Alberti et al. 2008), and thus contributes to reduce some of the impacts of urbanisation.

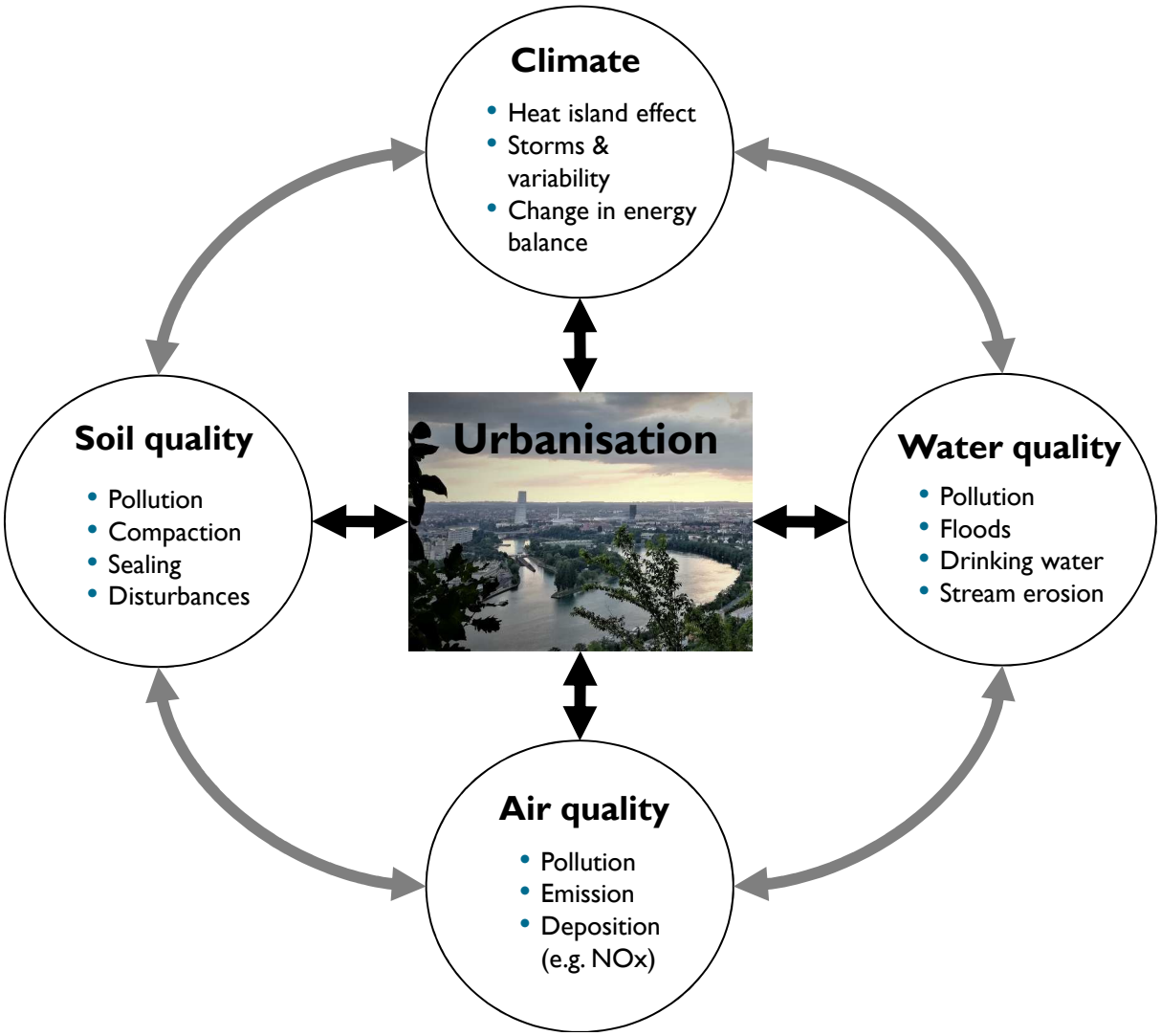


Figure 1.3 – Impact of urbanisation on main components of environmental quality. Modified after Lal (2018a).

1.4 Soils and soil functions

Soil is the skin of the earth and the central interface between biosphere, hydrosphere, atmosphere, and lithosphere (e.g. Greiner et al. 2017). Soil plays a crucial role in providing ES through its soil functions (e.g. Gobat et al. 2004, Adhikari and Hartemink 2016). The Millenium Ecosystem Assessment (MEA 2005) provided four major ES categories: (i) provisioning (e.g. food production e.g. Bender and van der Heijden (2015)), (ii) regulating (climate regulation e.g. Nielsen et al. (2011)), (iii) cultural (e.g. education e.g. Karlen et al. (2003)), and (iv) supporting services (soil as habitats for biodiversity e.g. Briones (2014)). Despite considerable knowledge on soil formation and distribution, the understanding of soil functions and its contributions to ES is still limited (Swinton et al. 2006, Adhikari and Hartemink 2016). Moreover, soil functions are often overlooked components in ecosystem service studies (Hewitt et al. 2015), although studies regularly emphasise the importance of soil ecosystem service assessments (e.g. Robinson et al. 2012, Bouma 2014, McBratney et al. 2014, Baveye et al. 2016, Greiner et al. 2017).

1.4.1 Urban soils

Soils are non-renewable natural resources (Breure et al. 2018) and they play an essential role in the development of human society, but soils are often only regarded as a space for infrastructure or food production (Blumlein et al. 2012). Moreover, they provide and regulate also other essential ES such as water storage and purification, nutrient cycling and storage or habitat provisioning for biodiversity (Bünemann et al. 2018). Soils are complex systems of physical, chemical and biological characteristics and processes interacting with soil organisms or being transformed by them (cf. Gobat et al. 2004). The very slow pedogenesis is driven by five soil forming factors: (i) parental material, (ii) climate, (iii) soil biota, (iv) topography, and (v) time (Jenny 1941). In the case of urban soils an additional sixth soil forming factor: (vi) anthropogenic activities (Pouyat et al. 2010) needs to be considered too (Figure 1.4). These urban soils are affected by the past history, the current land-use and also by the degree of disturbance (Szlavecz et al. 2018). Urban soils are often described as highly disturbed, due to the many anthropogenic actions (Lal 2018a). They differ from undisturbed natural soils and are generally characterised by a high level of compaction, due to various construction works, and the presence of contaminants (Pouyat et al. 2010). Moreover, they are often considered degraded and possibly polluted (Meuser 2010), showing a decreased soil biological activity (Craul 1985). Contamination of urban soils is highly variable in space and time resulting from transport emissions, coal combustion, industrial activities or waste disposal (Luo et al. 2012) contaminating the soil with organic pollutants (persistent organic pollutants such as PAHs, PCBs or DDT) and inorganic pollutants (heavy metals such as Pb, Cd, Cr, Hg, and As) (Li et al. 2018). For instance, organic pollutants are more concentrated in larger and more industrialised cities such as Lisbon (PT) or Glasgow (UK) compared to smaller and less urbanised cities such as Viseu (PT) or Tarragona (ES) (Li et al. 2018). Heavy metal concentrations can also be caused by a geogenic origin, except for Pb (Laidlaw and Filippelli 2008), but they are often found in larger concentrations than the corresponding background values, indicating the impact of anthropogenic activities (Li et al. 2018). Concentrations of heavy metals and persistent organic pollutants are generally larger in city centres and can be aligned with traffic intensity as well as point sources of industrial activities (Luo et al. 2012).

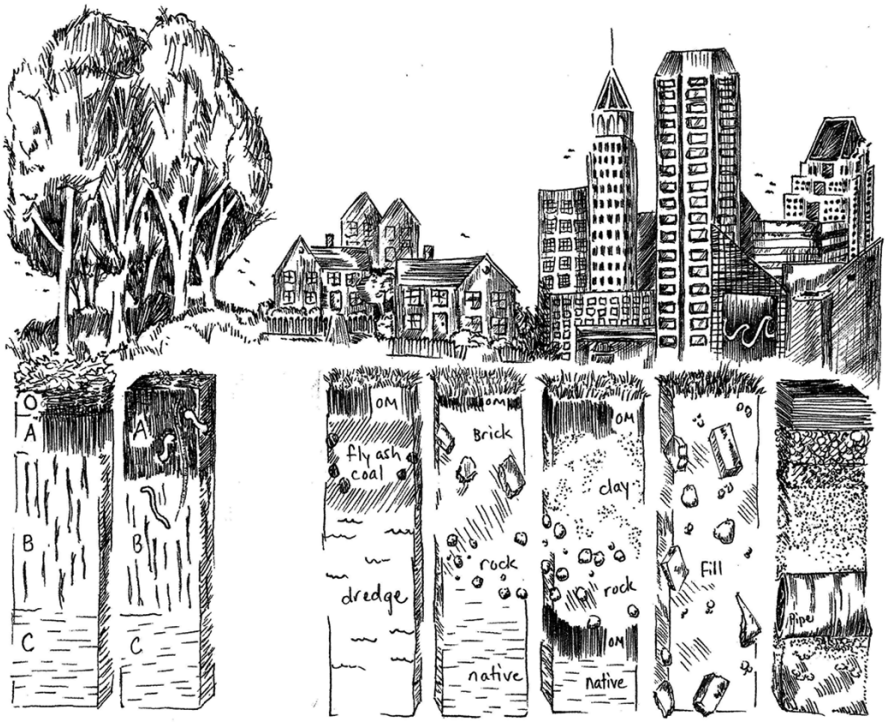


Figure 1.4 – Diverse urban soils (on the right side) are influenced by a combination of natural and anthropogenic soil forming factors, including management and site history. Semi-natural soils (on the left side), are less strongly influenced by humans. Even under the same land-cover type (i.e. grass) the soil profile can be considerably different (Szlavecz et al. 2018), which was also the case for urban garden soils in Zurich. Source of the picture: Yesilonis in Szlavecz et al. (2018).

Contaminations bear a large burden for many urban soils of both industrialised (Filippelli and Laidlaw 2010) and developing countries (Trujillo-González et al. 2016). However, soils also represent an important environmental sink for contaminants (Wang et al. 2018). Furthermore, it has to be mentioned that besides the concentration of pollutants, the bioavailability to humans is also a critical issue. Many factors can influence the bioavailability, such as the particle size or the amount of SOC (Cachada et al. 2016), and thus should be integrated into a health risk assessment. Overall, it can be concluded that urban sites close to traffic routes often show increased levels of heavy metals (Ferreira et al. 2016) and persistent organic pollutants (PAHs, PCBs, PAEs; Wang et al. (2018)), directly related to the frequency of vehicles, resulting form tyre particles and weathered street surfaces (Ferreira et al. 2016). As a consequence of the soil contamination and drastic anthropogenic disturbances, such as removing the topsoil during the period of construction (Chen et al. 2013), a decreased soil quality and ability for soil functions have been reported (Lal 2018b, Szlavecz et al. 2018). Those highly disturbed soils are low in biological activity (Lorenz and Kandeler 2005, Scharenbroch et al. 2005), SOC (Bradley et al. 2005) and water holding capacity (Szlavecz et al. 2018). Therefore, these soils may not be suitable for crop production (Jim 1998). Besides pollutants and direct anthropogenic effects (Lorenz and Lal 2009), urban soils are also affected by increased air temperature (Figure 1.3), known as the urban heat island effect (Oke 1995). For instance, temperatures in the centre of Baltimore (US) were 5 to 10°C warmer than in the surrounding areas (Brazel et al. 2000). These increased air temperatures also cause higher soil temperatures (Pouyat et al. 2007) and affect soil microbial activity (Carreiro et al. 2009).

In the case of carefully managed soils, such as most urban garden soils, the long tradition of organic soil management practices (Bretzel et al. 2018) favoured the development of soils with increased biological activity and organic matter content (Edmondson et al. 2014). These soils can contain even higher organic carbon contents than those in the surrounding rural landscapes (Edmondson et al. 2012), with levels of soil biological activities comparable to those in forest soils (Joimel et al. 2017). Hence, despite intensive soil use, urban garden soils can be of high ecological and ecosystem service value (Edmondson et al. 2014, Levin et al. 2017). Nonetheless, as Joimel et al. (2017) pointed out, there is a need for urban soil quality assessments, especially regarding biological soil quality indicators, evaluating anthropogenic influences on soil quality and functioning of urban soils.

Table 1.1 – Summary of important soil functions in urban ecosystems including the contribution of soil fauna adapted from Goddard et al. (2010), Dearborn and Kark (2010), Haase et al. (2014), Setälä et al. (2014), Pavao-Zuckerman (2012), Amossé et al. (2016), Filser et al. (2016), Schwarz et al. (2017), Szlavecz et al. (2018).

Soil functions	Role of soil fauna	Examples of important taxa
Decomposition	Shredding and consumption of organic matter, chemical breakdown	Various macrofauna (Earthworms, isopods, ants), mesofauna (springtails, mites, nematodes), and microfauna (bacteria and fungi)
Carbon sequestration	SOM build up	Various microorganisms (bacteria, fungi) also ecosystem engineers such as earthworms
Nitrogen cycling	N fixation, N mineralisation	Mainly bacteria
Water infiltration and storage	SOM, soil pores	Various macrofauna Plant roots
Plant growth	Nutrient availability, food production, soil structure, bioturbation	Microorganisms (fungi, bacteria) Macrofauna (earthworms, ants)
Soil structure formation	Binding and forming clay-humus complexes, mixing mineral and organic soil material	All soil fauna groups
Pest control	Regulating populations of harmful species	Predatory soil macro- and mesofauna (spiders, beetles, mites, nematodes)
Soil fertility	Maintaining and increasing soil fertility	All soil fauna groups
Resistance against disturbances	Better withstand disturbances, faster recovery after disturbance	Entire soil biota network
Habitat provisioning	Creating new habitats	Larger macrofauna especially ecosystem engineers (earthworms, termites,ants)
Resource for wildlife	Food for insectivores	Especially larger soil macrofauna (earthworms, larvae, snails)
Gene pool	Resource for future pharmaceutical products	Microfauna (bacteria, fungi)
Cultural services, e.g. aesthetics, education, inspiration, recreation	Species diversity, good soil quality, wildlife processes for gardening activities	Macrofauna (earthworms), mesofauna

1.4.2 Urban soil functions

Soils of urban ecosystems support and regulate many functions that are important for urban ES, e.g. decomposition, nutrient cycling, and C sequestration (Table 1.1; Szlavecz et al. (2018)). In urban areas, some of these soil functions are of great importance such as the potential of non-sealed soils to infiltrate water and thus prevent extreme surface runoff during high intensity rain events. Other functions can only be found in urban soils such as food waste removal. The consumption of littered food waste by arthropods can reduce the availability of food to less desirable fauna such as rats (Youngsteadt et al. 2015).

Urban soil functions are important to increase liveability in cities and contribute to the SDG goal 11 on sustainable cities (United Nations 2015b). However, soil degradation by sealing, compaction, contamination or erosion (Stolte et al. 2016) has several detrimental impacts on urban welfare and sustainability (Ferreira et al. 2018). For instance soil degradation diminishes important ES in urban areas (Pereira et al. 2018) and thus increases the vulnerability to natural hazards such as floods or heat waves in cities (Ferreira et al. 2018). Degraded urban soils can create soil disservices such as the loss of nutrients or biodiversity, release of stored C or decreased water holding capacity. Therefore soil management is of crucial importance for the potential for urban ES (Shuster and Dadio 2018).

1.5 Soil quality

The quality of soils is one of the main components of environmental quality (Figure 1.3, Andrews et al. (2002)). While the concept of integrating indices has been initially developed to assess water quality (Karr 1981), it has been adapted to assess soil quality at the beginning of the 1990s (Larson and Pierce 1994). However, soil quality can not be defined by the degree of pollution, the way it is done in air or water quality (Bünemann et al. 2018). A common definition of soil quality is given by Doran and Parkin (1994) defining soil quality as "the capacity of a soil to function within ecosystem and land-use boundaries to sustain biological productivity, maintain environmental quality, and promote plant and animal health", including human health. This complex definition is not only referring to the multifunctionality of soils being essential for many ES, but to the site specificity of the soil properties as well (Bünemann et al. 2018). Hence, soil characteristics must be carefully chosen to include the consideration of soil functions (Karlen et al. 1997). A lot of effort has gone into developing minimum datasets necessary to characterise soil quality (e.g. Doran and Parkin 1994, Schindelbeck et al. 2008). Many studies proposed minimum datasets based on expert knowledge (Bünemann et al. 2018, Rinot et al. 2019) with subsequent multivariate statistical reduction methods (e.g. Andrews and Carroll 2001, Shukla et al. 2006, Lima et al. 2013, Rojas et al. 2016, Fine et al. 2017). Soil properties which do not show much variation in a given study will be excluded in such a minimum dataset. The selection of soil properties is always a crucial step and and can include four principles (Bünemann et al. 2018, Rinot et al. 2019): measurements should (i) reflect main physical, chemical and especially biological soil processes, which are often underrepresented or even missing (Bünemann et al. 2018), (ii) be sensitive to targeted soil functions influenced by land management and (iii) at the same time should be relatively efficient in terms of cost and time. However, there are also different approaches for the selection of appropriate soil properties, such as the participatory approach (Ritz et al. 2009, Stone et al. 2016), but often this reduction from a larger set of soil quality indicators is a necessary step to control for financial and temporal limitations (Bünemann et al. 2018). Originally, the concept of soil quality has been developed for agricultural soils. Recently, soil quality has been increasingly included in sustainable land management (Hurni et al. 2015), land restoration (Schwilch et al. 2012), environmental change assessments (Sonneveld et al. 2010), or in socio-economic policy assessments (Robinson et al. 2017). The concept of soil quality most often addressed soil functions, providing and controlling ES. Thus several studies suggest that soil quality can only be assessed in relation to soil functions (e.g. Baveye et al. 2016, Volchko et al. 2014). In the case of several soil functions an overall soil quality index is often not meaningful. Overall, the selection of soil properties for a soil quality assessment in urban ecosystems must be chosen carefully in order to take into account the large differences in urban soils, as a result of land-use history and industrial facilities in close proximity (Lal 2018b).

1.6 Urban gardening

Urban gardening, providing fresh food for the growing urban population and vital environmental benefits for cities, is currently experiencing a global revival (Schram-Bijkerk et al. 2018, Egerer et al. 2018b). The production of food in urban gardens is the most obvious provisioning ES of soils (Szlavecz et al. 2018). Urban gardening is considered an im-

portant way of alternative food production (Zezza and Tasciotti 2010), due to the supply of fresh and local products (Deelstra and Girardet 2000) and reduced transportation (Tan and Jim 2017). Although the global scale of food provision from urban agriculture is difficult to assess (Thornton 2008, Siegner et al. 2018), it is estimated that approximately 800 million people are to a great extent reliant on food from urban gardens (Lee-Smith 2010). The human population growth is expected to increase the shortage in food supply in large cities (Tan and Jim 2017), particularly for those who have limited access to markets (Alaimo et al. 2008). Therefore, urban gardening can be an important resource to enhance urban food system sustainability (Wiskerke 2015).

Overall, the management of soil, water and vegetation through urban gardeners play an important role in the quality of the physical environment such as cooling of air temperatures or water storage, and the quality of the living environment, by contributing to well-being and interaction of citizens (Lin et al. 2015, Breure et al. 2018, Home et al. 2012, Hofmann et al. 2018, Egerer et al. 2018b).

1.7 Project *BetterGardens*

The *BetterGardens* project is a multidisciplinary project between the Research Institute of Organic Agriculture (FiBL) and the Federal Institute of Wood Snow and Landscape (WSL). Two humanities subprojects (SPA and SPB) examined the social and psychological influences of gardening as well as the factors influencing the behaviour of gardeners. The other two natural science subprojects assessed the impacts of gardening on biodiversity and ES. This thesis is part of the third subproject C (SPC, Figure 1.5) and has been conducted in close collaboration with the SPD.

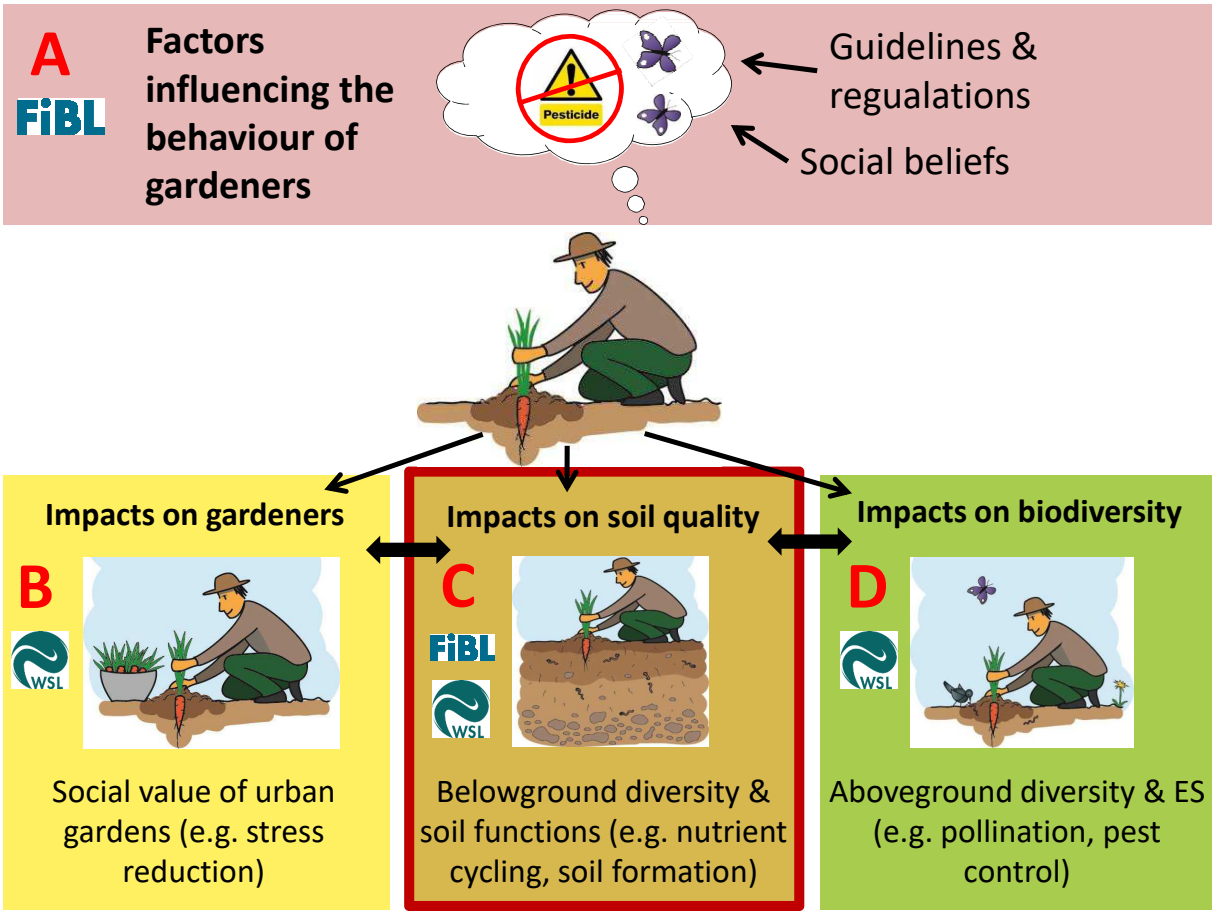


Figure 1.5 – *BetterGardens*: Integrated analysis of soil quality, biodiversity and social value of urban gardens in Switzerland. The multidisciplinary project *BetterGardens* can be divided into four subprojects (A-D). This thesis was part of the third subproject C (SPC) outlined in red.

Overall, one of the main aims of the *BetterGardens* project was to collect data about the biodiversity of plants and invertebrates of urban gardens. This was established by a large biodiversity assessment on 170 sites in 85 urban gardens of Zurich conducted by SPC and SPD. In addition to the biodiversity assessment, SPC focused on the impacts of gardeners on soil quality and soil functions. Although there is a general high interest in urban gardening activities (Lin and Egerer 2017, Ossola et al. 2018), urban green spaces are being contested by other mainly economical interests (Tappert et al. 2018), leading to a net decline of urban garden areas in many cities of Switzerland. In Zurich, the approximately 5500 allotment gardens cover 3 % of the city's settlement area (Grün Stadt Zurich 2010). The total garden area of Zurich is about 167 ha, which is comparatively larger than the area of sports facilities (136 ha) or parks (141 ha) (Grün Stadt Zurich 2018). However, urban gardens in Switzerland are facing an ongoing decline since the last decades. For example in Zurich they declined around 7 % from 2004 to 2015 (Grün Stadt Zurich 2018).

1.8 Research gaps and overall research questions

Soils play a crucial role in providing urban ES (Setälä et al. 2014). While considerable knowledge exists on how agricultural soils are being affected and modified by management practices such as tilling or the application of fertilisers and plant protection agents, little is known in the case of urban garden soils (Lin et al. 2015). There has been a long tradition and history of soil management by gardeners (Christl et al. 2004). Although, urban gardens are used to provide a considerable amount of food for citizens (Lee-Smith 2010), besides other important ES (cf. Table 1.1), little is known about the impact of management practices on soil quality and functions (Edmondson et al. 2014, Bretzel et al. 2018, Egerer et al. 2018b).

Despite the long tradition of urban gardening, the effects of different management practices on soil quality, soil fauna diversity and related soil functions are poorly investigated so far. This thesis is separated into four main chapters: the first chapter encompassed the measurements and data background of an urban soil quality assessment. In the second chapter effects of management practices on soil quality indicators and soil functions have been analysed. The third chapter investigated the soil function litter decomposition, while in the last chapter the relationship between aboveground and belowground diversity in urban gardens and its effect on soil multifunctionality has been analysed.

In order to address the respective research questions, we established an urban soil quality assessment with 36 soil quality indicators. The measurements of these indicators have been chosen carefully based on existing soil quality assessments from the literature (cf. Bünemann et al. (2018)). Furthermore, indicators, such as eco-physiological soil indices (Anderson and Domsch 1989), sensitive to soil management practices, were selected. Next, together with the large set of physical, chemical, and biological soil quality indicators we analysed which factors from the garden and landscape scale influenced the grouping of soil quality indicators (second chapter). Besides indicators of soil functions, we measured decomposition of organic material directly by using litter bags with different mesh sizes, to distinguish between macro- and mesofauna decomposition. In the third chapter, effects of garden management practices, plant and soil fauna species richness and the degree of urbanisation on litter decomposition has been analysed, together with the litter residue quality after decomposition. The last chapter was dedicated to the relationship between plants and soil fauna communities and related soil functions. In addition, plant ecological indicator values, reflecting the plant environmental requirements, based on all cultivated and spontaneously growing plants in the gardens, were used to characterise the anthropogenic plant assemblages and its effect on soil fauna communities and soil functioning.

1.9 Study sites

SPC and SPD were conducted in urban gardens of Zurich, a European city of average size with approximately 0.4 million inhabitants. There are two main types of gardens in Zurich, allotment gardens or family gardens and home or private gardens, which also represent the two most common garden types worldwide (Lin et al. 2017). The third form of urban gardening, community gardens, has a minor importance for the total garden area of Zurich. There is a long history and tradition of allotment gardens in Zurich, going back to self-supplying citizen gardens in the 16th century (Christl et al. 2004) until the first establishment of allotment gardens at the beginning of the 20th century (Bell et al. 2016).

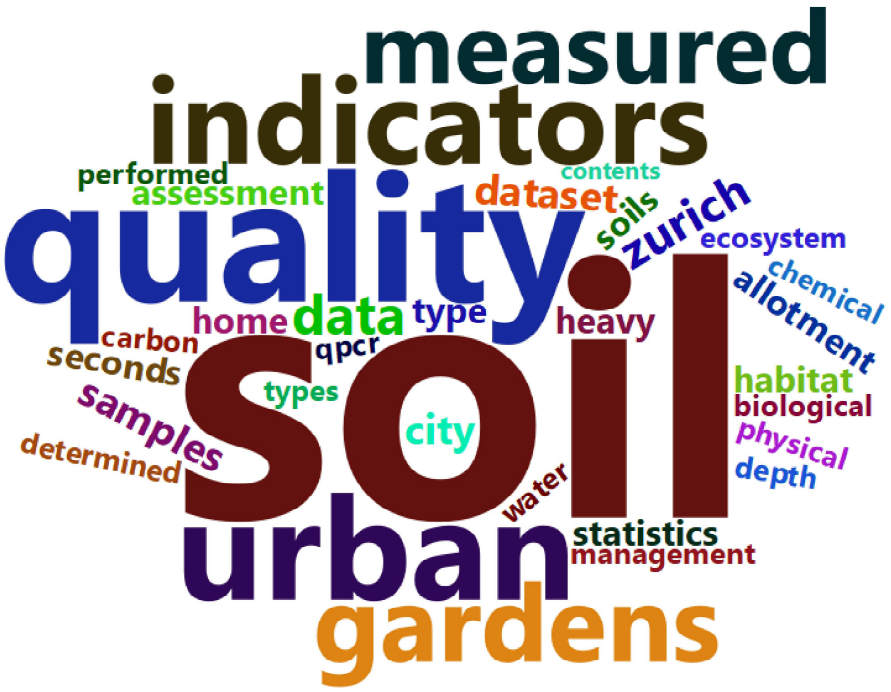
In this project we defined private or home gardens as gardens adjacent to single-occupancy houses and allotment gardens as publicly provided urban gardening lots for tenants (see examples of each type

of garden in Figure 4.1). The sampling design had to match the aims of both SPC and SPD under consideration of different sociological backgrounds of the gardeners for SPB. In close collaboration with SPD, 85 urban gardens consisting of 43 home and 42 allotment gardens have been selected in the first phase of the project. The gardens were chosen according to a stratified sampling design by evaluating aerial images and by visual inspection during field visits. We used three independent strata for the garden selection, (i) the type of garden (home vs. allotment), (ii) the management intensity, and (iii) the degree of urbanisation. The garden management intensity was assessed on-site, ranging from intensively managed gardens to very natural gardens. The urbanisation density was assessed by the location and the information about the built and sealed surface area around the garden and ranged from densely built-up areas to peripheral areas within the city margins (Figure 4.1). To ensure best statistical independence among observations (i.e. gardens) no adjacent garden lots were sampled and gardens were maximally spaced across the entire city to include all urban districts. Additionally, with two exceptions, only one garden lot was sampled per allotment garden area. However, it has to be noted that we did not sample front or easement gardens, where a spatial autocorrelation is usually detected, and that we focused on medium sized gardens with an average size of 312 $\pm$ 154 m^2. The average distance among gardens was 4.5 $\pm$ 2.2 km. More information about the garden selection and establishment of contact with the gardeners can be found in Young et al. (2019). Given the spatial and temporal complexity of urban environments and especially of urban soils, we decided to focus on one city in order to consider all possible spatial components affecting urban soils and increase the statistical power of the stratified sampling design. In addition, we decided to select two sampling sites of 2 m x 2 m in each of the 85 urban garden in order to account for expected differences in soil characteristics between different garden land-use types such as annual vegetable beds or perennial grass sites. Consequently, soil analyses and the biodiversity assessment of soil fauna has been done on those 170 selected sites.

Figure 1.6 – Increasing pressure on urban gardens due to ongoing soil sealing and rising land prices. Picture from autumn 2015 close to an investigated urban garden site in the city of Zurich.

2

Urban soil quality assessment - A comprehensive case study dataset of urban garden soils



Text mining: Most frequent 35 words from chapter 2, created with the "wordcloud2" R package (Lang 2016).

Urban soil quality assessment - A comprehensive case study dataset of urban garden soils

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Keywords: Urban soil quality assessment, Soil quality indicators, Urban heavy metal contents, Urbanization, Urban gardening, Allotment gardens, Home gardens, Private gardens

Frontiers in Environmental Science
DATA REPORT ARTICLE
Published: 13 November 2018
DOI: 10.3389/fenvs.2018.00136

2.1 Introduction

Soil is the foundation of ecosystem functioning in urban green spaces and provides key ecosystem services for a livable city (Zhu et al. 2018). Urban soils are mixture of natural soil-forming factors and anthropogenic activities (Shuster and Dadio 2018) and require therefore an adapted set of indicators for a soil quality assessment. Soil quality is one the three constituents of environmental quality, along with water and air quality (Andrews et al. 2002) and is generally referred to as 'the capacity of a soil to function within ecosystems and land-use boundaries to sustain biological productivity, maintain environmental quality, and promote plant and animal health', including human health (Doran and Parkin 1994). Here we present a comprehensive dataset of 37 soil quality indicators measured at 170 plots in 85 urban gardens within the city of Zurich, Switzerland. They represent all major inherent and dynamic soil properties for a soil quality assessment (Bünemann et al. 2018) including respectively eight physical, nine chemical and eleven biological soil quality indicators. Moreover, we provide data about ten soil heavy metal contents (As, Ba, Co, Cr, Cu, Ni, Pb, Sb, V, Zn). Results on the influence of garden management, local and landscape effects on the distribution of soil quality indicators can be found in (Tresch et al. 2018a). This dataset is useful for future studies on urban soil quality, ecosystem services or for modelling purposes, such as carbon dynamics or greenhouse gas inventory models in cities.

2.2 Material and Methods

2.2.1 Study sites and design

The dataset of 36 soil quality indicators (Figure 1) was measured on 85 urban gardens in the city of Zurich (Switzerland), comprising 42 allotment and 43 home gardens (Figure S 2.1). These two garden types are the most common in Switzerland, but also worldwide (Lin et al. 2017). Allotment gardens represent a plot of land rented by gardeners, usually located in urban or semi-urban areas, while home gardens are often situated around private houses. This study design is part of the interdisciplinary project BetterGardens (www.bettergardens.ch) that focuses on soil quality, biodiversity, ecosystem services and human wellbeing of urban gardens in Switzerland. Two plots within each garden were selected according to garden habitat and management practices such as lawn, vegetables or flower and berry beds. Five soil samples (0-20 cm depth, 3 cm wide soil auger Eijkelkamp, NL) were collected randomly within an area of 2 m x 2 m on the 170 plots, prior to the first soil gardening practices in March 2015. Soil samples were pooled for each plot, gently air dried, homogenized and sieved at 2 mm then stored at 4°C. Subsamples for chemical analysis were air dried at 20°C and stored under cool and dry conditions. For physical soil quality indicators, undisturbed samples were collected with three soil cores (5 cm diameter and depth, Eijkelkamp, NL) at a depth of 10-15 cm on each plot. Soil quality analyses were performed in accordance with Swiss standard methods for soil characterization (Agroscope 2012).

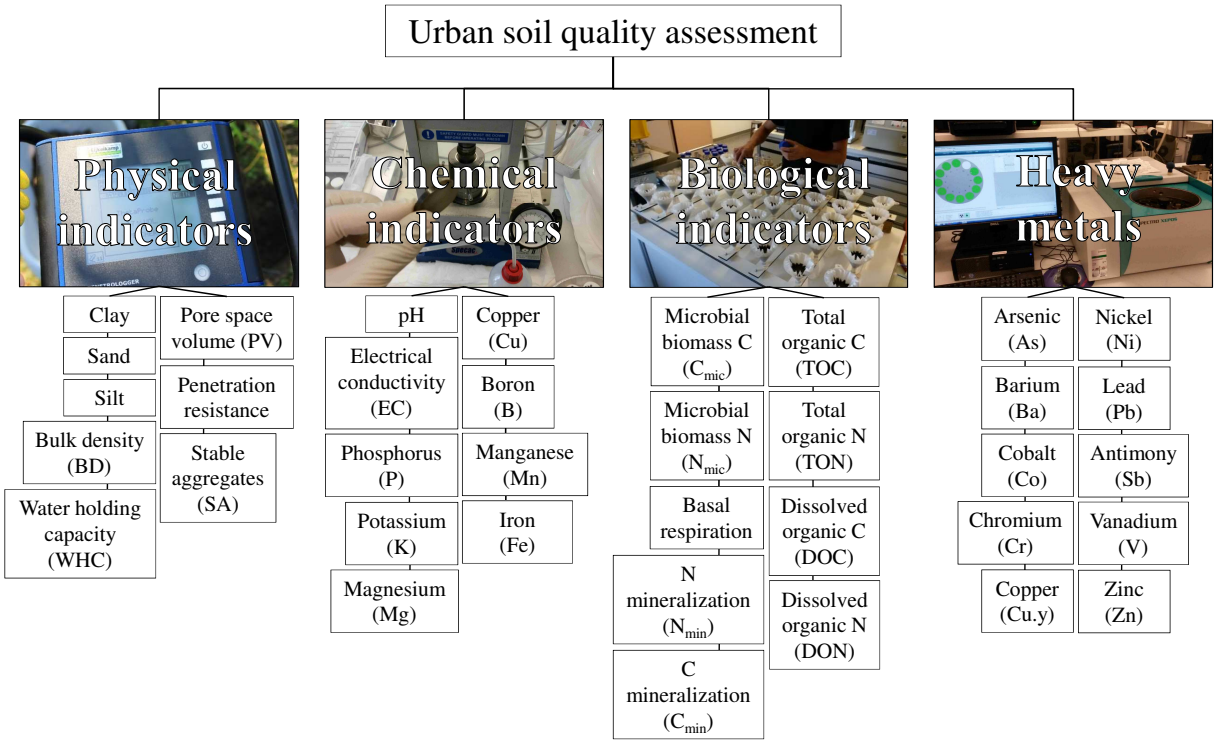

Figure 2.1 – Concept of urban soil quality indicators measured in gardens of Zurich, Switzerland. Details about the soil quality indicators can be found in Table S 2.2.

2.2.2 Physical soil quality indicators

Soil texture (i.e. the proportions of clay, silt and sand) was measured by a combined sieving and sedimentation technique (Schinner et al. 1996). Soil water holding capacity (WHC) was determined by a cylinder method on a sand bath (Schinner et al. 1996). Soil aggregate stability (SA) was determined from aggregates in the 1-2 mm fraction by wet sieving for 5 min on a multi-sieve device (42 cycles min^{-1}) and subtracting the sand fraction (Schinner et al. 1996). Soil bulk density (BD) and pore space volume (PV) were determined with the undisturbed soil cores following Swiss reference methods (Agroscope 2012). Penetration resistance was measured with a Penetrologger (cone type: 100 mm^2, penetration speed: 0.002 ms^{-1}; Eijkelkamp, NL), recording the penetration resistance every 1 cm down to a soil depth of 80 cm. Ten replicated measurements were taken and mean values from 0-20 cm soil depth were calculated for the penetration resistance.

2.2.3 Chemical soil quality indicators

Soil pH and electrical conductivity (EC) were measured in a soil suspension with deionized water (1:2.5 w/v). Soil nutrient contents (P, K, Mg, Cu, B, Mn, Fe) were measured at an external certified laboratory with ammonium acetate-EDTA.

2.2.4 Biological soil quality indicators

Total organic carbon (TOC) and nitrogen (TON) were determined by a CHN analyzer (Thermo Scientific Flash EA 1112, NL) after removing carbonates with 2 M HCl. Dissolved organic carbon (DOC) and nitrogen (DON) as well as mineralized N (N_{min}), were measured in an extract with 0.01 M CaCl2 (1:4 w/v) (Krauss et al. 2017). Soil microbial biomass carbon (C_{mic}) and nitrogen (N_{mic}) contents were determined by chloroform-fumigation-extraction (CFE) (Fliessbach et al. 2007). Soil basal respiration and C mineralization (C_{min}) were assessed by measuring CO2 evolution in defined intervals from soil samples using a gas chromatograph (TCD detector; 7890A, Agilent Technologies, USA) as described in Tresch et al. (2018a). Basal respiration rates were recorded after 1 week and C_{min} after 4 weeks of soil incubation at 20°C.

2.2.5 Heavy metals

Heavy metals in soils were measured as total element concentrations analyzed with dried and ball milled soil samples pressed with wax to tablets on an X-Ray Fluorescence device (XRF, X-lab 2000, SPECTRO Analytical Instruments, DE).

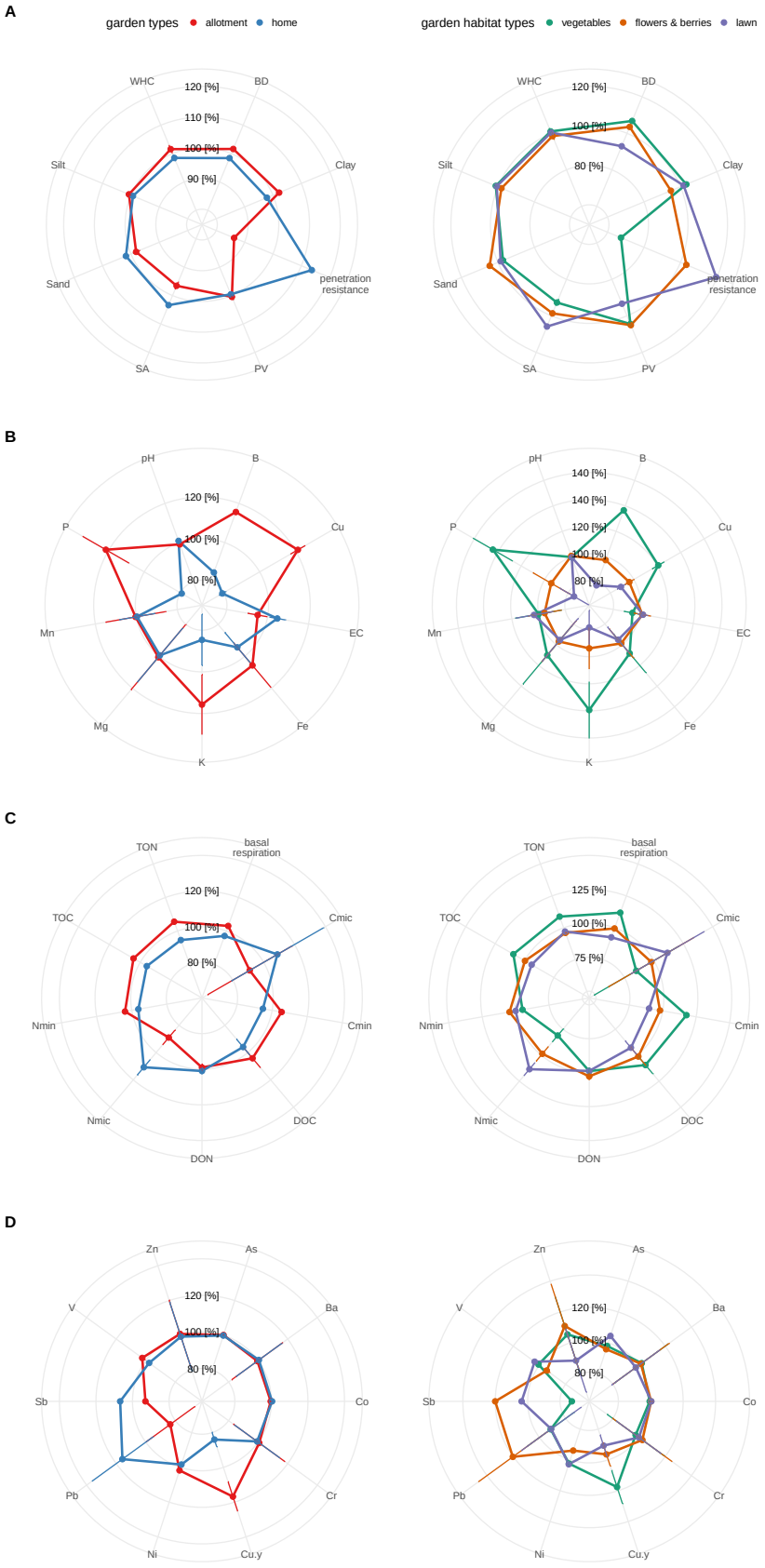

Figure 2.2 – Soil quality indicators per garden type (left) and habitat type (right) consisting of A) eight physical indicators, B) nine chemical indicators, C) nine biological indicators, D) ten heavy metal measurements. Data points represent the relative deviation [%] from the grand mean (Table S 2.2) that is set to 100 %. Standard error bars are shown with lines.

2.2.6 Statistics

All statistics were performed using R software version 3.4.2 (R Core Team 2017). Descriptive statistics (dplyr) and visualization (ggplot2) were obtained with the data manipulation package tidyverse (Wickham and Francois 2016) and the spatial plot (Figure S 2.1) with the package ggmap (Kahle and Wickham 2013). Example R codes (R project files) including the raw data (csv file) are provided in the supplementary material.

2.3 Conclusion

The dataset comprises a collection of soil quality indicators for urban garden soils. We measured eight physical, nine chemical, eleven biological soil quality indicators as well as ten heavy metal values (Figure 1), that are necessary for a comprehensive soil quality analysis in an urban context. This dataset was sampled at a scale of an entire representative medium-sized European city, Zurich CH (Figure S 2.1). The data is split according the two most common urban garden types: allotment and home gardens (Table S 2.1). Furthermore, garden habitat types were distinguished in vegetable beds (i.e. annual vegetable plants), flower beds and berry cultivations (i.e. perennial flowers, roses and berry shrubs), and lawn (i.e. meadows and turf). Descriptive statistics are given in Table S 2.2. A graphical representation of percentage deviations from the overall mean value split by garden and habitat type is given in Figure 2. In summary, this dataset provides information about a city-wide soil quality assessment of urban gardens. This data can be used for comparing soil properties among different cities or land use types. Moreover, our study may help analyzing the effect of garden management or urbanization on soil quality (see Tresch et al. (2018a)) or provide data for modelling carbon dynamics in urban soils or other soil based ecosystem services.

AUTHOR CONTRIBUTIONS

AF, MM, R-CL, PM, DF, and ST conceived and designed the research. ST, DF, and AZ performed the field work. Lab work was done by ST, AK, AM, and BS. ST analyzed the data. All authors contributed to the writing of the manuscript.

FUNDING

This work was part of the interdisciplinary project BetterGardens (www.bettergardens.ch), funded by the Swiss National Science Foundation in the frame of the Sinergia program (CRSII1_154416).

ACKNOWLEDGMENTS

We thank the project coordinator Dr. Robert Home for his integrative work. We are grateful to Prof. Dr. Rainer Schulin and Björn Studer for the XRF-analysis at ETH and Roger Köchlin for the soil texture analysis at WSL. We acknowledge in particular the willingness and interest of the 85 participating gardeners to give us access to their gardens and to have us dig and sample soils on their property.

SUPPLEMENTARY MATERIAL

The Supplementary Material for this article including R scripts and raw data files can be found online at: www.frontiersin.org/articles/10.3389/fenvs.2018.00136/full#supplementary-material

Conflict of Interest Statement

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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2.4 Supplementary Tables and Figures

2.4.1 Supplementary Figures

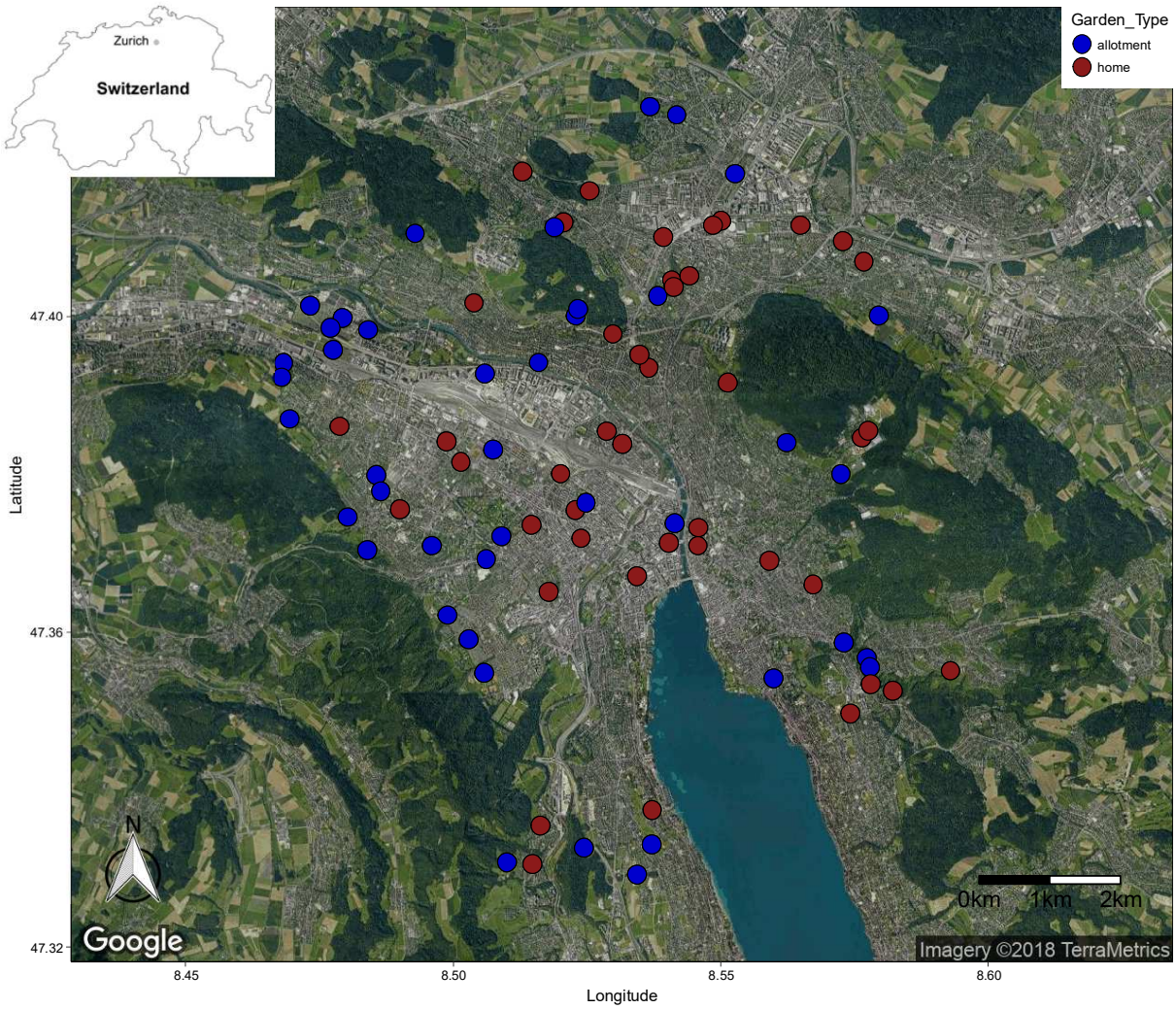

Figure S 2.1 – Spatial map of selected urban garden sites (N=85) within the city of Zurich, Switzerland. In total, 42 allotment gardens (blue) and 43 home gardens (red) were analyzed.

2.4.2 Supplementary Tables

Table S 2.1 – Urban garden sites measured in the city of Zurich, Switzerland. In total 85 urban gardens were selected, 42 allotment and 43 home gardens according to a systematic nested design of garden management intensity and degree of urbanization, for more information see Frey et al. (2018). Within each urban garden two distinct measurement plots (2 m x 2 m) were selected, corresponding to a typical garden habitat type (Tresch et al. 2018a).

	garden types		
garden habitat types	**allotment**	**home**	total sites
lawn	29	42	71
flowers & berries	19	33	52
vegetables	36	11	47
total sites	84	86	170

Table S 2.2 – 19 substrates used for the assessment of the Community level physiological profile (CLPP) based on the MicroResp™ technique (Campbell et al. 2003). We dissolved 18 substrates in H_2O_{demin} and added 25 µl aliquots to deliver 30 mg of C-substrate per g of soil water for each well. Each substrate was measured in five technical replicates. The absorbance of the detection plate is measured at 570 nm after 5 hours of incubation at 20°C in the dark. The detection plate contains a pH sensitive dye (Cresol Red) which is dissolved in a solution with 150 mM potassium chloride (KCl) and 2.5 mM sodium bicarbonate ($NaHCO_3$) in a matrix of 1% agarose gel. For the calibration equations 44 samples from five different soils together with four different quantities (10g, 20g, 30g and 40g) were amended with 0, 0.5, 2, 3, 5 and 10 mg of glucose or α-keto-glutaric acid per g soil. The substrates were dissolved in water so that 62.5 µl per g soil was added to each sample. Samples without substrates received the same amount of water. The calibration was obtained in 100ml Schott bottles containing 4 wells of breakable microstrips filled with the detection gel. These microstrips were measured immediately before and after the incubation on a plate reader (MRX II TC, Dynex, USA) at 570 nm. The bottles were sealed and CO_2 evolution was measured on a gas chromatograph (7890A, Agilent Technologies, USA). The difference in absorbance between the first and the second measurement is then plotted against the log of CO_2 evolution measured by the gas chromatograph. The linear fit between measured log(CO_2) concentrations $[\mu g CO_2 - C g^{-1} h^{-1}]$ was $y = -4.67 + 2.90$ with an $R^2$of 0.87.

Compound category	Substrate	Abbreviation
Amino acid	Gamma-aminobutyric acid	GABA
	Alanine	Ala
	Aspartic acid	Asc
	Glutamine	Gln
	Leucine	Leu
	Cysteine	Cys
Amino sugar	Glucosamine	Glca
Sugar	Arabinose	Ara
	Galactose	Gal
	Glucose	Gluc
	Fructose	Fruct
Carboxylic acid	Ascorbic acid	Asc
	Citric acid	Citr
	Malic acid	MA
	Alpha-keto-glutaric acid	KGA
Phenolic acid	Protocatechuic acid	Prot
	Vanillic acid	Van
Hemicellulose	Xylan	Xyl
Water	Destilled water	H_2O

Table S 2.3 – **qCPR assays for fungal and bacterial gene copy numbers.**

We extracted DNA from 135 mg of lyophilised soil using the FastDNA™-96 Soil Microbe DNA Kit (MP Bio). qPCR assays were conducted on a BioRad CFX™ Real-Time system with a C1000 Touch™ Thermal Cycler (BioRad Laboratories). qPCR assays were performed to estimate the gene copy number of bacterial 16S rDNA and fungal 18S rDNA. All reactions were performed in 15 µl volume containing 7.5 µl KAPA SYBR FAST universal qPCR Master Mix (2x) (KAPA Biosystems), 1.5 µl DNA sample. qPCR reactions for the estimation of the bacterial copy number contained 1.8 µl of each primer (BactQuant, Liu et al. (2012)) and 2.4 µ l H_2O. qPCR reactions for the estimation of fungal copy number contained 0.75 µl of each primer (FR1/FF390, Vainio and Hantula (2000)) and 4.5 µl H_2O. The assays were run in duplicates with an appropriate standard dilution series containing the target region in triplicate. For the 16S assay the PCR conditions were 3 minutes at 95°C followed by 40 cycles of 15 seconds at 95°C, 15 seconds at 62°C and 30 seconds at 72°C. For the 18S assay the PCR conditions were 3 minutes at 95°C followed by 36 cycles of 15 seconds at 95°C, 15 seconds at 50°C and 30 seconds at 72°C with a final elongation step of 10 minutes at 72°C. After each assay melting curve analysis was performed to make sure fluorescence signals originated from specific PCR products instead of primer dimers.

Table S 2.4 – Descriptive statistics of soil quality indicators in urban gardens of Zurich, Switzerland. Values per garden (Table S 2.5) and habitat types (Table S 2.6) and the functions and R packages used for the data management can be found in the appendix. SE represents standard errors. Tea bag decomposition values were assessed according to Keuskamp et al. (2013).

		N	Mean±SE	Median	Min	Max	Variance
Physical indicators							
BD	$[gcm^{-3}]$	170	1.08±0.01	1.08	0.57	1.45	0.03
Clay	[%]	170	23.79±0.43	22.98	9.40	39.25	31
Penetration resistance	$[MPa]$	168	1.42±0.04	1.36	0.36	3.28	0.3
PV	[%]	170	40.86±0.56	41.27	1.47	54.96	54
SA	[%]	170	82.22±0.83	85.57	46.64	95.69	118
Sand	[%]	170	42.06±0.72	41.75	13.75	71.95	87
Silt	[%]	170	34.15±0.4	34.15	18.65	49.15	28
WHC	[%]	170	81.56±0.93	80.36	54.17	145.99	146
Chemical indicators							
B	$[mgkg^{-1}]$	170	1.37±0.05	1.29	0.14	3.88	0.5
Cu	$[mgkg^{-1}]$	170	32.30±2.28	23.67	3.04	209.10	885
EC	$[\mu Scm^{-1}]$	170	184.41±3.12	181.95	82.20	354.00	1653
Fe	$[mgkg^{-1}]$	170	369.29±8.48	360.00	154.50	699.80	12214
K	$[mgkg^{-1}]$	170	165.82±9.63	122.14	43.91	831.34	15770
Mg	$[mgkg^{-1}]$	170	516.42±13.18	502.90	143.60	1125.00	29523
Mn	$[mgkg^{-1}]$	170	296.24±8.38	265.95	93.27	632.50	11936
P	$[mgkg^{-1}]$	170	189.60±9.31	171.38	5.19	465.19	14735
pH		170	7.27±0.02	7.30	6.25	7.75	0.06
Biological indicators							
basal respiration	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	170	0.24±0.01	0.21	0.08	0.72	0.01
C_{mic}	$[mgkg^{-1}]$	170	808.27±20.92	787.59	279.91	1593.98	74383
C_{min}	$[gCO_2 - Ckg^{-1}]$	170	0.15±0.01	0.13	0.06	0.46	0.01
DOC	$[mgkg^{-1}]$	170	158.21±4.55	144.98	62.33	435.56	3526
DON	$[mgkg^{-1}]$	170	40.00±1.28	36.50	17.23	113.34	279
N_{mic}	$[mgkg^{-1}]$	170	141.08±4.09	132.50	42.31	357.83	2841
N_{min}	$[mgkg^{-1}]$	170	1.70±0.09	1.58	0.01	5.85	1
TOC	[%]	170	4.65±0.12	4.43	1.63	9.89	2
TON	[%]	170	0.33±0.01	0.31	0.10	0.82	0.01
bacterial 16S	$[gene\ copies]$	164	6.8e+08±4.5e+07	5.0e+08	5.1e+07	3.3e+09	1.6e+17
fungal 18S	$[gene\ copies]$	164	5.3e+06±3.5e+05	3.7e+06	6.0e+05	2.7e+07	2.1e+13
Metals							
As	$[mgkg^{-1}]$	168	9.40±0.26	9.60	0.50	27.70	11
Ba	$[mgkg^{-1}]$	168	385.35±11.52	344.50	201.80	1062.00	22569
Co	$[mgkg^{-1}]$	168	31.56±0.35	32.15	18.30	45.40	21
Cu.m	$[mgkg^{-1}]$	168	75.96±4.76	57.30	15.60	407.30	3852
Ni	$[mgkg^{-1}]$	168	39.55±0.69	38.65	20.30	80.10	82
Pb	$[mgkg^{-1}]$	168	172.33±13.37	106.60	18.50	1076.00	30391
Sb	$[mgkg^{-1}]$	168	1.81±0.32	0.70	0.40	39.10	17
V	$[mgkg^{-1}]$	168	79.78±1.13	77.75	44.10	117.90	217
Zn	$[mgkg^{-1}]$	168	268.77±13.83	215.80	58.90	999.90	32505
CLPP MicroResp							
Ala	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	170	3.78±0.08	3.77	0.77	6.32	1
Ara	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	170	4.21±0.09	4.30	1.21	6.33	1
Asc	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	170	9.25±0.11	9.64	1.98	11.30	2
Asp	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	170	3.95±0.08	3.98	1.37	7.02	1
Citr	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	170	10.45±0.13	11.06	3.18	12.08	3
Cys	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	170	2.56±0.08	2.40	0.62	5.67	1
Fruct	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	170	5.42±0.10	5.49	1.27	7.69	2
GABA	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	170	2.69±0.08	2.52	0.77	5.64	1
Gal	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	170	4.00±0.09	4.03	1.04	8.31	1
Glca	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	170	4.06±0.09	4.14	0.90	6.70	1
Gln	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	170	3.90±0.08	3.87	1.01	6.25	1
Gluc	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	170	5.67±0.09	5.72	1.72	8.06	1
H2O	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	170	2.06±0.08	1.78	0.61	7.32	1
KGA	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	170	10.71±0.11	11.22	3.23	12.19	2
Leu	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	170	2.63±0.08	2.53	0.68	5.89	1
MA	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	170	10.44±0.14	11.17	1.94	12.11	3
Prot	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	170	2.65±0.08	2.52	0.83	5.40	1
Van	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	170	2.63±0.08	2.52	0.96	8.26	1
Xyl	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	170	5.48±0.09	5.52	1.24	7.85	1
Tea bag decomposition							
green tea	$[\%\ decomposed]$	161	0.59±0.01	0.58	0.49	0.75	0.01
rooibos tea	$[\%\ decomposed]$	161	0.29±0.01	0.30	0.20	0.39	0.01

Table S 2.5 – Descriptive statistics of soil quality indicators in urban gardens of Zurich, CH. Data is aggregated by garden type.

		N	Mean±SE	Median	Min	Max	Variance
Physical indicators							
Allotment							
BD	$[gcm^{-3}]$	84	1.1±0.02	1.12	0.57	1.42	0.1
Clay	[%]	84	24.31±0.63	24.60	9.40	38.50	33
penetration resistance	$[MPa]$	84	1.23±0.06	1.15	0.39	2.55	0.1
PV	[%]	84	41.05±0.95	42.56	1.47	54.12	76
SA	[%]	84	79.38±1.28	81.90	46.64	94.61	137
Sand	[%]	84	41.3±1.11	40.23	13.75	71.95	104
Silt	[%]	84	34.39±0.62	34.50	18.65	49.15	32
WHC	[%]	84	82.84±1.45	80.45	66.03	145.99	177
Home							
BD	$[gcm^{-3}]$	86	1.06±0.02	1.05	0.80	1.45	0.1
Clay	[%]	86	23.29±0.58	22.50	10.90	39.25	29
penetration resistance	$[MPa]$	84	1.62±0.06	1.69	0.36	3.28	0.1
PV	[%]	86	40.68±0.61	40.39	28.83	54.96	32
SA	[%]	86	84.99±1	88.52	56.83	95.69	86
Sand	[%]	86	42.81±0.91	43.15	19.30	65.85	71
Silt	[%]	86	33.91±0.52	33.90	20.75	47.90	24
WHC	[%]	86	80.31±1.15	79.93	54.17	105.26	114
Chemical indicators							
Allotment							
B	$[mgkg^{-1}]$	84	1.59±0.07	1.51	0.19	3.88	0.1
Cu	$[mgkg^{-1}]$	84	39.14±4.14	27.41	7.40	209.10	1438
EC	$[\mu Scm^{-1}]$	84	175.47±4.77	167.50	82.20	354.00	1914
Fe	$[mgkg^{-1}]$	84	390.44±13.94	384.55	154.50	699.80	16335
K	$[mgkg^{-1}]$	84	191.9±14.41	157.67	47.34	831.34	17447
Mg	$[mgkg^{-1}]$	84	519.52±20.4	504.15	150.60	1125.00	34942
Mn	$[mgkg^{-1}]$	84	297.32±14.56	262.65	93.27	632.50	17819
P	$[mgkg^{-1}]$	84	229.92±12.92	214.66	21.35	460.44	14017
pH		84	7.2±0.03	7.25	6.45	7.56	0.1
Home							
B	$[mgkg^{-1}]$	86	1.16±0.07	0.96	0.14	3.71	0.1
Cu	$[mgkg^{-1}]$	86	25.62±1.75	22.49	3.04	93.00	263
EC	$[\mu Scm^{-1}]$	86	193.15±3.83	190.25	120.50	331.00	1262
Fe	$[mgkg^{-1}]$	86	348.64±9.31	348.85	190.10	583.00	7461
K	$[mgkg^{-1}]$	86	140.36±12.29	104.19	43.91	748.42	12990
Mg	$[mgkg^{-1}]$	86	513.39±16.9	500.70	143.60	1015.00	24560
Mn	$[mgkg^{-1}]$	86	295.18±8.58	275.40	151.60	479.80	6330
P	$[mgkg^{-1}]$	86	150.21±12.02	120.57	5.19	465.19	12435
pH		86	7.33±0.02	7.36	6.25	7.75	0.1

		N	Mean±SE	Median	Min	Max	Variance
Biological indicators							
Allotment							
basal respiration	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	84	0.24±0.01	0.22	0.10	0.72	0.1
C_{mic}	$[mgkg^{-1}]$	84	734.96±27.3	705.63	301.62	1548.41	62612
C_{min}	$[gCO_2 - Ckg^{-1}]$	84	0.16±0.01	0.15	0.07	0.46	0.1
DOC	$[mgkg^{-1}]$	84	164.81±7	149.67	79.63	435.56	4119
DON	$[mgkg^{-1}]$	84	39.6±1.84	35.23	19.62	109.55	285
N_{mic}	$[mgkg^{-1}]$	84	125.55±5.38	119.67	44.70	357.83	2434
N_{min}	$[mgkg^{-1}]$	84	1.77±0.14	1.55	0.00	5.56	2
TOC	$[\%]$	84	4.85±0.19	4.42	1.82	9.89	3
TON	$[\%]$	84	0.35±0.01	0.32	0.16	0.82	0.1
bacterial 16S	$[gene\ copies]$	81	6.3e+08±5.6e+07	5.0e+08	5.1e+07	2.5e+09	2.6e+17
fungal 18S	$[gene\ copies]$	80	5.5e+06± 4.4e+05	4.3e+06	7.0e+05	2.0e+07	1.7e+13
Home							
basal respiration	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	86	0.23±0.01	0.20	0.08	0.69	0.1
C_{mic}	$[mgkg^{-1}]$	86	879.88±29.78	835.92	279.91	1593.98	76252
C_{min}	$[gCO_2 - Ckg^{-1}]$	86	0.14±0.01	0.12	0.06	0.42	0.1
DOC	$[mgkg^{-1}]$	86	151.77±5.81	136.58	62.33	336.93	2903
DON	$[mgkg^{-1}]$	86	40.39±1.79	38.57	17.23	113.34	276
N_{mic}	$[mgkg^{-1}]$	86	156.25±5.71	153.82	42.31	305.35	2800
N_{min}	$[mgkg^{-1}]$	86	1.64±0.11	1.65	0.00	5.85	1
TOC	$[\%]$	86	4.45±0.15	4.45	1.63	8.94	2
TON	$[\%]$	86	0.31±0.01	0.31	0.10	0.61	0.1
bacterial 16S	$[gene\ copies]$	82	7.3e+08±7.1e+07	5.1e+08	8.3e+07	3.3e+09	4.3e+17
fungal 18S	$[gene\ copies]$	83	5.1e+06±5.4e+05	2.9e+06	6.0e+05	2.8e+07	2.5e+13
Metals							
Allotment							
As	$[mgkg^{-1}]$	82	9.41±0.4	9.45	2.60	27.70	14
Ba	$[mgkg^{-1}]$	82	383.3±17.34	330.75	230.70	1062.00	25270
Co	$[mgkg^{-1}]$	82	31.44±0.53	32.45	18.30	43.80	24
Cu.m	$[mgkg^{-1}]$	82	88.72±8.5	59.95	27.40	407.30	6063
Ni	$[mgkg^{-1}]$	82	40.23±1.08	39.35	22.10	80.10	98
Pb	$[mgkg^{-1}]$	82	143.79±16.43	88.15	34.00	1076.00	22672
Sb	$[mgkg^{-1}]$	82	1.68±0.5	0.60	0.40	39.10	21
V	$[mgkg^{-1}]$	82	81.68±1.63	79.05	50.60	117.90	224
Zn	$[mgkg^{-1}]$	82	270.97±19.39	215.80	102.00	966.50	31577
Home							
As	$[mgkg^{-1}]$	86	9.39±0.33	9.85	0.50	19.40	9
Ba	$[mgkg^{-1}]$	86	387.3±15.35	352.20	201.80	841.20	20253
Co	$[mgkg^{-1}]$	86	31.68±0.46	31.90	21.40	45.40	19
Cu.m	$[mgkg^{-1}]$	86	63.79±4.15	51.70	15.60	208.80	1483
Ni	$[mgkg^{-1}]$	86	38.9±0.88	38.00	20.30	65.30	66
Pb	$[mgkg^{-1}]$	86	199.55±20.62	117.50	18.50	919.20	36568
Sb	$[mgkg^{-1}]$	86	1.93±0.4	1.00	0.40	33.00	14
V	$[mgkg^{-1}]$	86	77.97±1.55	77.15	44.10	112.20	207
Zn	$[mgkg^{-1}]$	86	266.67±19.81	215.85	58.90	999.90	33762

		N	Mean±SE	Median	Min	Max	Variance
CLPP MicroResp							
Allotment							
Ala	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	84	3.53±0.12	3.54	0.77	6.32	1
Ara	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	84	3.88±0.12	3.97	1.21	6.08	1
Asc	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	84	9.07±0.17	9.22	1.98	11.30	2
Asp	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	84	3.70±0.12	3.79	1.37	7.02	1
Citr	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	84	10.29±0.2	11.01	3.18	12.08	3
Cys	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	84	2.42±0.11	2.30	0.62	5.67	1
Fruct	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	84	5.07±0.15	5.15	1.27	7.62	2
GABA	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	84	2.45±0.09	2.41	0.77	5.14	1
Gal	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	84	3.68±0.13	3.67	1.04	8.31	1
Glca	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	84	3.57±0.14	3.79	0.90	6.70	2
Gln	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	84	3.68±0.11	3.67	1.01	5.84	1
Gluc	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	84	5.32±0.14	5.44	1.72	8.01	2
H2O	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	84	1.87±0.11	1.56	0.61	7.32	1
KGA	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	84	10.69±0.17	11.14	3.23	11.96	2
Leu	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	84	2.44±0.10	2.46	0.68	5.89	1
MA	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	84	10.35±0.21	11.08	1.94	12.11	4
Prot	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	84	2.47±0.10	2.37	0.83	5.10	1
Van	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	84	2.46±0.12	2.23	0.96	8.26	1
Xyl	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	84	5.18±0.14	5.43	1.24	7.38	2
Home							
Ala	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	86	4.03±0.11	4.06	1.77	6.05	1
Ara	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	86	4.54±0.11	4.63	1.96	6.33	1
Asc	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	86	9.43±0.15	9.72	4.55	11.12	2
Asp	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	86	4.20±0.10	4.32	2.17	6.53	1
Citr	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	86	10.61±0.15	11.10	6.61	12.03	2
Cys	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	86	2.69±0.12	2.48	0.85	5.59	1
Fruct	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	86	5.76±0.11	5.79	3.12	7.69	1
GABA	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	86	2.93±0.12	2.77	1.15	5.64	1
Gal	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	86	4.31±0.11	4.31	2.25	6.67	1
Glca	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	86	4.54±0.10	4.66	1.77	6.20	1
Gln	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	86	4.12±0.11	4.07	1.72	6.25	1
Gluc	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	86	6.00±0.11	6.11	3.24	8.06	1
H2O	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	86	2.25±0.12	1.99	0.67	4.85	1
KGA	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	86	10.72±0.14	11.25	6.81	12.19	2
Leu	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	86	2.82±0.11	2.76	1.05	5.28	1
MA	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	86	10.53±0.18	11.25	4.21	12.06	3
Prot	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	86	2.83±0.11	2.76	1.04	5.40	1
Van	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	86	2.79±0.11	2.60	1.10	5.25	1
Xyl	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	86	5.76±0.10	5.75	3.65	7.85	1
Tea bag decomposition							
Home							
green tea	$[\%\ decomposed]$	84	0.58±0.01	0.58	0.49	0.69	0.01
rooibos tea	$[\%\ decomposed]$	84	0.29±0.01	0.29	0.20	0.35	0.01
Home							
green tea	$[\%\ decomposed]$	77	0.60±0.01	0.59	0.53	0.75	0.01
rooibos tea	$[\%\ decomposed]$	77	0.30±0.01	0.30	0.22	0.39	0.01

Table S 2.6 – Descriptive statistics of soil quality indicators in urban gardens of Zurich, CH. Data is aggregated by garden habitat type.

		N	Mean±SE	Median	Min	Max	Variance
Physical indicators							
Vegetables							
BD	$[gcm^{-3}]$	47	1.15±0.02	1.16	0.80	1.42	0.1
Clay	[%]	47	24.55±0.69	25.10	16.65	37.65	22
penetration resistance	$[MPa]$	47	0.96±0.06	0.90	0.39	1.95	0.1
PV	[%]	47	42.68±1.46	44.27	1.47	54.12	100
SA	[%]	47	76.22±1.71	79.28	46.64	92.77	138
Sand	[%]	47	40.9±1.14	41.10	27.25	56.20	62
Silt	[%]	47	34.55±0.7	34.50	26.10	45.25	23
WHC	[%]	47	82.4±1.81	79.98	66.03	140.14	154
Flowers & Berries							
BD	$[gcm^{-3}]$	52	1.12±0.02	1.13	0.57	1.45	0
Clay	[%]	52	22.55±0.75	21.60	10.90	38.85	30
penetration resistance	$[MPa]$	51	1.47±0.08	1.29	0.36	2.72	0
PV	[%]	52	42.94±0.88	43.98	24.28	54.96	40
SA	[%]	52	81.08±1.48	83.67	56.83	95.69	114
Sand	[%]	52	43.97±1.24	44.58	19.30	65.85	79
Silt	[%]	52	33.49±0.73	33.05	23.25	49.15	28
WHC	[%]	52	80.33±1.77	77.83	54.17	139.16	162
Lawn							
BD	$[gcm^{-3}]$	71	1±0.02	0.98	0.78	1.32	0
Clay	[%]	71	24.2±0.72	23.70	9.40	39.25	37
penetration resistance	$[MPa]$	70	1.7±0.06	1.72	0.58	3.28	0
PV	[%]	71	38.14±0.55	38.14	28.83	49.86	21
SA	[%]	71	87.03±0.95	90.00	61.70	95.28	64
Sand	[%]	71	41.43±1.23	39.80	13.75	71.95	108
Silt	[%]	71	34.36±0.66	34.55	18.65	47.75	31
WHC	[%]	71	81.9±1.36	81.08	54.20	145.99	131
Chemical indicators							
Vegetables							
B	$[mgkg^{-1}]$	47	1.87±0.09	1.78	0.45	3.71	0.1
Cu	$[mgkg^{-1}]$	47	39.04±5.2	27.87	8.86	209.10	1269
EC	$[\mu Scm^{-1}]$	47	174.14±6.33	166.00	82.20	283.00	1880
Fe	$[mgkg^{-1}]$	47	400.52±18.53	386.80	157.90	678.20	16132
K	$[mgkg^{-1}]$	47	231.15±20.97	209.30	51.83	748.42	20665
Mg	$[mgkg^{-1}]$	47	568.94±27.88	561.50	150.60	1125.00	36520
Mn	$[mgkg^{-1}]$	47	296.2±17.41	264.20	93.27	602.40	14245
P	$[mgkg^{-1}]$	47	273.22±16.98	244.12	63.36	465.19	13555
pH		47	7.25±0.03	7.26	6.45	7.69	0.1
Flowers & Berries							
B	$[mgkg^{-1}]$	52	1.34±0.09	1.23	0.19	3.70	0.1
Cu	$[mgkg^{-1}]$	52	31.06±3.73	23.37	7.40	177.10	723
EC	$[\mu Scm^{-1}]$	52	187.84±6.44	178.20	124.20	354.00	2159
Fe	$[mgkg^{-1}]$	52	364.86±13.81	359.90	210.70	600.10	9922
K	$[mgkg^{-1}]$	52	155.61±15.34	118.75	47.34	691.92	12240
Mg	$[mgkg^{-1}]$	52	500.59±19.65	486.35	188.70	868.70	20074
Mn	$[mgkg^{-1}]$	52	282.73±12.23	258.20	124.40	556.00	7773
P	$[mgkg^{-1}]$	52	178.95±15.51	150.31	10.17	418.20	12506
pH		52	7.32±0.03	7.36	6.52	7.75	0.1
Lawn							
B	$[mgkg^{-1}]$	71	1.06±0.07	0.99	0.14	3.88	0.1
Cu	$[mgkg^{-1}]$	71	28.75±3.21	20.65	3.04	138.80	731
EC	$[\mu Scm^{-1}]$	71	188.7±3.91	187.80	117.40	276.00	1085
Fe	$[mgkg^{-1}]$	71	351.87±12.26	335.90	154.50	699.80	10680
K	$[mgkg^{-1}]$	71	130.06±12.64	107.40	43.91	831.34	11336
Mg	$[mgkg^{-1}]$	71	493.25±20.58	504.40	143.60	1080.00	30070
Mn	$[mgkg^{-1}]$	71	306.16±13.82	270.10	118.50	632.50	13558
P	$[mgkg^{-1}]$	71	142.04±12.15	116.16	5.19	427.74	10483
pH		71	7.24±0.03	7.29	6.25	7.72	0.1

		N	Mean±SE	Median	Min	Max	Variance
Biological indicators							
Vegetables							
basal respiration	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	47	0.26±0.02	0.22	0.10	0.69	0.1
C_{mic}	$[mgkg^{-1}]$	47	687.02±35.92	639.76	301.62	1362.47	60630
C_{min}	$[gCO_2 - Ckg^{-1}]$	47	0.18±0.01	0.14	0.07	0.42	0.1
DOC	$[mgkg^{-1}]$	47	173.21±8.34	155.00	79.63	336.93	3270
DON	$[mgkg^{-1}]$	47	39.54±2.46	36.08	19.92	109.55	284
N_{mic}	$[mgkg^{-1}]$	47	114.39±6.31	101.25	44.70	208.64	1872
N_{min}	$[mgkg^{-1}]$	47	1.61±0.15	1.55	0.00	4.10	1
TOC	[%]	47	5.09±0.25	4.61	1.82	9.68	3
TON	[%]	47	0.36±0.02	0.34	0.16	0.71	0.1
bacterial 16S	$[gene\ copies]$	45	5.3e+08±7.8e+07	3.7e+08	5.1e+07	2.7e+09	2.8e+17
fungal 18S	$[gene\ copies]$	45	6.4e+06±7.3e+05	4.6e+06	8.3e+05	2.1e+07	2.5e+13
Flowers & Berries							
basal respiration	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	52	0.24±0.02	0.21	0.09	0.72	0.1
C_{mic}	$[mgkg^{-1}]$	52	790.84±36.5	781.81	279.91	1462.61	69262
C_{min}	$[gCO_2 - Ckg^{-1}]$	52	0.15±0.01	0.13	0.06	0.46	0.1
DOC	$[mgkg^{-1}]$	52	160.2±8.94	145.94	62.33	415.26	4155
DON	$[mgkg^{-1}]$	52	41.1±2.74	35.27	18.95	113.34	390
N_{mic}	$[mgkg^{-1}]$	52	139.22±7.09	130.49	42.31	297.24	2615
N_{min}	$[mgkg^{-1}]$	52	1.78±0.17	1.71	0.00	5.34	2
TOC	[%]	52	4.63±0.22	4.62	1.63	9.53	2
TON	[%]	52	0.32±0.02	0.31	0.13	0.78	0.1
bacterial 16S	$[gene\ copies]$	50	7.2e+08±8.4e+07	5.9e+08	8.8e+07	3.3e+09	3.7e+17
fungal 18S	$[gene\ copies]$	50	5.5e+06±7.0e+05	3.9e+06	8.0e+05	2.8e+07	2.5e+13
Lawn							
basal respiration	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	71	0.22±0.01	0.20	0.08	0.39	0.1
C_{mic}	$[mgkg^{-1}]$	71	901.3±31.49	822.41	459.93	1593.98	70400
C_{min}	$[gCO_2 - Ckg^{-1}]$	71	0.14±0.01	0.13	0.07	0.25	0.1
DOC	$[mgkg^{-1}]$	71	146.83±6.55	136.44	85.03	435.56	3051
DON	$[mgkg^{-1}]$	71	39.5±1.68	36.60	17.23	95.54	201
N_{mic}	$[mgkg^{-1}]$	71	160.11±6.36	154.56	80.47	357.83	2874
N_{min}	$[mgkg^{-1}]$	71	1.7±0.14	1.58	0.00	5.85	1
TOC	[%]	71	4.37±0.18	4.22	2.24	9.89	2
TON	[%]	71	0.32±0.01	0.31	0.10	0.82	0.1
bacterial 16S	$[gene\ copies]$	68	7.4e+08±7.2e+07	5.2e+08	8.3e+07	2.5e+09	3.7e+17
fungal 18S	$[gene\ copies]$	68	4.4e+06±4.3e+05	3.0e+0	6.0e+05	1.8e+07	1.3e+13
Metals							
Vegetables							
As	$[mgkg^{-1}]$	47	9.2±0.56	8.90	3.20	27.70	14
Ba	$[mgkg^{-1}]$	47	393.37±20.39	366.90	275.20	1014.00	19546
Co	$[mgkg^{-1}]$	47	31.35±0.58	31.50	21.20	38.50	16
Cu.m	$[mgkg^{-1}]$	47	89.36±10.73	62.60	28.70	407.30	5408
Ni	$[mgkg^{-1}]$	47	40.5±1.21	39.30	22.50	58.30	69
Pb	$[mgkg^{-1}]$	47	157.45±17.81	120.10	39.30	528.70	14903
Sb	$[mgkg^{-1}]$	47	1.31±0.22	0.60	0.40	9.80	2
V	$[mgkg^{-1}]$	47	80.32±1.75	80.60	53.50	105.60	145
Zn	$[mgkg^{-1}]$	47	283.33±24.6	236.60	114.10	966.50	28434
Flowers & Berries							
As	$[mgkg^{-1}]$	51	9.01±0.51	8.60	0.50	21.00	13
Ba	$[mgkg^{-1}]$	51	390.85±21.38	346.30	230.70	841.20	23764
Co	$[mgkg^{-1}]$	51	31.62±0.71	32.60	21.60	45.40	27
Cu.m	$[mgkg^{-1}]$	51	73.25±7.57	61.00	26.00	339.70	2981
Ni	$[mgkg^{-1}]$	51	37.15±1.17	34.80	24.60	58.00	71
Pb	$[mgkg^{-1}]$	51	207.27±26.27	119.80	40.50	709.80	35895
Sb	$[mgkg^{-1}]$	51	2.17±0.66	0.70	0.40	33.00	22
V	$[mgkg^{-1}]$	51	75.29±2.06	73.40	50.60	114.90	221
Zn	$[mgkg^{-1}]$	51	298.07±27.35	225.30	103.90	999.90	38896
Lawn							
As	$[mgkg^{-1}]$	70	9.82±0.33	10.30	2.60	16.80	8
Ba	$[mgkg^{-1}]$	70	375.95±18.47	331.80	201.80	1062.00	24217
Co	$[mgkg^{-1}]$	70	31.66±0.55	32.35	18.30	43.80	21
Cu.m	$[mgkg^{-1}]$	70	68.93±6.9	50.20	15.60	336.20	3379
Ni	$[mgkg^{-1}]$	70	40.66±1.15	40.15	20.30	80.10	94
Pb	$[mgkg^{-1}]$	70	156.87±22.62	80.65	18.50	1076.00	36312
Sb	$[mgkg^{-1}]$	70	1.88±0.57	0.70	0.40	39.10	23
V	$[mgkg^{-1}]$	70	82.68±1.86	79.95	44.10	117.90	246
Zn	$[mgkg^{-1}]$	70	237.65±20.48	188.50	58.90	869.40	29767

		N	Mean±SE	Median	Min	Max	Variance
CLPP MicroResp							
Vegetables							
Ala	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	47	3.4±0.17	3.42	0.77	5.96	1
Ara	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	47	3.56±0.17	3.70	1.21	6.26	1
Asc	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	47	9.03±0.24	9.24	1.98	11.12	3
Asp	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	47	3.57±0.18	3.63	1.37	7.02	1
Citr	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	47	10.52±0.26	11.10	3.18	11.94	3
Cys	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	47	2.25±0.17	1.92	0.62	5.25	1
Fruct	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	47	4.66±0.2	4.62	1.27	7.58	2
GABA	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	47	2.42±0.15	2.31	0.77	5.64	1
Gal	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	47	3.41±0.19	3.48	1.04	8.31	2
Glca	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	47	3.36±0.16	3.39	1.41	5.62	1
Gln	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	47	3.48±0.16	3.57	1.01	6.15	1
Gluc	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	47	4.98±0.2	5.13	1.72	7.94	2
H2O	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	47	1.9±0.17	1.56	0.67	7.32	1
KGA	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	47	10.68±0.22	11.14	3.26	12.19	2
Leu	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	47	2.37±0.15	2.10	0.68	5.89	1
MA	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	47	10.48±0.27	11.21	1.94	12.06	4
Prot	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	47	2.38±0.14	2.16	0.83	4.70	1
Van	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	47	2.39±0.18	2.18	0.96	8.26	1
Xyl	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	47	4.83±0.2	4.78	1.24	7.85	2
Flowers & Berries							
Ala	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	52	3.88±0.14	3.77	1.58	6.05	1
Ara	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	52	4.25±0.15	4.45	1.36	6.12	1
Asc	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	52	9.69±0.17	10.04	4.55	11.30	2
Asp	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	52	4.07±0.13	4.15	2.17	6.53	1
Citr	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	52	10.74±0.19	11.24	6.19	12.08	2
Cys	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	52	2.57±0.14	2.30	0.95	4.62	1
Fruct	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	52	5.6±0.17	5.66	3.12	7.69	1
GABA	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	52	2.76±0.14	2.58	0.96	5.07	1
Gal	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	52	4.08±0.15	4.11	1.62	5.91	1
Glca	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	52	4.34±0.14	4.59	1.25	6.20	1
Gln	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	52	4.07±0.15	4.01	1.72	6.25	1
Gluc	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	52	5.82±0.16	5.82	3.24	7.88	1
H2O	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	52	2.05±0.13	1.80	0.61	4.80	1
KGA	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	52	10.93±0.16	11.31	6.81	12.02	1
Leu	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	52	2.66±0.14	2.52	0.75	5.10	1
MA	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	52	10.79±0.21	11.40	4.21	12.11	2
Prot	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	52	2.68±0.14	2.36	0.97	5.40	1
Van	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	52	2.65±0.12	2.60	1.07	4.62	1
Xyl	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	52	5.61±0.13	5.53	3.68	7.38	1
Lawn							
Ala	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	71	3.96±0.11	4.16	1.95	6.32	1
Ara	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	71	4.62±0.11	4.65	2.41	6.33	1
Asc	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	71	9.08±0.17	9.37	4.82	11.12	2
Asp	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	71	4.13±0.11	4.28	1.95	6.20	1
Citr	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	71	10.2±0.2	10.69	3.54	12.03	3
Cys	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	71	2.76±0.13	2.66	0.85	5.67	1
Fruct	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	71	5.8±0.11	5.74	2.29	7.56	1
GABA	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	71	2.82±0.11	2.69	1.29	5.27	1
Gal	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	71	4.33±0.11	4.21	2.46	6.67	1
Glca	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	71	4.31±0.14	4.25	0.90	6.70	1
Gln	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	71	4.05±0.11	4.05	2.47	5.84	1
Gluc	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	71	6.01±0.12	6.10	3.84	8.06	1
H2O	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	71	2.18±0.13	1.92	0.62	4.85	1
KGA	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	71	10.56±0.18	11.01	3.23	12.10	2
Leu	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	71	2.79±0.11	2.80	1.16	5.28	1
MA	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	71	10.16±0.23	10.81	2.90	11.99	4
Prot	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	71	2.82±0.11	2.80	1.04	5.10	1
Van	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	71	2.77±0.12	2.55	1.10	6.06	1
Xyl	$[\mu gCO_2 - Cg^{-1}h^{-1}]$	71	5.81±0.1	5.82	3.44	7.80	1
Tea bag decomposition							
Vegetables							
green tea	$[\%\ decomposed]$	45	0.57±0.01	0.57	0.49	0.71	0.01
rooibos tea	$[\%\ decomposed]$	45	0.29±0.01	0.30	0.20	0.38	0.01
Flowers & Berries							
green tea	$[\%\ decomposed]$	49	0.58±0.01	0.58	0.52	0.70	0.01
rooibos tea	$[\%\ decomposed]$	49	0.30±0.01	0.30	0.22	0.38	0.01
Lawn							
green tea	$[\%\ decomposed]$	67	0.60±0.01	0.59	0.53	0.75	0.01
rooibos tea	$[\%\ decomposed]$	67	0.29±0.01	0.29	0.23	0.39	0.01

3

A gardener's influence on urban soil quality

Text mining: Most frequent 100 words from chapter 3 (shape: earthworm), created with the "wordcloud2" R package (Lang 2016).

A gardener's influence on urban soil quality

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Keywords: urban gardening, soil quality indicators, C_{mic}/SOC, qCO_2, urban ecology theory, urban ecosystem services, earthworms, tea bag index

Frontiers in Environmental Science
ORIGINAL RESEARCH
Published: 08 May 2018
DOI: 10.3389/fenvs.2018.00025

Abstract

Gardens are hot spots for urban biodiversity and provide habitats for many plant and animal species, both above- and below-ground. Furthermore, gardens provide a wide range of ecosystem services, including carbon (C) storage and nutrient cycling. Although the soil is the foundation of sustainable gardens providing those ecosystem services, very little is known about the consequences of garden management on soil quality. Here we present a comprehensive assessment of urban garden soil quality, including biotic and abiotic site characteristics combined with land-use history and garden management information in a multivariate evaluation. A set of 44 soil quality indicators was measured at 170 sites of 85 gardens in the city of Zurich, Switzerland, comprising contrastingly managed garden habitats along a gradient of urban density. Taken together, our results show that garden management was the driving factor that influenced soil quality and soil functions. Eco-physiological soil quality indices were useful to identify differences in disturbance and intensity of soil use, showing highest microbial (microbial biomass (C_{mic})/soil organic carbon (SOC)) and lowest metabolic (qCO_2) quotients in perennial grass sites compared to annual vegetable sites. Despite the intensity of soil disturbance in annual vegetable and flower beds, the highest endogeic earthworm biomass and diversity were found in those habitats. Whereas decomposition of green tea bags was higher in grass sites. Soil heavy metal contents varied considerably and could not be linked with garden management practices, but with spatial patterns of industry and traffic. We conclude that understanding soil quality in urban ecosystems needs multi-indicator frameworks to capture the complexity of soil characteristics and the influencing factors in space and time. This study contributes to a better understanding of urban gardens and enhances the development of sustainable soil management strategies aimed at long-term improvement of soil quality and related ecosystem services in cities.

3.1 Introduction

Urban gardens are hot spots of urban biodiversity, serving as fertile islands for plants and animals in increasingly densified cities (e.g. Gilbert 1989, Owen 2010). Besides their importance as ecological niches for many species (Smith et al. 2006), they increase the connectivity of urban landscapes (Rudd et al. 2002) and provide multiple ecosystem services (Elmqvist et al. 2015). Urban gardens give the opportunity for people, particularly children (Hand et al. 2017), to interact with nature (Miller 2005). Soil is the foundation of urban habitats that provide key ecosystem services in cities (Zhu et al. 2018). Urban garden soils are important for regulating the microclimate by providing shade and allowing water to infiltrate and evaporate (Bowler et al. 2010). Moreover, they improve air quality (Janhäll 2015), prevent flooding by reducing surface-water run-off (Bolund and Hunhammar 1999), storing a considerable amount of soil organic carbon (SOC) (Edmondson et al. 2012) and improve pollination by hosting diverse insect species (Samnegård et al. 2011). But in many cities the sealed area is expanding tremendously with negative consequences for these ecosystem services (Sachs 2015), especially for contested urban green spaces like allotment gardens (Tappert et al. 2018), due to the need for accommodation and infrastructure of growing urban populations.

Urban soils are often associated with degraded and possibly polluted soils (Meuser 2010), low in SOC (e.g. Bradley et al. 2005, Craul 1999) and biological

activity (e.g. Lorenz and Kandeler 2005, Scharenbroch et al. 2005), compared to non-urban soils in forests or croplands. Urban soil properties may be altered by anthropogenic disturbances such as compaction due to construction activities or various soil management practices such as fertilization, mowing or drainage (Lorenz 2017). For this reason disturbed urban soils were generally considered to have a low physical and chemical quality, not suitable for crop production (Jim 1998). More recently urban soils are looked at from a different angle with the increasing interest in urban agriculture. Urban garden soils can be fertile and can support soil functions despite intensive soil use (Levin et al. 2017). For instance, Edmondson et al. (2014) and Vasenev et al. (2013) found increased values in urban gardens compared to non-urban soils.

Soil quality can be defined as 'the capacity of a soil to function within ecosystem and land-use boundaries, to sustain biological productivity, maintain environmental quality, and promote plant and animal health', including human health (Doran and Parkin 1994). This comprehensive but complex definition refers to the multi-functionality of soils providing ecosystem services but also to the site specificity of the soil properties (Bünemann et al. 2018). Urban gardens not only provide vegetables and fruits, they are also important to conserve and provide niches for above- and below-ground biodiversity. A definition of soil fertility is given in the Swiss ordinance on impacts on soils (Swiss Federal Council 1998) comprising among others the ability of harbouring a biologically active community, a typical site-specific soil structure, an undisturbed decomposition and no risk for humans and animals, when they take it up directly. In addition to 44 measures of soil quality (MSQ), we analysed the decomposability of tea bags and the suitability of habitats for earthworms as important soil functions. Moreover, the distribution and contents of heavy metals have been evaluated, since they are a major concern in urban soils (Kim et al. 2014). Earthworms are sensitive to both soil management (Pulleman et al. 2012) and soil pollution (Pérès et al. 2011) and functional groups of earthworms have distinct impact on soil functions such as soil structure and decomposition (Edwards 2004). Only few studies have investigated soil quality or soil functions in urban gardens (Beniston et al. 2016), which is probably because of the difficulties associated with gaining access to private properties (Goddard et al. 2010). Findings that gardening activities, such as regular clearing, digging, planting, weeding and watering the soil, can have a significant impact on above-ground biodiversity (Goddard et al. 2013, Smith et al. 2006), suggest that garden management will also affect the quality of urban garden soils. However, little is known how these management practices affect below-ground diversity and soil quality (e.g. Amossé et al. 2016, Edmondson et al. 2014).

In relation to the central principle in urban ecology theory, that anthropogenic management controls ecosystem processes (e.g. Alberti 1999), we determined the impact of gardening practices on soil properties and soil functions. A multivariate approach combining management and garden characteristics with a comprehensive set of MSQ (Table 3.1) was conducted on a city wide sampling approach including the two most common urban garden types (allotment and home gardens). Effects of soil disturbance were further studied by the use of ecophysiological soil indices introduced by Anderson and Domsch (1989), who used them to show the influence of management intensity on soil microbial communities. The authors found increased values for the metabolic quotient (qCO_2) and decreased for the microbial quotient (microbial biomass C (C_{mic})/SOC) in soils of monocropping systems compared to those under crop rotation. In this paper, we investigate the following four research hypothesis:

1. Garden management practices influence soil properties.

2. Biological soil quality measures, which are highly sensitive to management in agricultural soils (Mäder et al. 2002), are strongly affected by garden management practices.

3. Sites with more frequent soil disturbance have a lower microbial (C_{mic}/SOC) and a higher metabolic (qCO_2) quotient.

4. Heavy metal contents have a negative impact on decomposition and earthworm abundance.

This study was part of an interdisciplinary project (www.bettergardens.ch) that focuses on the importance of urban gardens for biodiversity, ecosystem services and human well-being. Our aim was to contribute to a key question in urban ecology (McPhearson et al. 2016): What is the impact of disturbance on soil functions? We propose an adapted multivariate method for assessing soil quality, which is urgently needed to better understand the conditions of urban soils and the ecosystem services they provide (Zornoza et al. 2015).

3.2 Material and Methods

3.2.1 Study sites and design

The study took place in the city of Zurich (Switzerland; 47°22'0" N, 8°33'0" E), which is an average size European city with approximately 0.4 million inhabitants. Zurich is located in the temperate climate zone with mean annual temperature of 9.3°C (1981-2010) and mean annual rainfall of 1134 mm (MeteoSwiss 2017). The total area of Zurich is approximately 8800 ha with 47% settlement area including buildings and gardens, 26% forest, 15% roads, 10% agriculture and 2% water bodies (Statistical office Zurich 2017). Allotment gardens, a plot of land rented by citizens interested in gardening, have been installed in Zurich since the beginning of the 20th century and follow a long history of self-supplying citizen gardens, which goes back to the 16th century (Christl et al. 2004). The 5500 allotment gardens cover 3% of total settlement area and 3% of urban green space in the city of Zurich, while home gardens, which are located around a private house, cover 11% of total settlement area and 25% of urban green space (Grün Stadt Zurich 2010). The city wide soil quality assessment was done on 85 urban gardens (42 allotment and 43 home gardens), which were selected following a systematic nested design (Fortin et al. 1989). Allotment and home gardens were selected across two independent gradients: An urbanization density gradient and a gradient of garden management. The urbanization density gradient was assessed by the geographical position and the information about the built and sealed surface area around a garden. The gradient of garden management was visually assessed on-site by a professional gardener, ranging from intensively managed gardens consisting of vegetable plots to very natural gardens with flower meadows. Nested in each garden two study sites were selected to account for varying management concepts within a garden and to test for the effect of similar parental soil conditions. We focused on medium sized gardens with an average size of 312 $\pm$ 154 m^2, to have a similar garden area in allotment and home gardens and because these small gardens account for the largest share of the total garden area in European cities (Loram et al. 2007). Gardens were first selected by aerial photographs (ArcGIS) and secondly by visual on-site inspection, before asking each garden owner for permission. Given the spatial and temporal complexity of urban environments we decided to focus on one city in order to consider all possible spatial components affecting urban soil quality in gardens and increase the statistical power of the nested design.

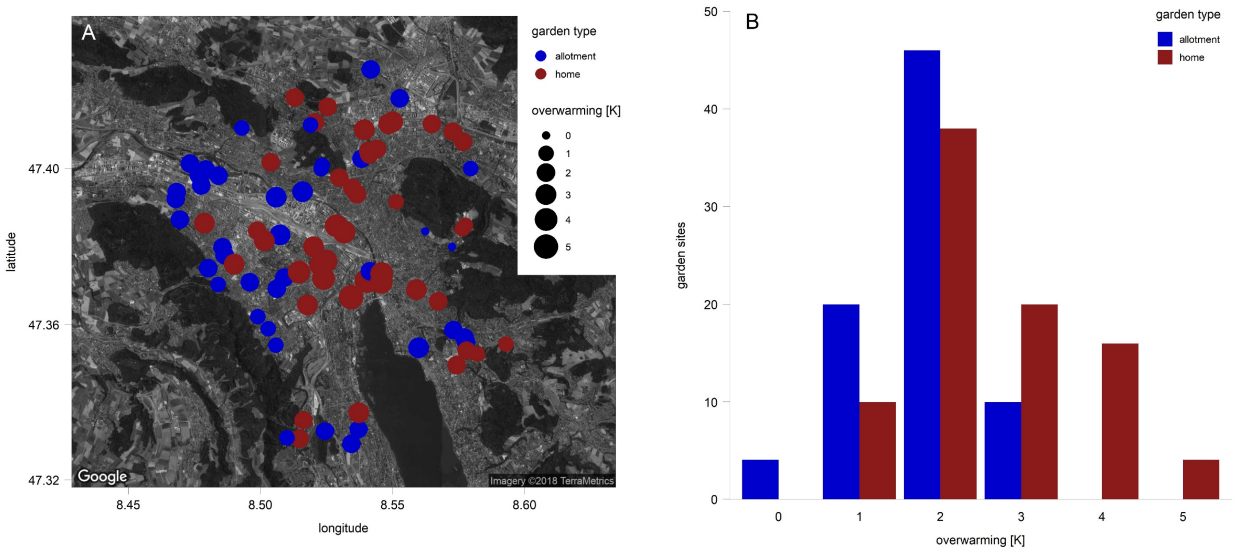


Figure 3.1 – A) Urban gardens analysed in the city of Zurich (85 urban gardens × 2 sampling sites per garden). Colours correspond to the garden types (blue: allotment, red: home) and the point size to the urbanization density gradient, which is represented by a regional climate model with local deviation of mean night temperatures near surface from 0 to + 6 K (Parlow et al. 2010). B) Values of local mean night temperature deviations (overwarming), representing the urbanization density gradient in this study.

3.2.2 Soil samples

Soil samples were taken from two distinct sites within each garden. One site was chosen in a more frequently disturbed garden area, usually cultivated with annual plants (e.g. vegetables) and the other in an area with less soil disturbance, such as sites covered with perennial vegetation (e.g. lawn). Sampling under the canopy of trees was avoided to minimize undesirable confounding effects on soil properties. Soil samples were taken in March 2015 before the beginning of the gardening season (i.e. before major soil disturbances such as tillage, fertilization or planting). At each of the 170 sites, within an area 2 m x 2 m, five soil samples were taken randomly from the 0-20 cm soil layer with a 30 mm wide soil auger. The soil samples were pooled per site and on arrival to the laboratory they were gently air dried prior to homogenisation with an analytical sieve (2 mm mesh size), removing visible roots, other plant material and stones. The samples were stored at 4°C until further analysis. For the soil physical measurements, three soil cores (5 cm diameter & depth, Eijkelkamp, NL) were taken at a depth of 10-15 cm at each site.

3.2.3 Measures of soil quality (MSQ)

A total of 44 measurements were obtained on all 170 sites (Table 3.1), according to Swiss standard methods for physical, chemical and biological soil characterization (Agroscope 2012), if not stated otherwise. Soil water content was determined gravimetrically, after drying soil at 105°C for 24 h. The pH and electrical conductivity (EC) of air dried sub samples were measured in a soil suspension with deionized water (1:2.5, w/v). The maximum soil water holding capacity (WHC) was determined by a cylinder method, where field moist soil is saturated with water on a sand bath (Schinner et al. 1996). Soil aggregate stability (SA) was measured s the proportion of stable aggregates (1-2 mm) left after 5 min. of wet sieving on a multi-sieve device with a frequency of 42 cycles min^{-1} subtracting the sand fraction (Schinner et al. 1996). Particle size distribution for clay, silt and sand contents was measured by a combined sieving and sedimentation technique, while soil texture was classified according to USDA taxonomy. Pore space volume and soil bulk density (BD) were determined with the undisturbed soil cores, where latter was determined as the soil mass dried at 105°C related to the total volume of the cylinders. Penetration resistance was measured using a Penetrologger (Eijkelkamp, NL) with a cone type of 10^{-4} m^2 and a penetration speed of 0.02 m sec^{-1} down to a maximum soil depth of 80 cm, recording the penetration resistance every 1 cm. Ten replicate measurements were taken for each 2 m x 2 m area and mean values for penetration resistance were taken from 0-20 cm. Soil nutrient contents (P, K, Mg, Cu, Fe, Mn, B) were measured externally at a certified laboratory with ammonium acetate-EDTA. SOC and total organic nitrogen (TON) were analysed by a CHN analyser (Thermo Scientific Flash EA 1112, NL) after removing carbonates by acidifying the soil with HCl (2 M). Dissolved organic carbon (DOC), dissolved organic nitrogen (DON) and mineral nitrogen (N_{min}) were measured in an extract with 0.01 M $CaCl_2$ (1:4, w/v). Mineral nitrogen (N_{min}) (nitrate and ammonium) was analysed spectrometrically (SAN-plus Segmented Flow Analyser, Skalar Analytical, NL) and DOC and DON with a TOC/TNb analyser (multi N/C 2100 S, Analytic Jena AG, D) described in Krauss et al. (2017). Soil biological analyses were conducted at 40-50% maximum WHC. Soil microbial biomass carbon (C_{mic}) and nitrogen (N_{mic}) contents were estimated by chloroform-fumigation-extraction (CFE) according to Vance et al. (1987), with triplicates of 20 g soil subsamples following Fliessbach et al. (2007). Soil respiration was measured with 30 g of soil on a gas chromatograph (7890A, Agilent Technologies, USA) equipped with a thermal conductivity detector (TCD; details are provided in Table S 3.12). Basal respiration (resp) rates were recorded after one week and C mineralisation (C_{min}) as cumulative values after four weeks of soil incubation at 20°C.

Soil organic matter characterisation

Soil organic matter (SOM) was characterised using diffuse reflectance Fourier transform mid-infrared spectroscopy (DRIFTS) following Rasche et al. (2013). Measurement details are given in Table S 3.13. Out of 19 calculated DRIFTS peaks two labile peaks (A & B) and two stable peaks (C & D) of organic functional groups were inspected in detail by consulting the DRIFTS fingerprint database and the literature on stable or labile SOM compounds. Peak A ranging from 1080 to 950 cm^{-1}, was characterised as a labile compound of SOM linked to polysaccharide-C (Lehmann et al. 2007)Ṗeak B (1148 to 1170 cm^{-1}) was linked to labile poly-alcoholic and ether functional groups (Demyan et al. 2012, Spaccini and Piccolo 2007). Peak C (1660 to 1580 cm^{-1}) was associated with aromatic compounds (Baes and Bloom 1989, Demyan et al. 2012) representing stable functional compounds of SOM. Peak D (3010 to 2800 cm^{-1}) was related to aliphatic compounds (Baes and Bloom 1989, Lehmann et al. 2007), representing a relatively labile SOM fraction (Mirzaeitalarposhti

et al. 2016), although it can also be correlated with SOC contents (Gerzabek et al. 2006) due to carbonate interference(Mirzaeitalarposhti et al. 2016) and is therefore suspected to represent rather stable functional organic groups of SOM in this study.

Table 3.1 – Measures of soil quality (MSQ) (N=44) obtained in all urban garden sites (N=170). Bold printed measurements (N=28) were used for the soil quality assessment after excluding variables with r >0.6 and/or a variable inflation factor >4 (Borcard et al. 2011).

Physical	Chemical	Biological	SOM
Clay [%]	**pH**	**Basal respiration [µg CO_2-C $g^{-1}h^{-1}$]**	**SOC [%]**
Silt [%]	**EC [µS cm^{-1}]**	C_{min} [g CO_2-C kg^{-1}]	**TON [%]**
Sand [%]	**P [mg kg^{-1}]**	**N_{min} [mg kg^{-1}]**	DOC [mg kg^{-1}]
Water holding capacity (WHC) [%]	**K [mg kg^{-1}]**	**C_{mic} [mg kg^{-1}]**	DON [mg kg^{-1}]
Pore space volume [%]	**Mg [mg kg^{-1}]**	N_{mic} [mg kg^{-1}]	**DRIFTS peak A (labile) [A.U. cm^{-1}]**
Bulk density (BD) [g cm^{-3}]	**Fe [mg kg^{-1}]**	**Anecic species richness [ind. m^{-2}]**	**DRIFTS peak B (labile) [A.U. cm^{-1}]**
Mean penetration resistance [MPa]	**Cu [mg kg^{-1}]**	**Anecic biomass [g m^{-2}]**	**DRIFTS peak C (stable) [A.U. cm^{-1}]**
Max penetration [MPa]	**Mn [mg kg^{-1}]**	Anecic abundance [ind. m^{-2}]	**DRIFTS peak D (stable) [A.U. cm^{-1}]**
Soil depth [cm]	**B [mg kg^{-1}]**	**Endogeic species richness [ind. m^{-2}]**	
Stable aggregates (SA) [%]		**Endogeic biomass [g m^{-2}]**	
		Endogeic abundance [ind. m^{-2}]	
		Epigeic species richness [ind. m^{-2}]	
		Epigeic biomass [g m^{-2}]	
		Epigeic abundance [ind. m^{-2}]	
		Earthworm species richness [m^{-2}]	
		Earthworm biomass [g m^{-2}]	
		Earthworm abundance [ind. m^{-2}]	

3.2.4 Explanatory variables

Garden management

A survey including all gardeners was carried out to collect information on garden management and gardener's intentions (see Table 3.2). An ethics approval was not required for this research according to institutional and national guidelines. The consent of the participants was obtained by virtue of survey completion. Relevant soil management questions were asked individually for each of the five common garden habitat types (lawn, meadow, vegetable bed, flower bed and berry cultivation). These habitat types were later grouped into three categories according to the degree of associated soil disturbance: annual vegetation (vegetables), perennial vegetation with herbaceous vegetation (berries and perennial flowers) and grass vegetation (meadows and lawn).

Garden characteristics

Garden characteristics (Table 3.2) were measured at the sampling site level within each garden. The variables 'former land-use' and 'history' were assessed using digital historic maps (1864-2016) from Swiss Federal Office of Topography (2017). The urbanization density was measured as a percentage of the sealed and built area around each garden with five radii (30, 50, 100, 250, 500 m) obtained in ArcGIS. In order to reduce the complexity of response variables, these sealed areas on five scales were replaced with a measure of the urban heat island (UHI), which was highly correlated with those variables (r = 0.7-0.8; Figure S 3.10). In general, the UHI effect refers to the warmer temperatures within a city compared to rural areas. We used the local deviation of mean night temperatures near the surface (0 to + 6 K) from a regional climate model of Parlow et al. (2010). This UHI effect will be referred as the urbanization density gradient ('overwarming'; Figure 3.1 and S1) in this study.

Spatial structure

To account for spatial autocorrelation of the MSQ (Figure 3.2) we used Moran Eigenvector Maps (MEM) following the instructions of Borcard et al. (2011). The MEM method decomposes the eigenvalues, which represent spatial relationships of the geographic connectivity matrix, into eigenvectors addressing significant spatial variation at various scales (Braaker et al. 2014). The MEM reflect correlations of the MSQ between and within the gardens and can therefore be used in the models to address for unaccounted variation due to spatial autocorrelation not covered by other variables (Figure S 3.3). Delaunay triangulation was chosen as an optimal spatial matrix, representing the connections of the urban gardens. Model selection was performed according to the lowest Akaike information criterion with small sample size correction (AICc; Burnham and Anderson 2003), resulting in 19 significant MEM.

Soil heavy metals

Total element concentrations of heavy metals (As, Ba, Co, Cr, Cu, Cd, Ni, Pb, Sb, V, Zn) were analysed using dried and ball milled soil samples pressed with wax to tablets using a Spectro X-lab 2000 X-Ray Fluorescence (XRF) spectrometer. This technique conforms with standard measurements of soil heavy metal concentrations (Horta et al. 2015), except for Cd (Christl et al. 2004), which were excluded.

Table 3.2 – Response variables potentially influencing soil quality of urban gardens. Most management questions were asked on a five level Likert scale and normalised by the total number of questions for the combined management variables. Survey questions are described in Table S 3.10.

Variables	Scales	**Description** (corresponding questions asked in gardener survey are in brakets)
Management variables		
Compost	no/yes	Use of compost (FertLawnCompost, FertVegCompost, FertVegFreshCompost, FertFlowerCompost,FertFlowerFreshCompost)
Disturbance	no/yes	Major soil disturbance (DiggingVeg, DiggingFlower, CareLawn)
Disturbance freq	num.	Frequency of major soil disturbance (DiggingVeg, DiggingFlower, CareLawn)
Fertilizer freq	num.	Frequency of appling fertilizer (FertLawn, FertVeg, FertFlower)
Leave freq	num.	Frequency of removing leaves in the garden (Leaves)
Pesticides	no/yes	Use of pesticides, insecticides and herbicides (PestLawn, PestFeg, PestFlower)
Pesticides freq	num.	Frequency of pesticides usage (PestLawn, PestFeg, PestFlower, PestTrees, WeedingHerbicides)
Visual	num.	Visual assessment of plant diversity per habitat type made by the gardeners (low, medium & high)
Water	no/yes	Use of additional water (WaterLawn, WaterVeg, WaterFlower)
Water freq	num.	Frequency of irrigation (WaterLawn, WaterVeg, WaterFlower)
Weeding freq	num.	Frequency of removing weeds in the garden (Weeds)
Garden characteristics		
Aspect	3 groups	Garden slope orientation (flat, NE or SW facing slope)
Bare soil	num.	Proportion of soil not covered with vegetation (digital image classification (10 m^2) of orthogonal photograph from 3 m height)
Former land-use	4 groups	Soil use before garden establishment (agriculture, landfill, urban green and vineyards) assessed by digital historic maps
Garden age	num.	Time since last major change in the garden (e.g., exchange of soil) with 2015 as reference year
Garden type	2 groups	Two types of gardens (allotment gardens and home gardens)
Habitat	3 groups	Three main garden habitat types (perennial lawn, perennial herbaceous and annual vegetables)
History	3 groups	Dominant land-use type (industry, settlement, agriculture) around each garden (500 m radius)
Impervious	num.	Proportion of sealed soils, calculated with an orthogonal photograph from 3 m height (10 m^2) and digital image classification
Urbanization	num.	Local deviation of mean night temperatures near surface (0 to + 6 K; Parlow et al. 2010) representing the urbanization
Plant SR	num.	Plant species richness identified at plot level (10 m^2)
Slope	num.	Mean inclination of the urban garden sites, measured with a digital elevation meter (10 measurements averaged)
Soil texture	5 groups	Soil texture according to USDA classification (loam, clay loam, sandy clay loam, sandy loam, silty clay loam, Figure S 3.5)
Sun exposure	num.	Solar hours measured with a solar compass at maximum vegetation stage in July 2015
Years managed	num.	How long the garden is managed by the same gardener in years

3.2.5 Soil functions

Habitat for earthworms

The suitability of habitat for soil fauna was investigated by the abundance, biomass and species richness of functional groups of earthworms. They were collected in September and October 2015 using a combined mustard (0.6%) extraction (Lawrence and Bowers 2002) and hand sorting method (Bartlett et al. 2010). The animals were collected from an area of 0.3 m × 0.3 m within the same 2 m × 2 m areas used for the soil samples and stored in 70% ethanol for species identification (Bouché 1977, Blakemore 2008, Sims and Gerard 1999). The species were classified according to three ecological groups with respect to their main vertical distribution in the soil (epigeic, endogeic and anecic) as defined by Bouché (1977). Juveniles could only be assigned to ecological categories and were used for the total earthworm biomass and abundance calculation.

Decomposition rates of tea bags

Decomposition was assessed by using standardised commercial tea bags buried in the soil, following the tea bag index (TBI) protocol by Keuskamp et al. (2013). Decomposition was mainly done by microorganisms, due to the small mesh size of the nylon bags (Setälä et al. 1996). Two types of litter decomposition were assessed by using rooibos tea as a slowly decomposable and green tea as a fast decomposable organic material. We buried four tea bags per tea type, resulting in 1360 buried tea bags (85 gardens × 2 sites × 2 different tea types × 4 replicates), for 90 days (mid-October 2015 until mid-January 2016) at a depth of 8 cm. Teabags were weighed after drying at 60 °C, before and after the incubation in the field. Additionally to the TBI protocol, soil particles, which entered the tea bags were subtracted after incineration of the bags.

3.2.6 Soil quality indices

Three indices of soil quality were calculated from the MSQ. The ratio of SOC to clay was calculated as an indicator of structural soil quality (Johannes et al. 2017). Further eco-physiological indices of soil quality were calculated using the ratio of C_{mic} to SOC (microbial quotient) as an indicator of SOM quality for microbes and the metabolic quotient (qCO_2) calculated as the ratio of basal respiration rate to C_{mic}, which describes the substrate mineralized per unit of

microbial biomass carbon (Anderson 2003).

3.2.7 Multivariate soil quality assessment

After variable selection (Figure 3.2) a fuzzy cluster analysis was performed as a multivariate technique to group garden sites with similar MSQ. Prior to the clustering, data was normalized and a Euclidean distance matrix was calculated, as it was the most appropriate distance matrix according to its rank correlations (Oksanen and Others 2011). A non-metric multidimensional scaling (NMDS) was applied to visualize the cluster groups and to draw inference about the response variables. NMDS produces a rank-based ordination on a distance or dissimilarity matrix and is regarded as the most robust unconstrained ordination method (Minchin 1987), because it tolerates quantitative, semi-quantitative, qualitative, or mixed variables (Borcard et al. 2011). A similar approach has been applied to group soils of different land-use types according to the soil quality in Italy (Marzaioli et al. 2010). Permutational multivariate analysis of variance (PERMANOVA) was performed with the MSQ as Euclidean distance response matrix and significant variables ($p \leq 0.05$) were fitted to the NMDS ordination while MEM variables were excluded to increase visibility of the main effects. Due to the influence of the habitat types in the PERMANOVA (Table S 3.2), a second NMDS analysis per habitat types (NMDS habitat) was conducted for the garden management questions (Figure & Table S 3.4). Correlations were analysed by Pearson correlation coefficients (r). Linear mixed effect models (LMEM) were fitted after analysing the model assumptions (independent and identical distributed residuals). Fixed effects were garden habitat types (3 factors), garden types (2 factors) and the urbanization density, while the garden identity was set as a random factor. Means and 95% credible intervals of the Bayesian inference posterior distributions were calculated following Korner-Nievergelt et al. (2015). All statistical analyses were performed using R 3.3.1 (R Core Team, 2017). A detailed description of each function and the corresponding R packages is given in Table S 3.14.

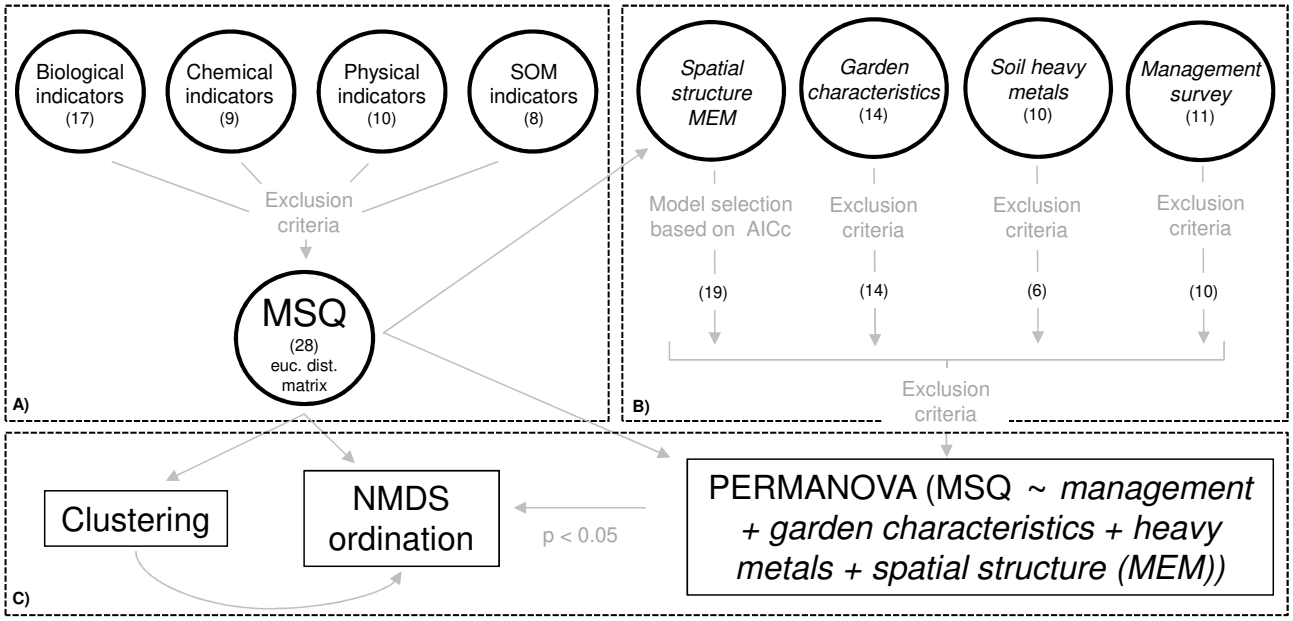


Figure 3.2 – Scheme for the multivariate soil quality assessment. A) There are in total 44 measures of soil quality (MSQ). Exclusion criteria (r <0.6 and or a variance inflation factor >4 (Borcard et al. 2011)) reduced MSQ by 16 variables. B) Explanatory variables are written in italics. Moran eigenvector maps (MEM) were selected based on smallest AICc (cf. section 3.2.4). C) MSQ as response matrix with Euclidean distances and explanatory variables were used in a PERMANOVA. Thereafter, significant explanatory variables (p<0.05) were fitted on the NMDS ordination of MSQ. A fuzzy clustering of the MSQ was obtained and the three groups were plotted on the NMDS plot (Figure 3.3).

3.3 Results

3.3.1 Urban soil quality assessment

Garden sites were grouped into three main clusters (fuzzy clustering). With the NMDS ordination (Figure 3.3) we could show that the first cluster (red dot) contained home garden sites, dominated by grass sites with high content of C_{mic}, penetration resistance and anecic earthworms biomass and species richness. The second cluster (blue triangle) included vegetables sites and allotment gardens, which have much 'bare soil' and are surrounded by 'industry' and are located within the more densified urban areas ('overwarming'). This group had high contents of basal respiration, N_{min}, nutrients (Fe, P, K, B), labile SOM compounds (Peak A & B), and endogeic earthworms biomass and species richness. The third cluster (green square) was characterised by herbaceous sites surrounded by 'agriculture' and consisting of a 'slope' (either facing NE or SW) with high contents of Mg, SOC, Cu (as micro-nutrient), As, aliphatic and aromatic SOM compounds (Peak C & D). The most important variables for the NMDS ordination ($p \leq 0.001$, Table S 3.1) were chemical (Mg, P, Fe, K, pH, Mn, B and Mn), biological (C_{mic}, resp and N_{min}), physical (SA, penetration resistance and BD) as well as SOM (Peaks A, B, C, D and SOC) measurements.

3.3.2 Effects of garden management

The frequency of using pesticides (PERMANOVA; F=6.4, $P<0.001$) and fertilizers (PERMANOVA; F=2.4, P=0.05) as well as the use of compost (PERMANOVA; F=6, $P<0.001$) indicated the biggest effects of management variables on the NMDS ordination (Table S 3.2). For perennial grass sites the date of the first lawn mowing in a year, was the only significant management variable (NMDS habitat; P=0.04, Table S 3.4). If the first cut was earlier, P (r = -0.26, P= 0.04) and Fe contents (r = -0.29, P = 0.02) were higher, but BD (r= 0.28, P= 0.03; Table S 3.3) was lower. For the sites under perennial herbaceous vegetation, the use of relatively young compost, less than one year old ('FertFlowerFreshCompost'), showed a positive correlation with the biomass of anecic earthworms (r=0.36, P=0.02) and P contents (r=0.36, P=0.04). Annual vegetable sites were significantly influenced by the cropping sequence (NMDS habitat; $P< 0.001$), correlating negatively with penetration resistance (r=-0.65, $P<0.001$) and C_{mic} (r= -0.45, P=0.01).

3.3.3 Effects of garden characteristics

Overall, the urban garden habitat type had the strongest effect (PERMANOVA; F=10.3, $P< 0.001$) and was grouped into distinct classification groups (Figure 3.3), while 'overwarming' had the second strongest effect (Table S 3.2). Urban garden soils in more densely sealed areas showed higher rates of disturbances, such as the use of 'pesticides' and 'fertilizers' accompanied by higher values of basal respiration and labile SOM compounds within the second clustering group. Allotment gardens were grouped with 'vegetables' and home gardens with 'grass' sites, respectively. In addition, the 'slope' of each urban garden site, the aspect (inclination of the garden), the history of the neighbourhood ('settlement', 'industry' or 'agriculture') and the amount of 'bare soil' showed significant effects in the PERMANOVA (Table S 3.2).

Table 3.3 – Univariate statistic of soil quality measures, used for the soil quality assessment and additional values of tea bag decomposition and heavy metals. Estimates and p-values are reported from linear mixed effect models with overwarming, garden habitat and garden type as fixed effects and garden identity as random effect. M = earthworm biomass, SR = earthworm species richness, cv = coefficient of variation.

	mean	cv	mean vegetables	mean herbaceous	mean grass	herbaceous estimate	herbaceous se	herbaceous p	grass estimate	grass se	grass p	home estimate	home se	home p	overwarming estimate	overwarming se	overwarming p
Physical																	
Clay [%]	24.0 ± 0.5	23	24.4 ± 0.8	22.9 ± 0.8	24.5 ± 0.8	-1.09	1.03	0.29	-0.03	0.93	0.98	0.21	1.15	0.86	-1.14	0.56	**0.04**
Water holding capacity (WHC) [%]	0.82 ± 0.01	16	0.83 ± 0.02	0.81 ± 0.02	0.82 ± 0.02	0.02	0.02	0.36	0.02	0.02	0.38	-0.04	0.03	0.13	0.02	0.01	0.19
Bulk density (BD) $[gcm^{-3}]$	1.07 ± 0.01	15	1.14 ± 0.02	1.11 ± 0.03	1.01 ± 0.02	-0.05	0.03	0.16	-0.14	0.03	**<0.001**	0	0.03	0.97	-0.01	0.01	0.73
Mean penetration resistance $[MPa]$	1.44 ± 0.05	40	0.98 ± 0.07	1.46 ± 0.08	1.7 ± 0.07	0.35	0.09	**<0.001**	0.64	0.08	**<0.001**	0.1	0.1	0.32	0.16	0.05	**0.01**
Soil depth $[cm]$	44.0 ± 1.6	43	50.2 ± 2.8	41.1 ± 2.6	42.5 ± 2.6	-6.16	2.82	**0.03**	-7.68	2.49	**0.01**	-1.96	4.11	0.64	-3.83	2.00	0.06
Stable aggregates (SA) [%]	83.46 ± 0.8	12	79.24 ± 1.6	82.07 ± 1.45	87.00 ± 1.1	2.7	1.88	0.16	7.23	1.72	**<0.001**	3.11	1.86	0.10	0.06	0.89	0.95
Chemical																	
pH	7.26 ± 0.02	3	7.25 ± 0.04	7.32 ± 0.03	7.23 ± 0.04	-0.02	0.04	0.68	-0.07	0.04	0.06	0.15	0.05	**0.01**	-0.02	0.03	0.57
Electrical conductivity (EC) $[\mu Scm^{-1}]$	185.4 ± 3.3	22	177.9 ± 7.5	187.3 ± 6.3	188.5 ± 4.2	3.9	8.19	0.63	5.69	7.48	0.45	14.68	7.98	0.07	1.67	3.82	0.66
P$[mgkg^{-1}]$	180.5 ± 9.5	63	260.6 ± 17.8	172.4 ± 16.1	138.6 ± 12.3	-67.08	20.17	**0.01**	-109.94	18.27	**<0.001**	-59.18	21.22	**0.01**	15.89	10.21	0.12
K $[mgkg^{-1}]$	163.5 ± 10.4	76	222.9 ± 22.9	162.2 ± 17.1	129.0 ± 14.1	-53.76	26.42	**0.04**	-87.53	24.55	**<0.001**	-39.36	22.49	0.08	18.55	10.63	0.08
Mg $[mgkg^{-1}]$	516.0 ± 14.6	34	565.5 ± 33.9	508.8 ± 20.0	491.8 ± 22.7	-46.48	26.16	0.08	-85.18	23.15	**<0.001**	42.97	37.57	0.26	-48.38	18.31	**0.01**
Fe $[mgkg^{-1}]$	367.2 ± 9.1	30	395.5 ± 21.	370.4 ± 15.3	347.9 ± 12.4	-13.17	17.13	0.44	-41.6	15.26	**0.01**	-57.5	21.89	**0.01**	44.72	10.63	**<0.001**
Cu $[mgkg^{-1}]$	31.9 ± 2.3	87	35.5 ± 4.4	31.7 ± 4.1	29.9 ± 3.6	-0.09	2.35	0.97	-4.43	2.03	**0.03**	-21.11	6.38	**0.01**	9.88	3.14	**0.01**
Mn $[mgkg^{-1}]$	293.7 ± 9.2	38	291.9 ± 20.2	279.6± 13.3	305.3 ± 14.9	-2.96	15.79	0.85	17.64	13.93	0.21	29.26	24.01	0.23	-34.1	11.72	**0.01**
B $[mgkg^{-1}]$	1.36 ± 0.06	53	1.88 ± 0.11	1.34 ± 0.1	1.06 ± 0.07	-0.4	0.12	**0.01**	-0.72	0.11	**<0.001**	-0.37	0.13	**0.01**	0.13	0.06	**0.05**
Biological																	
Basal respiration $[\mu gCO_2 - Cg^{-1}h^{-1}]$	0.24 ± 0.01	47	0.27 ± 0.02	0.24 ± 0.02	0.22 ± 0.01	-0.02	0.02	0.44	-0.05	0.02	**0.02**	-0.01	0.02	0.52	0.01	0.01	0.22
C_{mic} $[mgkg^{-1}]$	814.57 ± 22.2	33	713.06 ± 41.4	796.28 ± 36.9	888.72 ± 34.1	75.11	51.76	0.15	157.13	47.07	**0.01**	119.25	52.32	**0.03**	-27.95	25.1	0.27
N_{mic} $[mgkg^{-1}]$	142.29 ± 4.4	37	118.66 ± 7.3	140.76 ± 7.4	157.53 ± 6.9	19.32	10.14	0.06	35.59	9.23	**<0.001**	23.57	10.21	**0.02**	-3.92	4.9	0.43
N_{min} $[mgkg^{-1}]$	1.78 ± 0.1	65	1.70 ± 0.1	1.84 ± 0.2	1.78 ± 0.2	0.21	0.23	0.36	0.09	0.2	0.67	-0.27	0.24	0.28	0.08	0.12	0.51
Anecic SR $[ind.m^{-2}]$	1.26 ± 0.1	71	1.35 ± 0.2	1.48 ± 0.1	1.05 ± 0.1	0.18	0.19	0.33	-0.26	0.17	0.14	-0.12	0.17	0.47	-0.02	0.08	0.77
Anecic M $[gm^{-2}]$	7.93 ± 0.5	80	7.52 ± 0.9	8.78 ± 1.2	7.55 ± 0.7	1.52	1.38	0.27	0.1	1.27	0.94	0.49	1.26	0.70	-0.48	0.6	0.43
Endogeic SR $[ind.m^{-2}]$	1.73 ± 0.1	73	2.27 ± 0.2	1.59 ± 0.1	1.52 ± 0.2	-0.53	0.26	**0.04**	-0.75	0.24	**0.01**	-0.11	0.25	0.66	-0.13	0.12	0.28
Endogeic M $[gm^{-2}]$	3.31 ± 0.3	93	5.15 ± 0.7	2.74 ± 0.4	2.65 ± 0.3	-2.27	0.64	**<0.001**	-2.53	0.59	**<0.001**	-0.08	0.56	0.88	-0.07	0.27	0.80
Epigeic SR $[ind.m^{-2}]$	0.62 ± 0.1	92	0.43 ± 0.1	0.83 ± 0.1	0.64 ± 0.	0.001	0.001	1.00	0.001	0.001	1.00	-0.33	0.3	1.00	0.06	0.18	1.00
Epigeic M $[gm^{-2}]$	3.43 ± 0.4	130	1.43 ± 0.3	4.26 ± 0.6	4.24 ± 0.7	4.28	2.27	0.07	2.66	1.78	0.16	-3.26	2.27	0.17	-0.08	1.00	0.94
Earthworm SR $[ind.m^{-2}]$	3.1 ± 0.2	57	3.7 ± 0.3	3.17 ± 0.3	2.68 ± 0.2	-0.32	0.36	0.38	-0.99	0.33	**0.01**	-0.29	0.35	0.41	-0.1	0.17	0.55
Earthworm M $[gm^{-2}]$	125.52 ± 7.2	69	141.05 ± 14.8	128.55 ± 15.2	114 ± 8.9	-7.9	18.83	0.68	-26.84	17.34	0.12	4.08	17.09	0.81	-5.93	8.13	0.47
Earthworm abundance $[ind.m^{-2}]$	227.35 ± 15.5	82	336.63 ± 40.8	183.57 ± 20.8	194.62 ± 18.5	-135.79	38.72	**<0.001**	-134.45	35.91	**<0.001**	-16.22	33.51	0.63	-19.29	15.86	0.23
SOM																	
SOC [%]	4.71 ± 0.1	35	5.18 ± 0.3	4.82 ± 0.2	4.36 ± 0.2	-0.15	0.28	0.59	-0.63	0.25	**0.01**	-0.55	0.35	0.12	0.39	0.17	**0.02**
TON [%]	0.33 ± 0.01	34	0.36 ± 0.02	0.33 ± 0.02	0.32 ± 0.01	-0.02	0.02	0.29	-0.03	0.02	0.10	-0.03	0.02	0.23	0.01	0.01	0.57
Peak A $[A.U.cm^{-1}]$	1.23 ± 0.03	32	1.02 ± 0.05	1.25 ± 0.06	1.34 ± 0.05	0.25	0.08	**0.01**	0.35	0.07	**<0.001**	-0.13	0.07	0.08	0.06	0.04	0.12
Peak B $[A.U.cm^{-1}]$	0.07 ± 0.001	24	0.06 ± 0.001	0.07 ± 0.001	0.07 ± 0.001	0.001	0.001	0.24	0.01	0.001	0.03	0.001	0.001	0.73	0.001	0.001	0.25
Peak C $[A.U.cm^{-1}]$	0.33 ± 0.01	24	0.31 ± 0.01	0.34 ± 0.01	0.34 ± 0.01	0.02	0.02	0.32	0.02	0.01	0.21	0.02	0.02	0.29	-0.01	0.01	0.31
Peak D $[A.U.cm^{-1}]$	0.9 ± 0.03	36	0.91 ± 0.05	0.96 ± 0.04	0.86 ± 0.05	0.03	0.05	0.52	-0.04	0.05	0.36	-0.02	0.07	0.81	0.05	0.03	0.15
Soil quality indices																	
C_{mic}/SOC [%]	1.9 ± 0.07	42	1.47 ± 0.09	1.78 ± 0.1	2.24 ± 0.11	0.25	0.13	0.06	0.68	0.12	**<0.001**	0.46	0.15	**0.01**	-0.19	0.07	**0.01**
qCO_2 $[\mu gCO_2 - Cg^{-1}h^{-1}C_{mic}^{-1}h^{-1}]$	0.31 ± 0.01	40	0.38 ± 0.02	0.3 ± 0.02	0.27 ± 0.01	-0.06	0.02	**0.01**	-0.1	0.02	**<0.001**	-0.06	0.02	**0.01**	0.02	0.01	**0.04**
SOC/clay	0.21 ± 0.01	45	0.23 ± 0.02	0.22 ± 0.01	0.19 ± 0.01	0	0.02	0.91	-0.03	0.02	**0.05**	-0.02	0.02	0.23	0.03	0.01	**0.01**
TBI																	
Green tea decomposition [%]	59.0 ± 3.7	7	57.7± 7.1	58.5 ± 5.8	60.1 ± 5.8	0.01	0.01	0.61	0.02	0.01	**0.01**	-0.01	0.01	0.44	0.01	0.001	**0.02**
Rooibos tea decomposition [%]	29.6 ± 2.8	11	29.4 ± 4.9	29.8 ± 5.4	29.5 ±4.2	0.001	0.01	0.99	0.001	0.01	0.78	0.01	0.01	0.36	0.001	0.001	0.44
Heavy metals																	
Sb $[mgkg^{-1}]$	1.91 ± 0.4	231	2.47 ± 0.9	1.62 ± 0.3	1.79 ± 0.6	-1.35	0.88	0.13	-0.74	0.8	0.35	0.87	0.92	0.35	-0.14	0.44	0.75
As $[mgkg^{-1}]$	9.41 ± 0.3	37	8.85 ± 0.7	9.55 ± 0.5	9.63 ± 0.4	0.01	0.43	0.99	0.51	0.38	0.18	0.89	0.81	0.27	-0.39	0.4	0.33
Co $[mgkg^{-1}]$	31.60 ± 0.4	15	31.31 ± 0.8	32.62 ± 0.7	31.02 ± 0.6	0.64	0.94	0.50	-0.72	0.86	0.40	2.38	0.94	**0.01**	-1.1	0.45	**0.02**
Cu $[mgkg^{-1}]$	77.30 ± 4.9	77	85 ± 9.0	77.44 ± 9.0	72.6 ± 7.8	-9.32	5.79	0.11	-12.59	5.02	**0.01**	-4.77	14.28	0.74	8.06	7.02	0.25
Pb $[mgkg^{-1}]$	179.14 ± 15.0	101	195.47 ± 27.5	171.54 ± 24.0	175.02 ± 25.7	-15.06	28.53	0.60	-10.51	25.27	0.68	-15.52	40.16	0.70	8.22	19.56	0.68
V $[mgkg^{-1}]$	79.40 ± 1.3	19	75.73 ± 2.2	81.72 ± 2.3	79.88 ± 2.0	2.97	2.4	0.22	3.45	2.13	0.11	7.08	3.22	**0.03**	-4.41	1.57	**0.01**

3.3.4 Effects of heavy metals

As (PERMANOVA; F=7.5, P<0.001) and Cu (PERMANOVA; F=3.0, P=0.02) were the only heavy metals with a significant effect on the NMDS ordination of MSQ. In turn, the variables with effects on the heavy metal distribution were intensive soil management variables, represented by the use of mineral fertilizer and removing weeds and leaves. The slope and the former land-use showed an effect on the heavy metal distribution, being positively correlated with Cu (r=0.28, P=0.01), Pb (r=0.20, P=0.02) and Sb (r=0.19, P=0.02). The variation of heavy metals was considerably high between and within gardens. For instance Pb concentrations ranged from 18.5 to 1076.0 mg kg^{-1} (Table S 3.6). Generally, more Cu was found in vegetables sites and more Co as well as V was found in allotment gardens (Table 3.3).

3.3.5 Differences in functional groups of earthworms and decomposition

The total abundance as well as the species richness and biomass of endogeic earthworms were enhanced in vegetable sites, but not between garden types (Table 3.3). Total earthworm species richness was highest in vegetable sites, followed by herbaceous sites. Compost application was correlated with endogeic species richness (p=0.04), while 'bare soil' was correlated with biomass (r=0.24, P<0.001) and species richness of endogeic earthworms (r=0.18, P=0.03) as well as total earthworm species richness (r=0.2, P=0.02). The biomass of earthworms correlated with the decomposition of rooibos tea (r=0.17, P=0.04). Decomposition of green tea was higher (Table 3.3) in grass sites (60.1% decomposed material) than in vegetable sites (57.7%). In addition to the garden habitat, urbanization density had a positive (r=0.2, P=0.02) and 'bare soil' a negative (r=-0.17, P=0.05) correlation with green tea decomposition. There was no significant correlation between decomposition and heavy metal concentrations (Table S 3.7).

3.3.6 Soil quality indices

The C_{mic}/SOC ratio was lowest in vegetables and highest in grass sites, while the qCO_2 was lowest in grass and highest in vegetable sites (Figure 3.4). The amount of 'bare soil' was positively correlated with qCO_2 (r=0.36, P<0.001) and negatively with the C_{mic}/SOC ratio (r=-0.35, P<0.001). The structural soil quality indicator from Johannes et al. (2017) revealed that 85% of the gardens (Figure S 3.2) had a SOC/clay ratio >1:13 and 4% were below the 1:10 threshold.

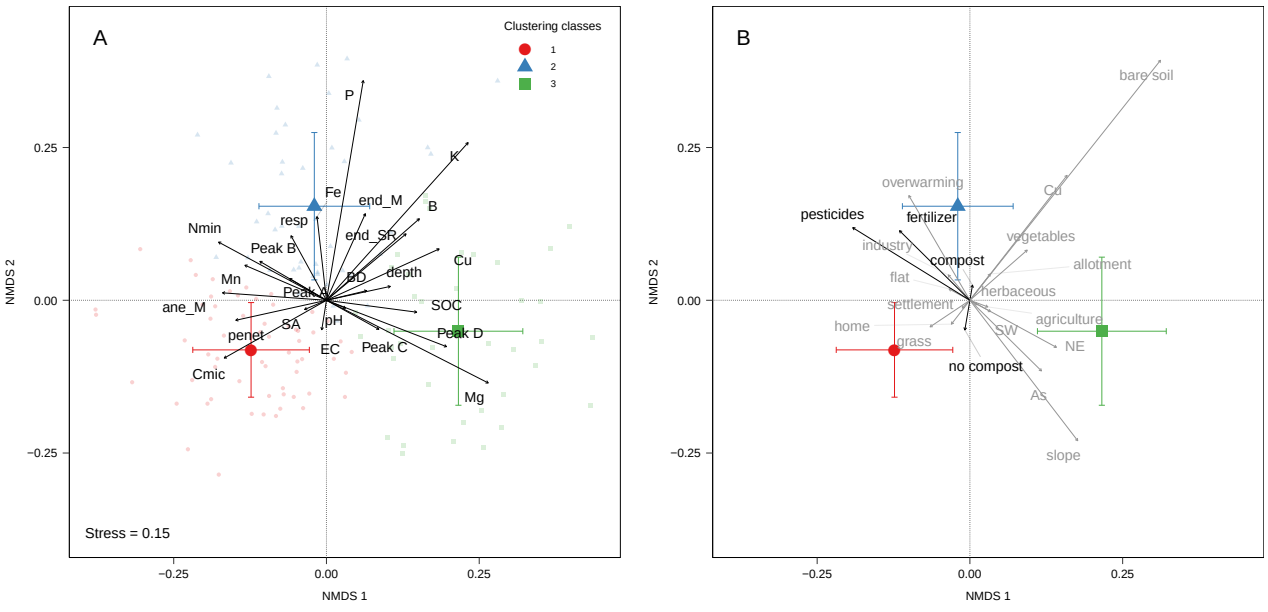


Figure 3.3 – A) Multivariate NMDS ordination of significant (PERMANOVA, Table S 3.1) measures of soil quality (MSQ). Colours and symbols correspond to different groups of soil quality based on a fuzzy clustering. B) Biplot of management (black) and garden characteristics (grey) fitted on the NMDS ordination of plot A). Only significant variables from PERMANOVA $p \leq 0.05$ are displayed in B), while MEM were excluded. ane_M = anecic biomass, end_M = endogeic biomass, end_SR = endogeic species richness, resp = basal respiration, penet = penetration resistance, BD = bulk density, SA = stable aggregates.

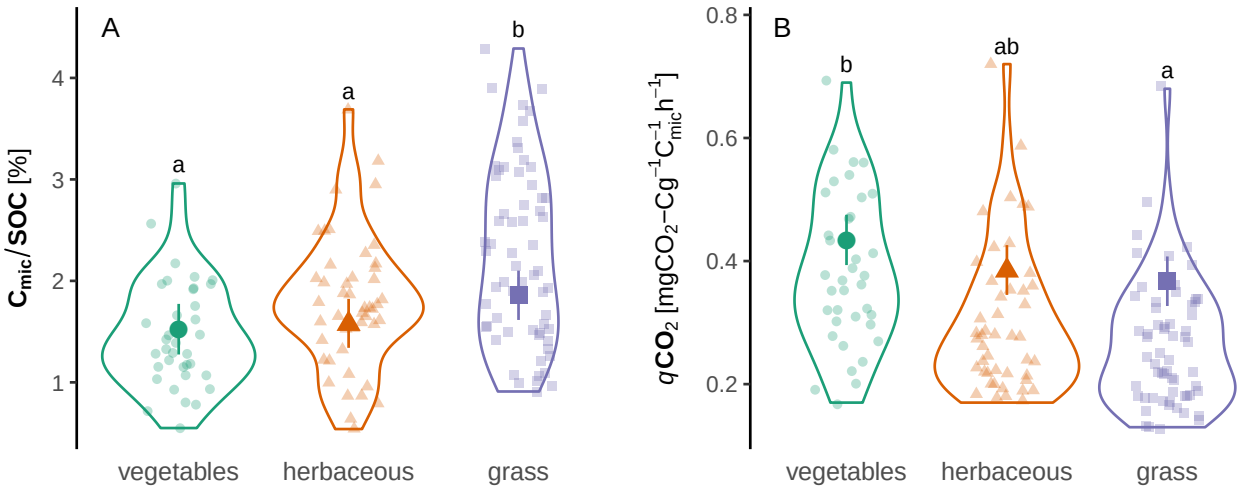


Figure 3.4 – Violin plots showing probability density of the raw data of A) Microbial quotient C_{mic}/SOC and B) the metabolic quotient qCO_2. Colours correspond to the different garden habitat types with the following observation numbers: Annual vegetables (N=37), perennial herbaceous (N=46) and perennial grass (N=62). Bold points are mean values of the simulated Bayesian inference posterior distribution taking into account the random effect of garden identity with the 95% credible intervals as lines.Letters are least square means from linear mixed effect models (cf. Table S 3.11) with a significance level alpha = 0.05.

3.4 Discussion

3.4.1 Soil quality assessment

Along with the ongoing discussion about the effect of urbanization on plant communities and soil properties (Groffman et al. 2017), we have found that garden habitats showed distinct MSQ. Thus, the way gardeners manage their habitats acts as an important driver of soil properties. In contrast to Edmondson et al. (2014), who found no differences in topsoil contents of SOC, TON, BD and C/N ratio between soils of allotment, home and non-domestic green spaces, we found distinct differences in physical, chemical and biological MSQ (Table 3.3) between allotment and home gardens but most pronounced between the contrastingly managed habitat types. This conforms with the first hypothesis, that garden management practices have an effect on soil properties. Our second hypothesis that biological soil quality measures will be most affected by garden management practices could not be confirmed, since other measures, such as physiochemical soil properties were equally important in the NMDS ordination. This underpins the importance of assessing soil quality with a comprehensive set of inherent and dynamic soil quality measurements. Overall, our results support the importance of anthropogenic management in urban ecology theory (Alberti 1999), but we found little evidence for a convergence of soil properties with urbanization contrary to the urban convergence hypothesis of McKinney (2006). Whereas (Pouyat et al. 2015) found that topsoil contents of SOC and TON converged, but others such as K and P rather diverged in five metropolitan areas. In contrast to this study we found higher SOC with increasing urbanization density. This may be explained by less intensive managed urban gardens such as more grass sites in the inner parts of Zurich. Our results support the findings of Greinert (2015), that variation in soil properties increases with anthropogenic management and land-use types. The urban convergence theory does not take ecosystem services into account, therefore further research is needed to close this gap. Similar to the findings of Pouyat and Carreiro (2003), who showed increased litter decomposition rates in urban than in rural forests, we found a positive correlation of decomposition and urbanization density. This might be due to the enhanced microbial decomposer activity caused by the UHI effect (Craine et al. 2010).

It has been proven that ecological gardening practices such as the application of organic inputs (Cogger 2005, Sax et al. 2017) or less soil disturbance in gardens (Grewal et al. 2011) contribute to the aboveground biodiversity (e.g. Fuller et al. 2007, Sperling and Lortie 2010). Further consequences on the nutrient cycle or soil quality are still poorly explored. We could show that sites with frequent compost application had higher P contents, endogeic earthworm species richness and biomass as well as lower penetration resistance. Other studies showed that organic matter inputs such as compost or mulch improved physical soil properties (Beniston and Lal 2012). We

support the fact that compost is a valuable regenerative soil management practice, but its quality must be tested, as urban organic waste may contain soil contaminants (Alvarenga et al. 2015).

In general, our results confirm the findings of Edmondson et al. (2014) that soil quality can be better in urban gardens than in agricultural soils. For example, topsoil content of C_{mic}, was on average 57% higher (cf. Table 3.4) than the reference values for Swiss croplands (Oberholzer et al. 1999), but was still 2.5 times lower than extensive grassland values (Oberholzer and Scheid 2007). Similarly, N_{mic} content was found to be 82% higher than Swiss croplands (Oberholzer et al. 1999) but four times lower than extensive grassland values (Oberholzer and Scheid 2007). The mean density of 225 earthworms per m^2 was higher compared to conventionally tilled Swiss agricultural soils (Kuntz et al. 2013) but at the same level as urban soils in Neuchâtel (CH) found by Amossé et al. (2016). The mean biomass of earthworms was lower by a factor of 2.4 compared to the reference value for Swiss grasslands (Stähli et al. 1997). This is probably because agricultural fields are dominated by anecic earthworm species, known to have a high body weight and abundance (Bouché 1977). Bulk density was 13% lower and stable soil aggregates were 51% higher in comparison with agricultural soils (Mäder et al. 2002) and nutrient contents such as P or K have been enriched by a factor of 8.4 and 1.7, respectively.

3.4.2 Effect of soil disturbance on urban garden soils

Overall, we found that intensive soil management such as the use of pesticides and fertilizers were associated with higher values of respiration rates and labile compounds of SOM (Figure 3.3). We hypothesized that more intense soil management results in a lower C_{mic}/SOC ratio and a higher qCO_2. We observed lower C_{mic}/SOC and higher qCO_2 values (Figure 3.4) in vegetables than in the less frequently disturbed grass sites, which is comparable with the findings of Anderson (2003) who reported higher C_{mic}/SOC and lower qCO_2 values for soils under crop rotation. However, these two indices cannot discriminate between disturbed or stressed (e.g. nutrient poor) ecosystems but are still valuable bio-indicators of soil quality (Wardle and Ghani 2018). Inspired by Johannes et al. (2017) we found increased SOC/clay ratios compared to agriculture sites, but this relationship must be further investigated for other non-arable soils.

3.4.3 Heavy metal assessement

Soil contamination is a potential health risk for gardeners who consume their own vegetables and especially for children who play on the ground and ingest contaminated soil directly (Kim et al. 2014). Soil contamination varies widely and depends on emission sources such as industry (Salvagio Manta et al. 2002) and traffic (Beniston and Lal 2012, Filippelli and Laidlaw 2010) but also on geogenic factors.Pb contamination in urban soils is mainly due to historical use of leaded paint, vehicle (Chaney and Ryan 1994) and industrial emissions (Beniston and Lal 2012). Although the use of Pb is strongly restricted today, it still represents a serious risk for human health due to its high persistence in soils (Filippelli and Laidlaw 2010). Total Pb concentrations in urban gardens worldwide vary widely (e.g. from 60 to $>$ 2500 mg kg^{-1} Attanayake et al. (2014), Cheng et al. (2011)), this is also confirmed in this study, whereby the concentrations ranged between 18.5 to 1076 mg kg^{-1}. Nonetheless, the mean value of Pb was relatively low compared to other cities (Table 3.4). A potential public health risk exists in soils with $>$ 1000 mg kg^{-1} Pb (Whitzling et al. 2010), which makes them unsuitable for urban gardening (Beniston and Lal 2012). This is also the upper guideline limit for Pb in Swiss soils (Swiss Federal Council 1998), which was exceeded only once in this study. The threshold requiring a further inspection of the soils (300 mg Pb kg^{-1}) was exceeded in 16% of all garden sites (18 allotments and 10 home gardens), thus requiring further examination. It has to be mentioned, that the XRF device rather overestimates heavy metal concentrations by 10-20% compared to the standard method with aqua regia digestion (Christl et al. 2004).

We assumed that heavy metals had a negative impact on decomposition and functional groups of earthworms, but no correlation was found. A possible explanation is that the positive influence of garden management particularly the high SOM content, may have masked potentially toxic effects (Fliessbach et al. 1994). Besides that, we found positive correlations of Cu, Pb and Sb with the 'slope' as well as the broad scale spatial variable 'MEM2' (Figure S 3.7). The latter can be interpreted as an increased exposure to industrial facilities in the North Eastern parts of Zurich. A certain bias of our results may derive from the fact, that the survey questions represent only the current but not the past management practices. While we have focused on heavy metals, there are other contaminations common to urban areas, such as polycyclic aromatic hydrocarbons and polychlorinated biphenyls (Vane et al. 2014), which can

have negative impact on urban food production and therefore should be addressed in future studies.

3.4.4 Gardening effects on functional groups of earthworms and decomposition

It has been shown that the loss of soil biota has a negative impact on soil functions and thus also on the health of ecosystems in natural soils (Handa et al. 2014). We found that garden management not only influences soil quality, but also functional groups of earthworms. Surprisingly, we found the highest diversity and biomass of earthworms in vegetable sites that are more frequently disturbed than grass sites. Although intensive agricultural soil management practices adversely affect total earthworm species numbers (Smith et al. 2008), the impacts may also depend on ecological groups of earthworms. For example, the number of endogeic species can also increase with tillage due to increased food supply, while the abundance of deep burrowing anecic species decreases (Chan 2000). This could also explain the positive correlation between 'bare soil' and endogeic biomass. Interestingly, 'compost' was also positively related to endogeic earthworms, highlighting the importance of providing food to earthworms. In contrast to the study carried out in urban forests of New York (Steinberg et al. 1997), we found no correlation between earthworm abundance and urban density. Moreover, we found higher decomposition rates in grass than in vegetables sites (Table 3.3, S11), unlike Grewal et al. (2011) who reported no differences in the decomposition rates between vacant lots, mainly consisting of grass sites and urban gardens, dominated by vegetables and fruits. With the modification of the TBI protocol given by Keuskamp et al. (2013), we tried to minimise bias in mass loss due to small soil particles which could enter the mesh size of the tea bags. However, this did not prevent from possible ingrowth of fungal hyphae, representing a deficiency of the TBI method. Beyond that, Pelosi et al. (2016) observed that functional but not structural diversity decreases with soil tillage intensity. Therefore, future research on the effects of soil management on soil fauna should not only focus on structural but also on functional diversity.

3.4.5 Conclusion

Soil is essential for the functioning of urban ecosystems (Levin et al. 2017). Although urban garden soils are an important part of urban green spaces, they are still poorly investigated. The multivariate approach has allowed for an integrated and comprehensive evaluation of the impact of garden management on soil quality. Garden management was identified as the main driver of the differentiation of gardens according to soil quality. Our results show the fragmentation of soil quality within a garden and the city, which was mainly dependent on the habitat (annual vegetables, herbaceous and perennial grass). Other gardening practices, such as mulching or the use of compost, improve soil quality, whereas soil disturbance such as frequent digging and compaction may change soil biological activity. The eco-physiological indicators of soil quality qCO_2 and C_{mic}/SOC were useful in identifying soil disturbance and the intensity of soil use, indicating diverging biogenic soil quality properties in contrastingly managed urban garden soils. Moreover, we found that the gardening practices lead to an increased SOC/clay ratio indicating a higher structural soil quality and that biological measures of soil quality such as C_{mic} or earthworm densities and species richness are strongly increased compared to agricultural soils. The multivariate approach by combining management, garden as well as city characteristics with soil quality measurements, allowed for disentangling the gardener's impact on urban soils. This may also be applied to other research questions focusing on soil quality in urban as well as in rural areas. In summary, this study contributes to a better understanding of the role of garden soils in urban ecosystems and provides inputs for a sustainable management practice of garden soils.

Author Contributions

AF, MM, RL, PM DF, ST conceived and designed the research, DF, AZ and ST performed the field work. ST performed the lab work and analysed the data. All authors contributed to the writing of the manuscript.

Funding

We gratefully acknowledge the financial support for this interdisciplinary project BetterGardens provided by the Swiss National Science Foundation in frame of the Sinergia programm (CRSII1_154416).

Table 3.4 – Comparison of selected physical, chemical and biological properties of urban soils compared to other urban studies or reference values from Swiss agricultural topsoils. Mean values calculated by urban garden habitats annual vegetables (N=39), perennial herbaceous (N=44) and perennial grass (N=62).

	this study		other studies		references
Physical properties					
Stable aggregates [%]	vegetables	77.3 ± 1.8	55	Conventional agricultural fields, CHE	Mäder et al. (2002)
	herbaceous	82.7 ± 1.4			
	grass	87 ± 1.1			
	mean	83 ± 0.8			
Bulk density [g cm^{-3}]	vegetables	1.15 ± 0.02	1.09	Urban garden soils, Bottrop, GER	Burghardt and Schneider (2016)
	herbaceous	1.10 ± 0.03	1.23	Conventional agricultural fields, CHE	Mäder et al. (2002)
	grass	1.01 ± 0.02			
	mean	1.07 ± 0.01			
Chemical properties					
Phosphorus [mg kg^{-1}]	vegetables	247.5 ± 16.6	21.4	Conventional agricultural fields, CHE	Mäder et al. (2002)
	herbaceous	176.0 ± 16.6	90 ± 15	Urban cites including gardens of Baltimore, USA	Pouyat et al. (2007)
	grass	138.7 ± 12.4			
	mean	179.2 ± 9.3			
Kalium [mg kg^{-1}]	vegetables	212.2 ± 22.1	97.5	Conventional agricultural fields, CHE	Mäder et al. (2002)
	herbaceous	166.6 ± 17.5	106 ± 5.2	Urban cites including gardens of Baltimore, USA	Pouyat et al. (2007)
	grass	129.16 ± 14.1			
	mean	162.8 ± 10.3			
Lead [mg kg^{-1}]	vegetables	181.3 ± 26.8	36	European agricultural soils GEMAS, EU	Reimann et al. (2012)
	herbaceous	175.7 ± 24.9	74.6	Vegetable garden soils three french cities, FRA	Joimel et al. (2016)
	grass	175.0 ± 25.7	202	Urban parks Palermo, ITA	Salvagio Manta et al. (2002)
	mean	176.9 ± 15.1	231 ± 53	Urban cites including gardens of Baltimore, USA	Pouyat et al. (2007)
			266	Garden soils of 50 UK cities, GBR	Alloway (2004)
			291	Urban garden soils, Bottrop, GER	Burghardt and Schneider (2016)
Biological properties					
C_{mic} [mg kg^{-1}]	vegetables	699.1 ± 39.9	321	Urban park soils of Aberdeen, GBR	Yuangen et al. (2006)
	herbaceous	808.6 ± 37.5	518	Reference values Swiss cropland, CHE	Oberholzer et al. (1999)
	grass	888.7 ± 34.1	2077	Reference values Swiss grassland extensive, CHE	Oberholzer and Scheid (2007)
	mean	813.2 ± 22.4			
N_{mic} [mg kg^{-1}]	vegetables	116.3 ± 7.0	78	Reference values Swiss cropland, CHE	Oberholzer et al. (1999)
	herbaceous	142.9 ± 7.5	573	Reference values Swiss grassland extensive, CHE	Oberholzer and Scheid (2007)
	grass	157.5 ± 6.8			
	mean	142.0 ± 42			
SOC [%]			2.6	Vegetable garden soils three french cities, FRA	
	vegetables	5.0 ± 0.3	2.9	Urban park soils of Aberdeen, GBR	Yuangen et al. (2006)
	herbaceous	4.9 ± 0.2	5.2	Urban soils of Neuchâtel, CHE	Amossé et al. (2016)
	grass	4.3 ± 0.2	5.2	Allotment gardens (vegetable beds) GER	Burghardt and Schneider (2016)
	mean	4.7 ± 0.1	5.8	Allotment gardens (lawn), GER	Burghardt and Schneider (2016)
Earthworm density [ind. m^{-2}]	vegetables	321 ± 40	25	Urban-rural oak forest stands New York City, USA	Steinberg et al. (1997)
	herbaceous	183 ± 22	54	Conventional agricultural fields, CHE	Kuntz et al. (2013)
	grass	195 ± 18	93	Floodplains, Thur river, CHE	Fournier et al. (2012)
	mean	225 ± 18	220	Urban soils of Neuchâtel, CHE	Amossé et al. (2016)
Earthworm biomass [g m^{-2}]	vegetables	133.3 ± 14.9	301	Reference values Swiss grassland sites, CHE	Stähli et al. (1997)
	herbaceous	127.6 ± 15.9			
	grass	114.0 ± 8.9			
	mean	123.3 ± 7.3			
Total earthworm species	vegetables	13	13	Urban parks Basel, CHE	Glasstetter and Nagel (2001)
	herbaceous	15	18	Urban cites including gardens of Baltimore, USA	Pouyat et al. (2010)
	grass	15	19	Urban sites Brussels, BEL	Pižl and Josens (1995)
	total	18			

Acknowledgments

We thank the project coordinator Dr. Robert Home and the whole BetterGardens team. We are grateful to Prof. Rainer Schulin (XRF analysis), Dr. Frank Rasche and Dr. Scott Demian (DRIFTS analysis), Dr. Lukas Pfiffner (earthworm identification) as well as to Dr. Daniel Haefelfinger, Adolphe Munyangabe, Anton Kuhn, Lena Fischer, Stefan Grubelnig, Reto Henzmann, Bernhard Stehle, Rebekka Tresch, Roger Köchlin and Björn Studer for their extraordinary support in the field and lab work. We acknowledge in particular the willingness and interest of the 85 participating gardeners of this study, who provided access and let us dig and sample soils in their gardens.

Supplementary Material

The Supplementary Material for this article can be found online at: www.frontiersin.org/articles/10.3389/fenvs.2018.00025/full#supplementary-material

Conflict of Interest Statement

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

Copyright

3.5 Supplementary Tables and Figures

3.5.1 Supplementary Figures

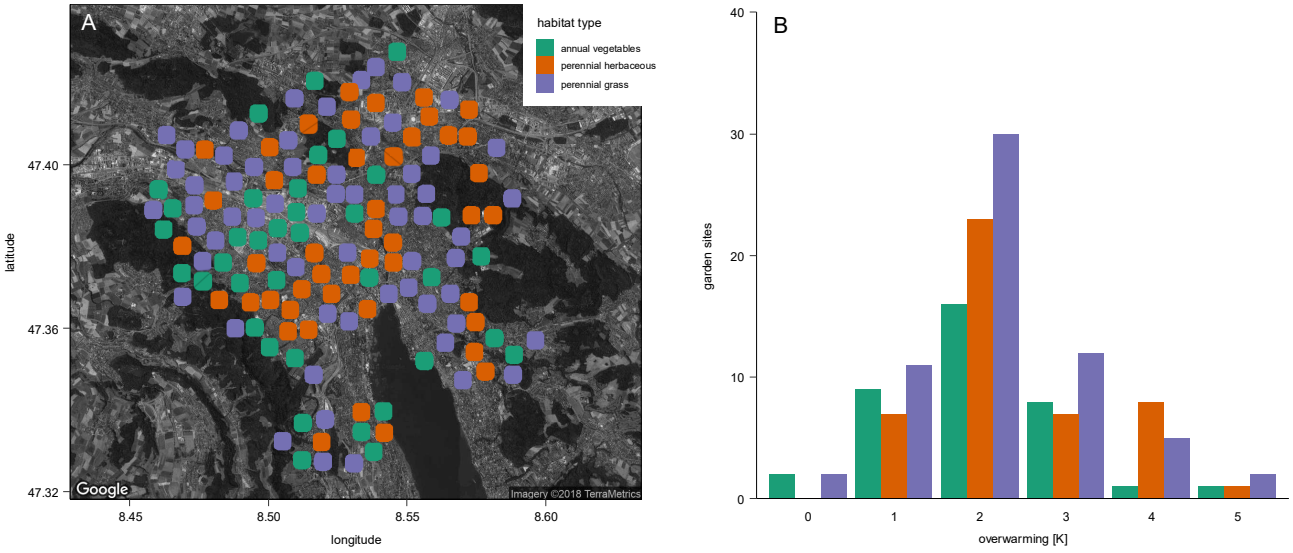


Figure S 3.1 – Urban gardens habitat types analysed in the city of Zurich (85 urban gardens x 2 sampling sites per garden). Colours correspond to the garden habitat types. Right: Density map of the urbanization gradient according to the three garden habitat types. The urbanization gradient is represented in this study by a regional climate model with local deviation of mean night temperatures near surface from 0 to + 6 K (Parlow et al. 2010).

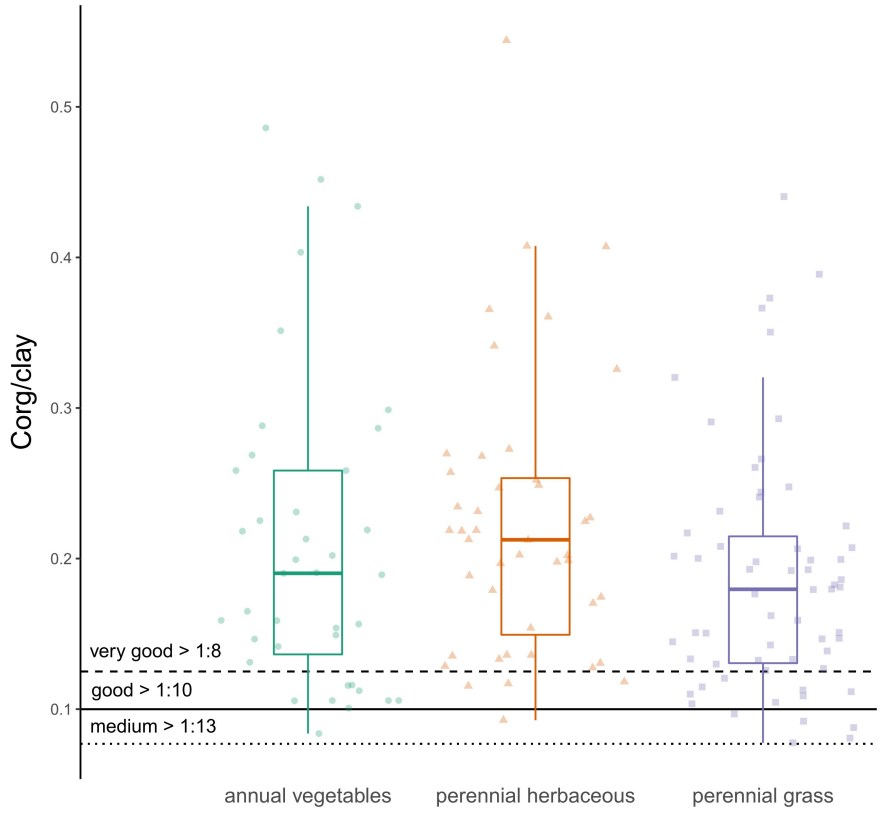


Figure S 3.2 – Structural soil quality defined as the ratio of C_{org}/clay of urban gardens of Zurich. Colours correspond to the main garden habitat types (annual vegetables N=39, perennial herbaceous N=44 and perennial grass N=62). Threshold values as defined by Johannes et al. (2017).

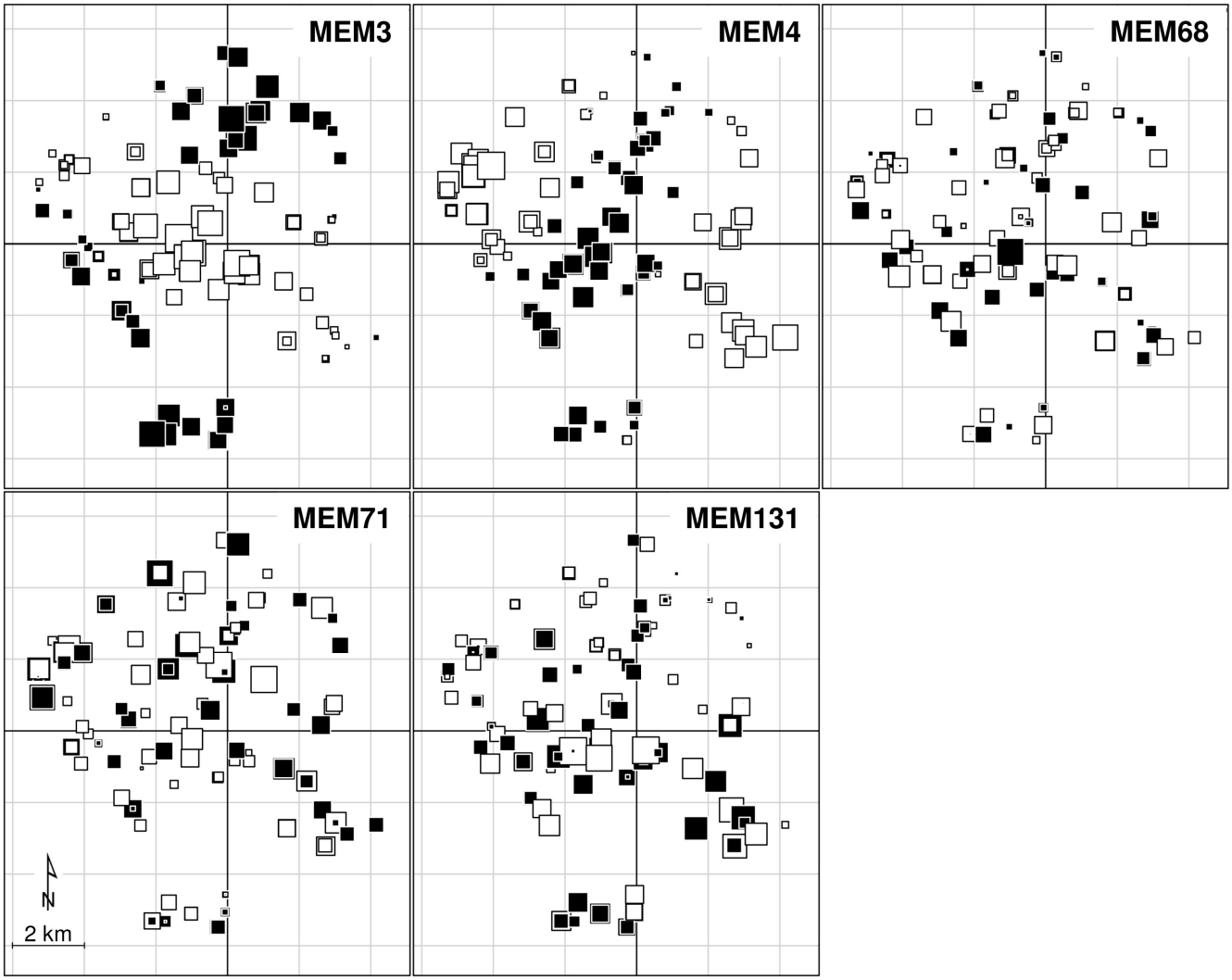


Figure S 3.3 – Moran eigenvector maps (MEM) illustrating spatial correlation and similarity of soil quality measurements (SQM). Increasing size of the symbols correspond to increasing positive values in black and increasing negative values in white, of the eigenvector (Hawkins et al. 2007).

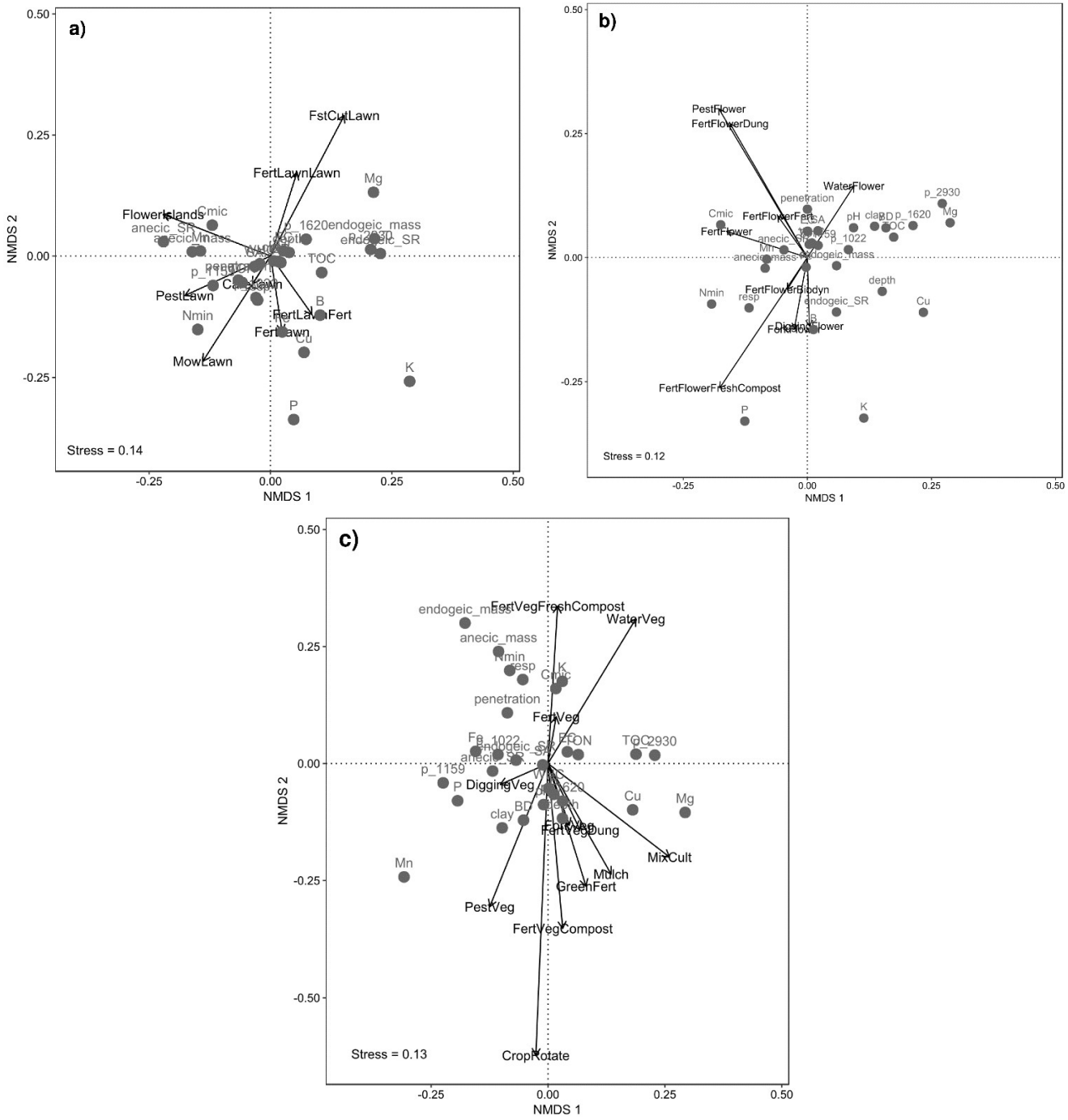


Figure S 3.4 – NMDS ordination of soil quality measurements with fitted garden management variables (NMDS habitat). Total data set was divided into sites consisting of a) lawn (N=62), b) herbaceous (N=46) and c) vegetables (N=37).

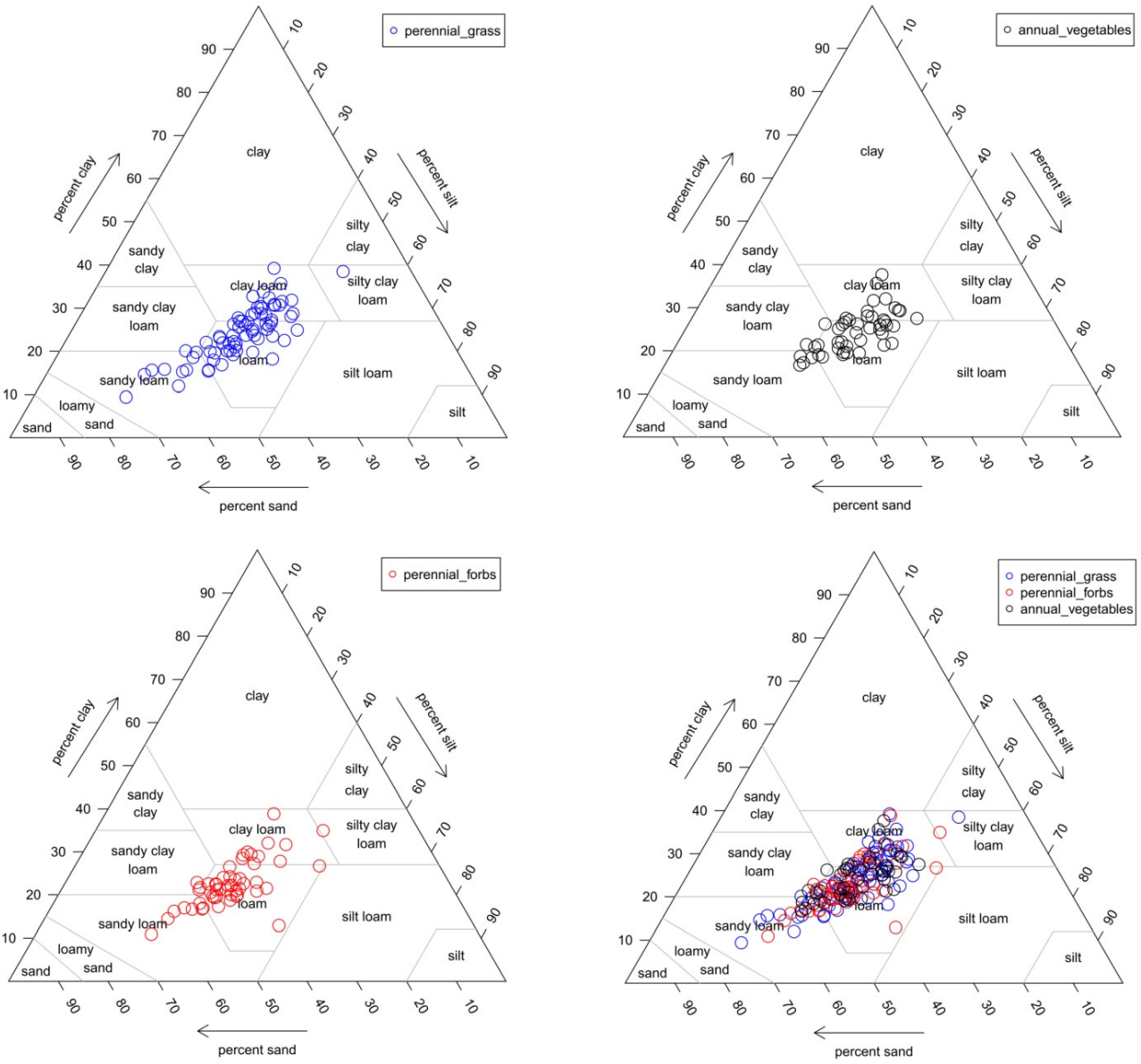


Figure S 3.5 – Soil texture according to USDA classification. Perennial grass N=71, Perennial herbaceous N=52, Annual vegetables N=47.

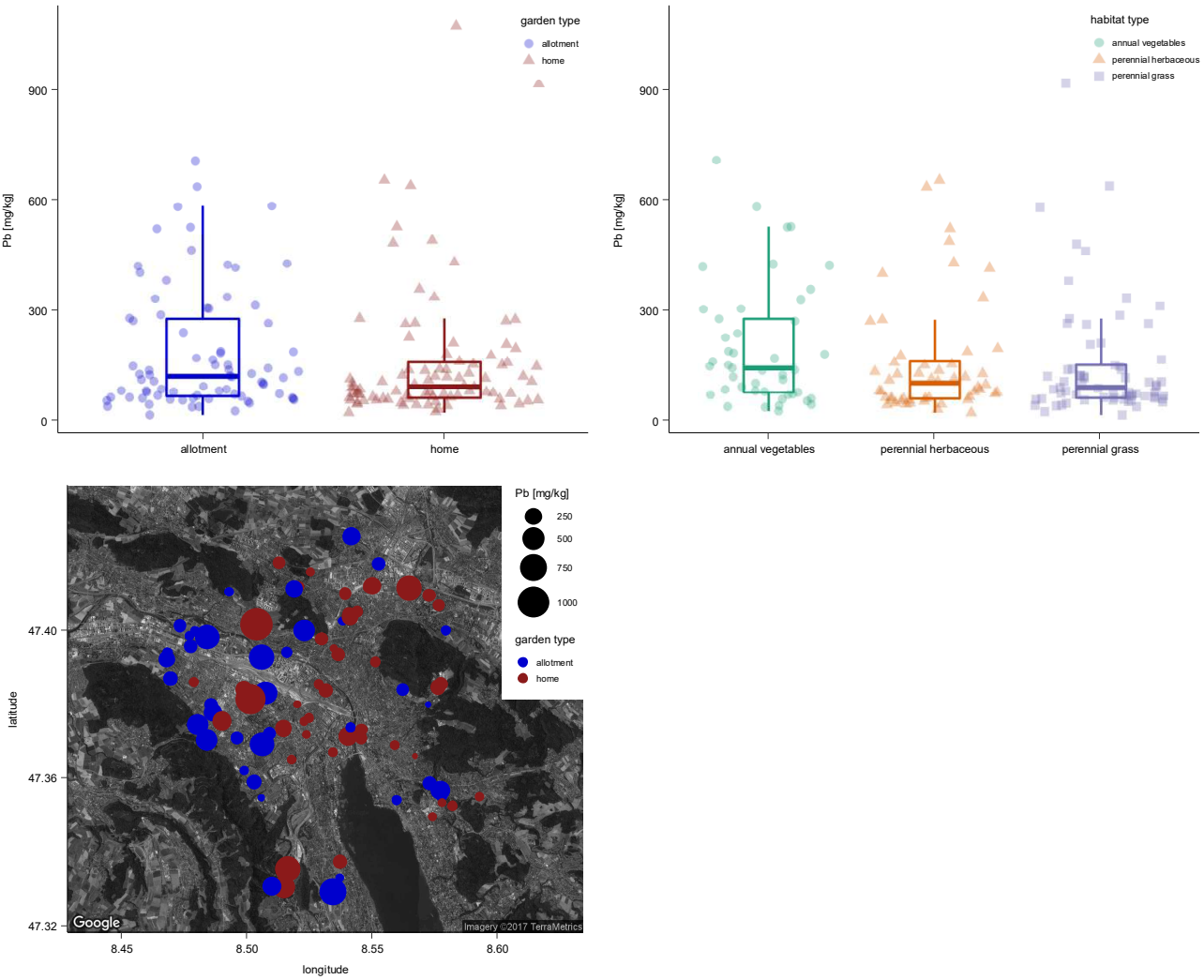


Figure S 3.6 – Distribution of Pb concentrations [mg/kg] measured with XRF technique. Boxplots on the left show concentrations per garden type and on the right per urban garden habitat type. Garden types: allotment N = 40, home = 44. Habitat types: perennial grass N = 70, perennial herbaceous N = 52, annual vegetables N = 46.

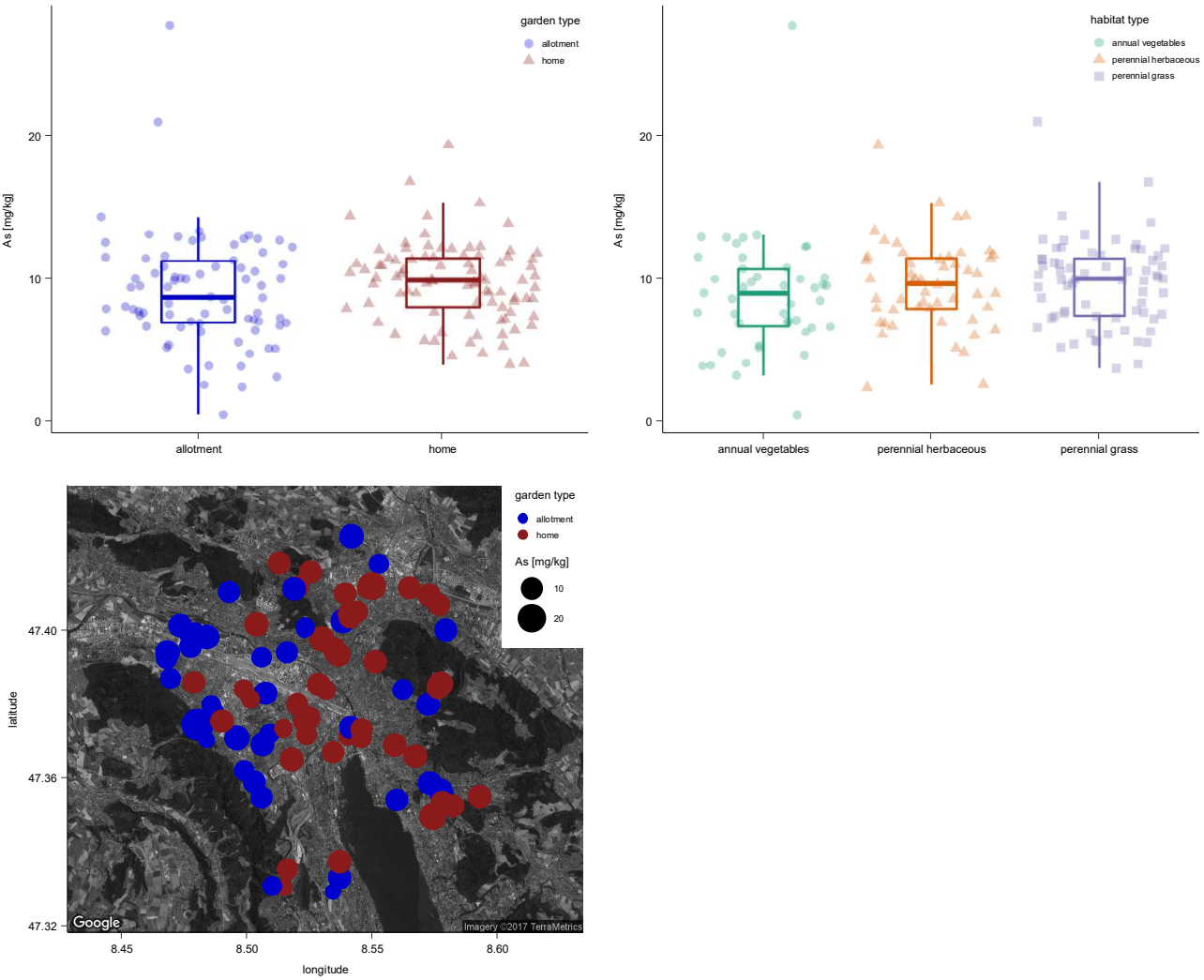


Figure S 3.7 – Distribution of As concentrations [mg/kg] measured with XRF technique. Boxplots on the left show concentrations per garden type and on the right per urban garden habitat type. Garden types: allotment N = 40, home = 44. Habitat types: perennial grass N = 70, perennial herbaceous N = 52, annual vegetables N = 46.

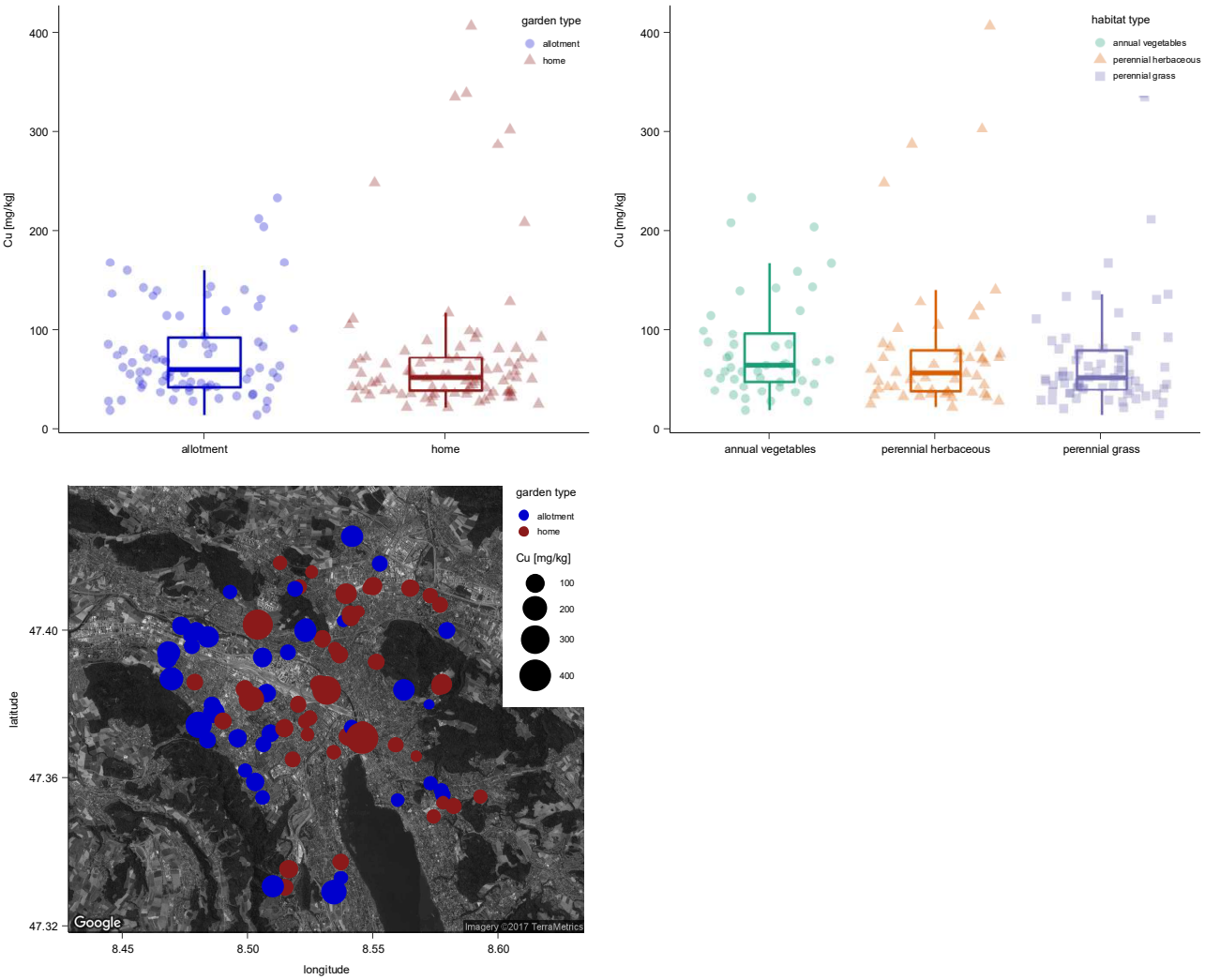


Figure S 3.8 – Distribution of Cu concentrations [mg/kg] measured with XRF technique. Boxplots on the left show concentrations per garden type and on the right per urban garden habitat type. Garden types: allotment N = 40, home = 44. Habitat types: perennial grass N = 70, perennial herbaceous N = 52, annual vegetables N = 46.

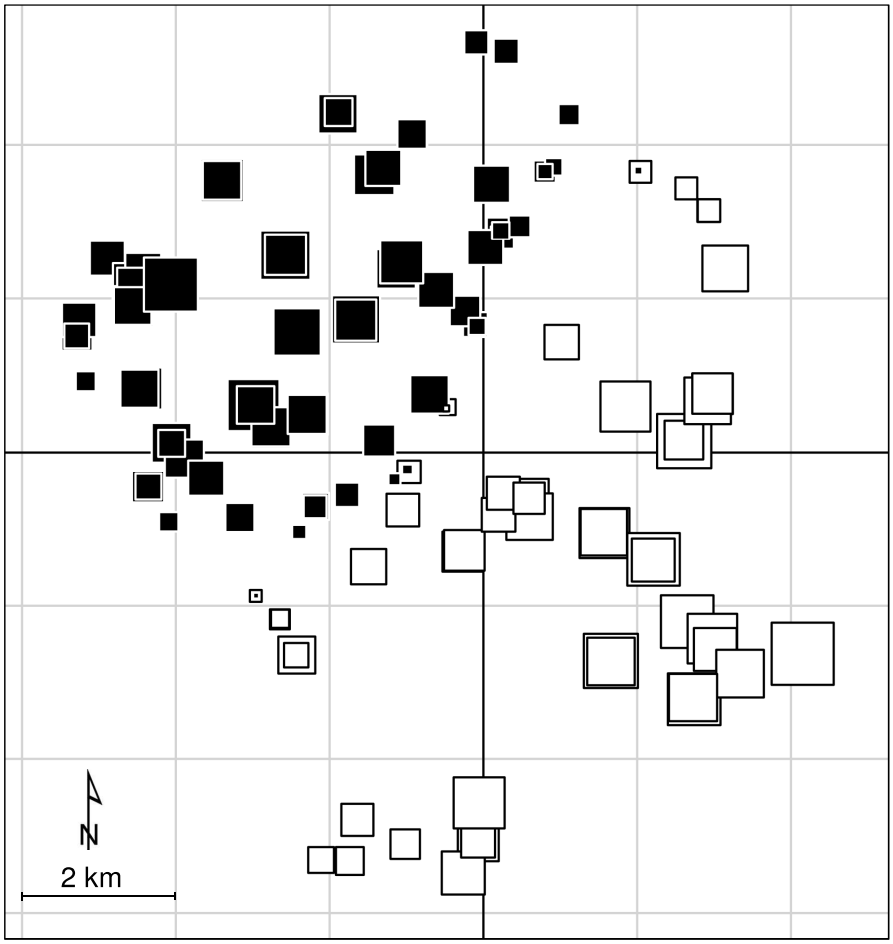


Figure S 3.9 – Moran eigenvector map (MEM) 2, which was a significant variable in the PERMANOVA with heavy metals as response variables (S 3.5). Increasing size of the symbols correspond to increasing positive values in black and increasing negative values in white, of the eigenvector (Hawkins et al. 2007).

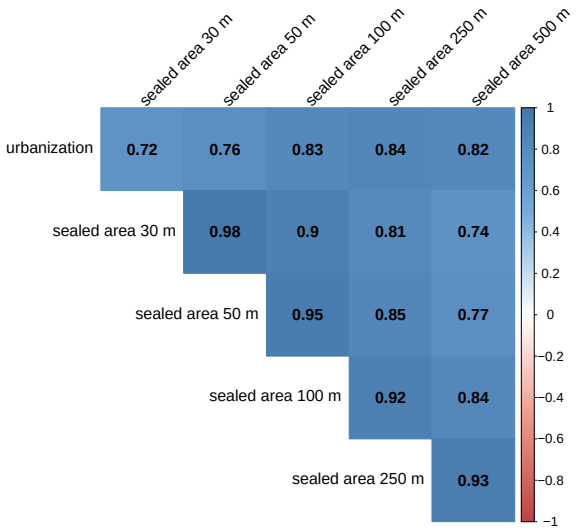


Figure S 3.10 – Pearson correlation matrix of sealed area around each garden (N = 85) and the variable 'overwarming', which stands for the urbanisation density in this study. 'Overwarming' is a measure of the local deviation of average night temperatures near surface in the city of Zurich. It is derived from a regional climate model by Parlow et al. (2010) and consists of six categories from 0 to + 6 K.

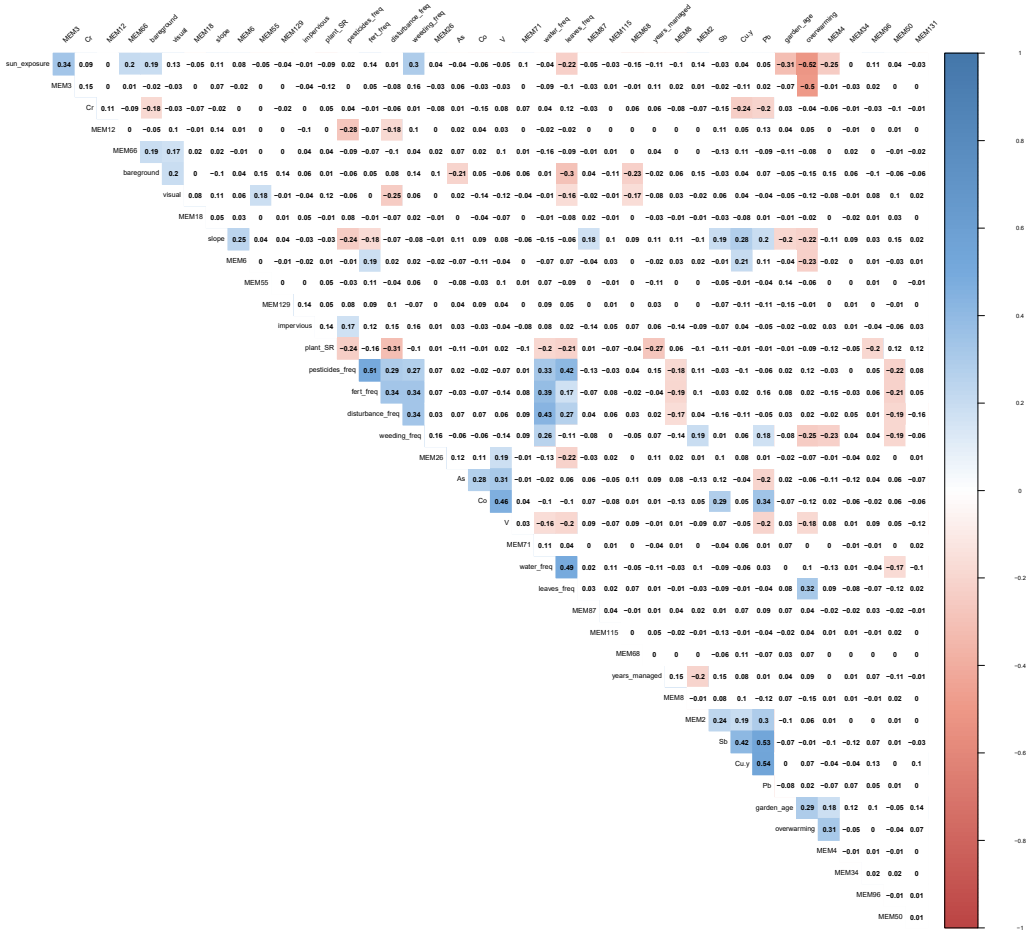


Figure S 3.11 – Pearson correlation matrix of all variables after variables selection approach, used for assessing its impact on the soil quality ordination. Variance inflation factor (VIF) < 3.9.

3.5.2 Supplementary Tables

Table S 3.1 – Goodness of fit statistic of soil quality measurements on NMDS ordination. Calculated with 10000 permutations. Values are ordered according to the pseudo squared correlation coefficient R^2 and printed in bold type if $p \leq 0.05$.

	R^2	P-value
Mg	0.63	$<$ **0.001**
P	0.49	$<$ **0.001**
C_{mic}	0.42	$<$ **0.001**
Fe	0.37	$<$ **0.001**
p_1159	0.32	$<$ **0.001**
K	0.31	$<$ **0.001**
pH	0.26	$<$ **0.001**
p_2930	0.21	$<$ **0.001**
SA	0.19	$<$ **0.001**
B	0.19	$<$ **0.001**
penetration	0.17	$<$ **0.001**
Mn	0.16	$<$ **0.001**
BD	0.13	$<$ **0.001**
resp	0.11	$<$ **0.001**
N_{min}	0.11	$<$ **0.001**
C_{org}	0.10	$<$ **0.001**
p_1620	0.10	$<$ **0.001**
p_1022	0.09	$<$ **0.001**
Cu	0.07	**0.01**
endogeic_SR	0.07	**0.01**
EC	0.06	**0.02**
anecic_M	0.05	**0.03**
depth	0.04	**0.05**
endogeic_M	0.04	**0.05**
WHC	0.03	0.16
anecic_SR	0.02	0.21
TON	0.01	0.64
clay	0.00	0.88

Table S 3.2 – Permutational multivariate analysis of variance (PERMANOVA) analysis on Euclidean distance matrix of soil quality measurements as response matrix and selected management and garden characteristic variables as predictors, calculated with 10000 permutations.

	Df	F Model	R^2	P	
Management					
visual	1	0.6	0.01	0.67	
pesticides	1	0.9	0.01	0.45	
pesticides_freq	1	6.4	0.03	< 0.001	***
compost	1	6	0.03	< 0.001	***
fert_freq	1	2.4	0.01	0.05	*
water	1	0.3	0.01	0.88	
water_freq	1	0.8	0.01	0.53	
disturbance	1	2.2	0.01	0.07	
disturbance_freq	1	1	0.01	0.4	
weeding_freq	1	0.9	0.01	0.45	
leaves_freq	1	2.2	0.01	0.07	
Garden characteristics					
overwarming	1	10.4	0.04	< 0.001	***
habitat	2	10.3	0.09	< 0.001	***
garden_type	1	7.6	0.03	< 0.001	***
slope	1	5.3	0.02	< 0.001	**
soil_texture	4	1.1	0.02	0.31	
bare soil	1	2.7	0.01	0.04	*
impervious	1	0.6	0.01	0.66	
plant_SR	1	1.5	0.01	0.21	
sun_exposure	1	1.5	0.01	0.2	
garden_age	1	1.4	0.01	0.23	
years_managed	1	1.7	0.01	0.14	
aspect	2	4.5	0.04	< 0.001	***
former_landuse	3	1.8	0.02	0.05	
history	2	3.2	0.03	< 0.001	**
Heavy metals					
Pb	1	1.7	0.01	0.14	
V	1	0.6	0.01	0.65	
Sb	1	0.9	0.01	0.44	
As	1	7.5	0.03	< 0.001	***
Co	1	1.4	0.01	0.22	
Cu	1	3	0.01	0.02	*
Spatial structure					
MEM2	1	1.2	0.01	0.29	
MEM3	1	2.4	0.01	0.06	
MEM4	1	3.1	0.01	0.02	*
MEM6	1	1	0.01	0.37	
MEM8	1	0.4	0.01	0.85	
MEM12	1	1.9	0.01	0.11	
MEM18	1	0.5	0.01	0.72	
MEM26	1	2	0.01	0.09	
MEM34	1	0.8	0.01	0.49	
MEM50	1	1.1	0.01	0.32	
MEM55	1	1.2	0.01	0.3	
MEM66	1	1.1	0.01	0.37	
MEM68	1	3.2	0.01	0.02	*
MEM71	1	3.2	0.01	0.02	*
MEM87	1	2	0.01	0.09	
MEM96	1	1.3	0.01	0.26	
MEM115	1	1.3	0.01	0.26	
MEM129	1	2.3	0.01	0.06	
MEM131	1	3.6	0.02	0.01	*
Residuals	86		0.36		

Table S 3.3 – Pearson correlation coefficients of selected garden management questions and soil quality measurements.

	Perennial herbaceous		**Annual vegetables**		**Perennial lawn**	
	FertFlowerFreshCompost		CropRotate		FstCutLawn	
	r	P	r	P	r	P
Physical						
clay	0.05	0.76	0.25	0.13	0.06	0.63
WHC	0.04	0.78	-0.07	0.68	-0.23	0.07
BD	-0.03	0.87	0.28	0.09	**0.28**	**0.03**
penetration	-0.2	0.2	**-0.65**	**<0.001**	0	1
depth	0.18	0.25	0.02	0.9	-0.17	0.19
SA	0.07	0.64	-0.32	0.05	-0.19	0.14
Chemical						
pH	-0.19	0.22	-0.05	0.78	0.02	0.87
EC	0.03	0.83	-0.31	0.06	-0.21	0.1
P	**0.31**	**0.04**	0.13	0.45	**-0.26**	**0.04**
K	0.22	0.16	-0.28	0.08	-0.13	0.33
Mg	-0.1	0.52	0.11	0.53	0.12	0.37
Fe	0	0.99	-0.24	0.14	**-0.29**	**0.02**
Cu	-0.17	0.28	**-0.39**	**0.02**	-0.03	0.83
Mn	0.29	0.06	0.29	0.08	-0.22	0.09
B	0.09	0.54	0.11	0.49	-0.21	0.1
Biological						
resp	0.09	0.58	**-0.46**	**0.01**	-0.07	0.57
C_{mic}	0.24	0.11	**-0.45**	**0.01**	-0.13	0.3
N_{min}	0.13	0.39	-0.17	0.31	-0.15	0.26
anecic_SR	0.08	0.61	0.16	0.35	0.02	0.88
anecic_M	**0.36**	**0.02**	-0.01	0.97	-0.1	0.46
endogeic_SR	-0.02	0.91	-0.08	0.65	-0.06	0.63
endogeic_M	-0.13	0.39	-0.24	0.15	0.02	0.88
SOM						
C_{org}	-0.07	0.64	-0.2	0.23	-0.01	0.94
TON	0.1	0.54	0.02	0.92	-0.21	0.11
p_1159	0.1	0.5	-0.1	0.56	-0.21	0.1
p_1620	-0.12	0.45	-0.31	0.06	0.19	0.13
p_1022	-0.03	0.84	**-0.36**	**0.03**	-0.06	0.62
p_2930	-0.21	0.17	**-0.42**	**0.01**	0.14	0.29

Table S 3.4 – Goodness of fit statistic of garden management questions fitted on a ordination of soil quality measurements (NMDS habitat). Calculated with 10000 permutations. Values are ordered according to the squared correlation coefficient R^2. Data set of N = 170 was divided into sites consisting of a) perennial lawn (N = 62), b) perennial herbaceous (N = 44) and c) annual vegetables (N = 39).

a) Garden habitat perennial lawn Garden management question	R^2	P
FstCutLawn	0.11	**0.04**
MowLawn	0.07	0.14
FlowerIslands	0.06	0.18
PestLawn	0.04	0.33
FertLawnLawn	0.03	0.38
FertLawn	0.02	0.48
FertLawnFert	0.02	0.53
CareLawn	0	0.87

b) Garden habitat perennial herbaceous Garden management question	R^2	P
PestFlower	0.12	**0.05**
FertFlowerFreshCompost	0.1	0.11
FertFlowerDung	0.1	0.13
WaterFlower	0.03	0.55
FertFlower	0.03	0.55
DiggingFlower	0.02	0.68
ForkFlower	0.02	0.64
FertFlowerFert	0.01	0.8
FertFlowerBiodyn	0.01	0.89

c) Garden habitat annual vegetables Garden management question	R^2	P
CropRotate	0.39	**<0.001**
WaterVeg	0.13	0.09
FertVegCompost	0.12	0.09
PestVeg	0.11	0.13
MixCult	0.11	0.13
FertVegFreshCompost	0.11	0.13
GreenFert	0.08	0.25
Mulch	0.07	0.26
ForkVeg	0.02	0.71
FertVegDung	0.02	0.65
FertVeg	0.01	0.83
DiggingVeg	0.01	0.8

Table S 3.5 – Permutational multivariate analysis of variance of heavy metals (PERMANOVA HM) as response matrix (Euclidean distance matrix) and selected garden characteristics and management variables see PERMANOVA in table S 3.2. Calculated with 10000 permutations.

	Df	F Model	R^2	P	
Management					
visual	1	0.7	0.01	0.49	
pesticides	1	1.7	0.01	0.18	
pesticides_freq	1	1.4	0.01	0.24	
compost	1	1.8	0.01	0.17	
fert_freq	1	4.5	0.02	0.02	*
water	1	2.4	0.01	0.1	
water_freq	1	0.3	0.01	0.75	
disturbance	1	0.5	0.01	0.58	
disturbance_freq	1	1.4	0.01	0.23	
weeding_freq	1	4.9	0.03	0.02	*
leaves_freq	1	6.5	0.04	0.01	**
Garden characteristics					
overwarming	1	2.3	0.01	0.1	
habitat	2	0.7	0.01	0.59	
garden_type	1	0.6	0.01	0.52	
slope	1	4.4	0.02	0.02	*
soil_texture	4	1.2	0.03	0.29	
bare soil	1	0.4	0.01	0.63	
impervious	1	0.5	0.01	0.59	
plant_SR	1	0.2	0.01	0.78	
sun_exposure	1	0.2	0.01	0.82	
garden_age	1	0.6	0.01	0.51	
years_managed	1	0.3	0.01	0.74	
aspect	2	0.7	0.01	0.58	
former_landuse	3	2.9	0.05	0.02	*
history	2	1.2	0.01	0.29	
Spatial structure					
MEM2	1	10.3	0.06	< 0.001	***
MEM3	1	1.7	0.01	0.18	
MEM4	1	0.3	0.01	0.76	
MEM6	1	0.2	0.01	0.76	
MEM8	1	0.4	0.01	0.64	
MEM12	1	1.5	0.01	0.22	
MEM18	1	0	0.01	0.97	
MEM26	1	1	0.01	0.36	
MEM34	1	4.3	0.02	0.05	
MEM50	1	0.8	0.01	0.43	
MEM55	1	0.4	0.01	0.65	
MEM66	1	2.7	0.02	0.07	
MEM68	1	2.8	0.02	0.07	
MEM71	1	0.4	0.01	0.62	
MEM87	1	0.4	0.01	0.62	
MEM96	1	1.5	0.01	0.22	
MEM115	1	0.2	0.01	0.77	
MEM129	1	3.9	0.02	0.05	
MEM131	1	0.7	0.01	0.5	
Residuals	92		0.51		

Table S 3.6 – Descriptive statistics of total heavy metal concentrations in urban garden sites (N = 145). Values are in mg kg^{-1}, measured on XRF device.

a) All garden sites N = 145

	mean	se	min	max	range
Sb	1.9	0.4	0.4	39.1	38.7
As	9.4	0.3	0.5	27.7	27.2
Co	31.6	0.4	18.3	45.4	27.1
Cu	77.3	4.9	15.6	339.7	324.1
Pb	179.1	15.0	18.5	1076.0	1057.5
V	79.4	1.3	44.1	117.9	73.8

b) Perennial lawn N = 62

	mean	se	min	max	range
Sb	1.8	0.6	0.4	39.1	38.7
As	9.6	0.4	3.7	21.0	17.3
Co	31.0	0.6	21.4	43.8	22.4
Cu	72.6	7.8	15.6	339.7	324.1
Pb	175.0	25.7	18.5	1076.0	1057.5
V	79.9	2.0	44.1	117.9	73.8

c) Perennial herbaceous N = 46

	mean	se	min	max	range
Sb	2.5	0.9	0.4	33.0	32.6
As	8.9	0.7	0.5	27.7	27.2
Co	31.3	0.7	21.2	40.6	19.4
Cu	85.0	9.0	20.1	233.9	213.8
Pb	195.5	27.5	28.9	709.8	680.9
V	75.7	2.2	50.6	105.6	55.0

d) Annual vegetables N = 37

	mean	se	min	max	range
Sb	1.6	0.3	0.5	11.9	11.4
As	9.5	0.5	2.4	19.4	17.0
Co	32.6	0.7	18.3	45.4	27.1
Cu	77.4	9.0	23.1	303.1	280.0
Pb	171.5	24.0	24.4	656.3	631.9
V	81.7	2.3	51.9	112.2	60.3

e) Allotment gardens N = 69

	mean	se	min	max	range
Sb	1.6	0.2	0.4	9.8	9.4
As	9.1	0.5	0.5	27.7	27.2
Co	30.8	0.6	21.2	40.6	19.4
Cu	79.2	6.0	15.6	233.9	218.3
Pb	185.0	20.0	18.5	709.8	691.3
V	77.1	1.8	44.1	116.4	72.3

f) Home gardens N = 76

	mean	se	min	max	range
Sb	2.2	0.7	0.4	39.1	38.7
As	9.7	0.3	4.0	19.4	15.4
Co	32.3	0.5	18.3	45.4	27.1
Cu	75.6	7.8	22.7	339.7	317.0
Pb	173.8	22.2	24.4	1076.0	1051.6
V	81.5	1.7	51.9	117.9	66.0

Table S 3.7 – a) Pearson correlation of heavy metal (HM) variables and selected indices of soil quality as well as variables from PERMANOVA (S 3.5; p <0.05).Rooibos and green tea values are mean decomposition values measured with the TBI (tea bag index). b) Pearson correlation of heavy metal (HM) variables and earthwors including ecological groups. SR = species richness, M = earthworm biomass, r = pearson correaltion coefficient.

a)	C_{mic}/C_{org}		qCO_2		C_{org}/clay		rooibos_tea		green_tea		fert_freq		MEM2		slope	
	r	p	r	p	r	p	r	p	r	p	r	p	r	p	r	p
Sb	-0.02	0.81	0.03	0.75	0.00	1.00	0.11	0.20	-0.02	0.83	-0.03	0.75	0.24	**0.01**	0.19	**0.02**
As	-0.10	0.22	0.05	0.54	0.03	0.69	-0.07	0.41	-0.06	0.45	-0.03	0.76	-0.13	0.13	0.11	0.2
Co	-0.02	0.81	0.02	0.77	0.00	0.96	-0.04	0.62	-0.12	0.17	-0.07	0.39	0.05	0.54	0.09	0.27
Cu	-0.01	0.89	0.07	0.40	-0.00	1.00	-0.00	0.96	0.01	0.88	0.02	0.81	0.19	**0.02**	0.28	**0.01**
Pb	0.02	0.79	0.09	0.28	-0.02	0.78	0.02	0.77	-0.05	0.57	0.16	**0.05**	0.30	**0.01**	0.20	**0.02**
V	0.11	0.20	-0.16	0.06	-0.14	0.09	-0.04	0.62	-0.07	0.44	-0.14	0.09	-0.09	0.29	0.08	0.33

b)	earthworms_SR		earthworms_M		endogeic_SR		endogeic_M		anecic_SR		anecic_M		epigeic_SR		epigeic_M	
	r	p	r	p	r	p	r	p	r	p	r	p	r	p	r	p
Sb	-0.05	0.53	-0.04	0.62	-0.04	0.62	0.02	0.81	0.02	0.81	-0.06	0.48	-0.27	0.20	-0.12	0.59
As	0.07	0.41	0.07	0.44	0.07	0.44	0.13	0.12	0.13	0.12	0.02	0.84	-0.11	0.62	0.15	0.48
Co	-0.12	0.14	-0.15	0.08	-0.15	0.08	-0.10	0.23	-0.10	0.23	-0.13	0.11	-0.16	0.45	0.10	0.63
Cu	-0.09	0.30	-0.05	0.56	-0.05	0.56	-0.04	0.62	-0.04	0.62	-0.04	0.67	-0.40	**0.05**	-0.20	0.35
Pb	-0.15	0.08	-0.09	0.30	-0.09	0.30	-0.07	0.38	-0.07	0.38	-0.07	0.41	-0.29	0.18	0.04	0.87
V	0.05	0.51	-0.02	0.81	-0.02	0.81	-0.07	0.43	-0.07	0.43	0.01	0.92	-0.08	0.72	-0.19	0.38

Table S 3.8 – Pearson correlation of community composition of earthworms and variables from PERMANOVA (S 3.2; p <0.05).SR = species richness, M = earthworm biomass, r = pearson correaltion coefficient, η^2 = Effect size for categorical variables see Bakeman (2005).

a) Management	pesticides_freq		compost		fert_freq	
	r	p	η^2	p	r	p
anecic_SR	0.12	0.15	0.00	0.58	0.04	0.60
anecic_M	0.01	0.95	0.01	0.38	0.09	0.28
endogeic_SR	0.02	0.82	0.03	**0.04**	0.09	0.28
endogeic_M	0.06	0.44	0.02	0.06	0.14	0.10
epigeic_mass	-0.12	0.58	0.05	0.31	0.13	0.54
epigeic_SR	-0.15	0.47	0.02	0.46	-0.26	0.21
EW_SR	0.08	0.33	0.02	0.10	0.09	0.29
EW_mass	0.03	0.71	0.02	0.14	0.13	0.12

b) Garden	overwarming		habitat		garden_type		slope		bare soil		aspect		history	
characteristics	r	p	η^2	p	η^2	p	r	p	r	p	η^2	p	η^2	p
anecic_SR	-0.05	0.57	0.05	**0.03**	0.01	0.36	-0.06	0.45	0.14	0.09	0.00	0.99	0.00	0.72
anecic_M	-0.05	0.56	0.01	0.55	0.00	0.73	-0.10	0.24	-0.02	0.8	0.01	0.49	0.00	0.77
endogeic_SR	-0.14	0.08	0.06	**0.01**	0.02	0.10	-0.00	0.98	0.18	**0.03**	0.02	0.24	0.03	0.15
endogeic_M	-0.07	0.41	0.12	< **0.001**	0.01	0.19	-0.08	0.34	0.24	< **0.001**	0.00	0.90	0.05	**0.02**
epigeic_mass	-0.12	0.59	0.09	0.39	0.03	0.42	0.24	0.25	-0.27	0.2	0.01	0.68	0.17	0.14
epigeic_SR	0.08	0.70	0.07	0.47	0.00	0.93	0.04	0.84	-0.04	0.84	0.06	0.24	0.02	0.83
EW_SR	-0.10	0.22	0.06	**0.02**	0.02	0.10	-0.05	0.54	0.20	**0.02**	0.01	0.47	0.01	0.39
EW_mass	-0.07	0.44	0.02	0.31	0.00	0.80	-0.11	0.18	0.07	0.38	0.01	0.67	0.02	0.24

c) Spatial	MEM4		MEM68		MEM71		MEM131	
structure	r	p	r	p	r	p	r	p
anecic_SR	0.17	**0.05**	0.09	0.29	0.14	0.11	0.11	0.19
anecic_M	0.03	0.71	0.02	0.85	0.02	0.84	0.07	0.42
endogeic_SR	-0.08	0.36	-0.07	0.38	-0.03	0.69	-0.09	0.27
endogeic_M	0.03	0.75	-0.11	0.19	-0.00	1.00	-0.05	0.59
epigeic_mass	0.07	0.76	0.26	0.22	-0.09	0.67	0.12	0.59
epigeic_SR	0.24	0.25	0.38	0.06	-0.36	0.09	0.23	0.27
EW_SR	0.04	0.63	0.03	0.75	0.02	0.84	0.00	0.95
EW_mass	0.04	0.66	-0.03	0.76	0.01	0.89	0.04	0.65

d) SQ indices	C_{mic}/C_{org}		qCO_2		C_{org}/clay		rooibos_tea		green_tea	
& TBI	r	p	r	p	r	p	r	p	r	p
anecic_SR	0.09	0.27	0.01	0.88	-0.01	0.89	-0.03	0.76	-0.14	0.10
anecic_M	0.13	0.13	-0.01	0.93	-0.04	0.61	0.15	0.08	-0.03	0.71
endogeic_SR	-0.10	0.23	0.22	**0.01**	0.01	0.93	0.10	0.23	-0.03	0.77
endogeic_M	-0.11	0.17	0.25	**0.01**	0.10	0.25	0.11	0.21	-0.05	0.54
epigeic_mass	0.15	0.50	0.09	0.67	0.04	0.84	0.20	0.35	0.17	0.44
epigeic_SR	-0.39	0.06	-0.20	0.34	0.34	0.10	-0.08	0.71	-0.16	0.47
EW_SR	-0.05	0.55	0.17	**0.04**	0.02	0.79	0.09	0.32	-0.09	0.28
EW_mass	0.06	0.49	0.09	0.27	0.00	0.96	0.17	**0.05**	-0.04	0.61

Table S 3.9 – Pearson correlation of tea bag index and soil quality indices versus variables from PERMANOVA (S 3.2; p <0.05).SR = species richness, M = earthworm biomass, r = pearson correaltion coefficient, η^2 = Effect size for categorical variables see Bakeman (2005).

a) Management	pesticides_freq		compost		fert_freq	
	r	p	η^2	p	r	p
C_{mic}/C_{org}	0.13	0.12	0.01	0.19	0.08	0.36
qCO_2	0.04	0.59	0.05	<**0.001**	0.09	0.28
C_{org}/clay	0.01	0.94	0.00	0.59	0.13	0.11
rooibos_tea	-0.02	0.85	0.01	0.33	0.16	0.06
green_tea	0.07	0.44	0.00	0.94	0.10	0.24

b) Garden	overwarming		habitat		garden_type		slope		bare soil		aspect		history	
characteristics	r	p	η^2	p	η^2	p	r	p	r	p	η^2	p	η^2	p
C_{mic}/C_{org}	-0.11	0.19	0.16	<**0.001**	0.06	<**0.001**	-0.04	0.67	-0.35	<**0.001**	0.02	0.22	0.00	0.91
qCO_2	0.05	0.59	0.15	<**0.001**	0.07	<**0.001**	-0.05	0.52	0.36	<**0.001**	0.01	0.59	0.05	**0.02**
C_{org}/clay	0.25	<**0.001**	0.04	0.07	0.00	0.85	-0.07	0.39	0.09	0.27	0.03	0.08	0.04	0.07
rooibos_tea	0.11	0.21	0.00	0.88	0.01	0.18	0.04	0.61	-0.07	0.43	0.00	0.89	0.01	0.58
green_tea	0.20	**0.02**	0.05	**0.03**	0.00	0.58	-0.13	0.13	-0.17	**0.05**	0.07	**0.01**	0.04	0.06

c) Spatial	MEM4		MEM68		MEM71		MEM131	
structure	r	p	r	p	r	p	r	p
C_{mic}/C_{org}	-0.12	0.15	0.15	0.07	0.21	0.01	-0.06	0.51
qCO_2	-0.05	0.52	-0.20	**0.01**	-0.03	0.71	0.18	**0.03**
C_{org}/clay	0.17	**0.05**	0.05	0.57	-0.06	0.51	0.21	**0.01**
rooibos_tea	0.06	0.49	-0.08	0.36	-0.08	0.38	0.15	0.09
green_tea	0.03	0.69	-0.02	0.83	-0.03	0.76	-0.01	0.95

d) SQ indices	C_{mic}/C_{org}		qCO_2		C_{org}/clay		rooibos_tea		green_tea	
& TBI	r	p	r	p	r	p	r	p	r	p
C_{mic}/C_{org}	1.00	<**0.001**	-0.42	<**0.001**	-0.50	<**0.001**	-0.03	0.75	0.09	0.29
qCO_2	-0.42	<**0.001**	1.00	<**0.001**	0.24	<**0.001**	0.15	0.07	0.11	0.22
C_{org}/clay	-0.50	<**0.001**	0.24	<**0.001**	1.00	<**0.001**	0.06	0.46	0.12	0.15
rooibos_tea	-0.03	0.75	0.15	0.07	0.06	0.46	1.00	<**0.001**	0.03	0.70
green_tea	0.09	0.29	0.11	0.22	0.12	0.15	0.03	0.70	1.00	<**0.001**

Table S 3.10 – Assessment of urban garden management questions. Questions were originally asked in German for the 85 participating gardeners.

PestLawn How often do you use pesticeds, fungicides or herbicides to protect your lawn? Never (1) Less than once per year (2) 1 to 3 times per year (3) 4 to 10 times per year (4) More than 10 times per year (5)	**PestFlower** How often do you use pesticeds, fungicides or herbicides (without slug pellets) to protect your flowers? Never (1) Less than once per year (2) 1 to 3 times per year (3) 4 to 10 times per year (4) More than 10 times per year (5)	**PestVeg** How often do you use pesticeds, fungicides or herbicides (without slug pellets) to protect your vegetables? Never (1) Less than once per year (2) 1 to 3 times per year (3) 4 to 10 times per year (4) More than 10 times per year (5)
FertLawnCompost Do you use compost or plant slurry for your lawn? (0/1)	**FertVegCompost** Do you use compost or plant slurry for your vegetables? (0/1)	**FertVegFreshCompost** Do you use fresh compost (less than 1 year old) or plant slurry for your vegetables? (0/1)
FertLawn How often do you use fertilizers for your lawn? Never (0) 4 to 5 years (1) 2 to 3 years (2) once a year (3) More than once a year (4)	**FertVeg** How often do you use fertilizers for your vegetables? Never (0) 2 to 3 years (1) Once a year (2) 2 to 3 per year (3) More than three per year (4)	**FertFlower** How often do you use fertilizers for your flowers? Never (0) 2 to 3 years (1) Once a year (2) 2 to 3 per year (3) More than three per year (4)
Weeds How often do you remove most of the weeds in your garden? Never (1) Rarely (2) Sometimes (3) Often (4) Very often (5)	**PestTrees** How often do you use insecticides, fungicides or herbicides to protect our trees and shrubs? Never (1) Less than once a year (2) 1 to 3 times per year (3) 4 to 10 times per year (4) More than 10 times per year (5)	**Leaves** How often do you remove most of the leaves in your garden? Never (1) Spring (2) Autumn (3) Every 2 to 3 weeks (4) Weekly in autumn (5)
FertFlowerCompost Do you use compost or plant slurry for your flowers? (0/1)	**FertFlowerFreshCompost** Do you use fresh compost (less than 1 year old) or plant slurry for your flowers? (0/1)	**WeedingHerbicide** Do you use commercial herbicides? (0/1)

WaterLawn
How often do you water your lawn?

Never (1)
When dry (2)
once a week (3)
twice a week (4)
More than twice a week (5)

CareLawn
How often do you scarify your lawn (including reseeding)
Never (1)
6 to 10 years (2)
4 to 5 years (3)
2 to 3 years (4)
Annually (5)

DrySticks
Do you leave withered flowers and sticks during the winter in your garden?
Always (1)
Mostly (2)
Sometimes (3)
Rarely (4)
Never (5)

MowLawn
How often do you mow your lawn?
1 to 2 (1)
3 to 4 (2)
5 to 8 (3)
9 to 20 (4)
over 20 (5)

WaterVeg
How often do you water your vegetable beds?
Never (1)
When dry (2)
once a week (3)
twice a week (4)
More than twice a week (5)

DiggingFlower
How often do you till your soil in the flower beds?
Never (1)
3 years or less (2)
Every two years (3)
Once per year (4)
More than once per year (5)

FstCutLawn
When is the first time point of cutting your lawn?

April (1)
May (2)
Start June (3)
End June (4)
After June (5)

WaterFlower
How often do you water your flower beds?
Never (1)
When dry (2)
once a week (3)
twice a week (4)
More than twice a week (5)

DiggingVeg
How often do you till your soil in the vegetable beds?
Never (1)
3 years or less (2)
Every two years (3)
Once per year (4)
More than once per year (5)

Mulch
Do you use organic material (mulch) to cover your vegetable beds?

Never (1)
Rarely (2)
Sometimes (3)
Mostly (4)
Always (5)

Table S 3.11 – Least square means calculation from linear mixed effect models. Fixed effects are garden type (allotment and home), habitat type (annual vegetables, perennial herbaceous, perennial grass) and clustering classification (1-3). Garden identity was set as random factor and significance level alpha = 0.05.

	home			allotment			grass			herbaceous			vegetables			clustering 1			clustering 2			clustering 3		
	lsmean	se		lsmean	se		lsmean	se		lsmean	se		lsmean	se		lsmean	se		lsmean	se		lsmean	se	
Physical																								
clay [%]	23.77	0.78	a	24.20	0.79	a	24.64	0.74	a	23.17	0.79	a	24.16	0.87	a	23.16	0.82	a	24.51	0.84	a	24.28	0.94	a
WHC [%]	0.80	0.02	a	0.83	0.02	a	0.81	0.02	a	0.82	0.02	a	0.81	0.02	a	0.83	0.02	a	0.83	0.02	a	0.78	0.02	a
BD $[gcm^{-3}]$	1.09	0.02	a	1.09	0.02	a	1.02	0.02	a	1.11	0.02	b	1.14	0.02	b	1.05	0.02	a	1.08	0.02	ab	1.14	0.02	b
penetration $[MPa]$	1.50	0.07	b	1.26	0.07	a	1.68	0.07	c	1.40	0.07	b	1.06	0.08	a	1.41	0.07	a	1.43	0.07	a	1.31	0.08	a
depth $[cm]$	42.16	2.72	a	46.89	2.79	a	42.24	2.38	a	42.93	2.51	a	48.41	2.68	a	42.54	2.62	a	44.30	2.68	a	46.75	3.00	a
SA [%]	83.62	1.16	a	81.07	1.17	a	85.11	1.16	b	81.47	1.26	ab	80.47	1.40	a	85.68	1.27	b	84.29	1.32	b	77.07	1.45	a
Chemical																								
pH	7.32	0.03	b	7.21	0.03	a	7.22	0.03	a	7.29	0.03	a	7.28	0.04	a	7.26	0.03	ab	7.17	0.04	a	7.36	0.04	b
EC $[\mu Scm^{-1}]$	189.90	5.19	a	177.45	5.19	a	182.04	5.26	a	184.24	5.73	a	184.74	6.39	a	196.04	5.73	b	182.46	5.96	ab	172.52	6.51	a
P $[mgkg^{-1}]$	183.60	12.09	a	201.54	12.18	a	157.14	11.82	a	177.96	12.76	a	242.61	14.08	b	140.37	12.97	a	267.65	13.43	b	169.68	14.82	a
K $[mgkg^{-1}]$	174.09	14.63	a	174.16	14.49	a	146.65	15.47	a	166.60	16.98	ab	209.12	19.15	b	122.85	16.58	a	230.26	17.37	b	169.26	18.74	a
Mg $[mgkg^{-1}]$	532.93	20.97	a	526.28	21.35	a	511.03	19.34	a	533.24	20.61	a	544.54	22.32	a	472.47	21.34	a	469.17	21.94	a	647.18	24.45	b
Fe $[mgkg^{-1}]$	367.15	13.99	a	372.85	14.26	a	349.15	12.86	a	370.04	13.69	ab	390.81	14.82	b	356.94	14.19	a	423.72	14.58	b	329.34	16.26	a
Cu $[mgkg^{-1}]$	27.02	4.34	a	39.18	4.53	a	29.68	3.32	a	33.91	3.38	ab	35.69	3.45	b	34.29	3.47	ab	37.09	3.50	b	27.91	3.74	a
Mn $[mgkg^{-1}]$	289.60	15.73	a	290.25	16.18	a	297.26	13.58	a	277.52	14.25	a	295.00	15.12	a	307.46	14.89	b	311.17	15.20	b	251.14	16.97	a
B $[mgkg^{-1}]$	1.34	0.09	a	1.52	0.09	a	1.14	0.08	a	1.39	0.09	b	1.76	0.10	c	1.25	0.09	a	1.64	0.10	b	1.40	0.11	ab
Biological																								
basal respiration $[\mu gCO_2 - Cg^{-1}h^{-1}]$	0.25	0.01	a	0.24	0.01	a	0.22	0.01	a	0.24	0.02	ab	0.27	0.02	b	0.24	0.01	a	0.30	0.02	b	0.20	0.02	a
C_{mic} $[mgkg^{-1}]$	807.17	30.81	a	761.94	31.03	a	806.43	30.21	a	778.02	32.64	a	769.21	36.03	a	960.21	33.14	c	780.52	34.32	b	612.93	37.85	a
N_{mic} $[mgkg^{-1}]$	140.66	6.02	a	131.43	6.07	a	141.31	5.84	a	136.74	6.29	a	130.09	6.93	a	172.69	6.42	c	132.60	6.64	b	102.84	7.34	a
N_{min} $[mgkg^{-1}]$	1.65	0.15	a	1.85	0.15	a	1.68	0.15	a	1.82	0.16	a	1.75	0.18	a	1.93	0.17	b	2.02	0.17	b	1.30	0.19	a
anecic_SR $[ind.m^{-2}]$	1.17	0.12	a	1.38	0.12	a	0.97	0.12	a	1.48	0.13	b	1.38	0.15	ab	1.50	0.13	a	1.25	0.13	a	1.08	0.15	a
anecic_M $[gm^{-2}]$	7.81	0.84	a	7.91	0.84	a	7.15	0.88	a	8.72	0.96	a	7.70	1.07	a	8.73	0.94	a	8.22	0.99	a	6.63	1.07	a
endogeic_SR $[ind.m^{-2}]$	1.74	0.17	a	1.89	0.17	a	1.55	0.17	a	1.70	0.18	ab	2.19	0.21	b	1.63	0.18	a	1.90	0.19	a	1.90	0.21	a
endogeic_M $[gm^{-2}]$	3.57	0.38	a	3.49	0.37	a	2.69	0.39	a	2.79	0.43	a	5.10	0.49	b	3.17	0.42	a	4.33	0.44	a	3.10	0.48	a
epigeic_SR $[ind.m^{-2}]$	0.71	0.17	a	0.86	0.17	a	0.72	0.15	a	1.07	0.22	a	0.56	0.19	a	0.56	0.18	a	0.56	0.15	a	1.23	0.25	a
epigeic_M $[gm^{-2}]$	2.57	1.53	a	5.59	1.56	a	4.36	1.44	a	6.45	2.06	a	1.44	1.83	a	3.11	1.70	a	3.66	1.42	a	5.47	2.41	a
earthworm_SR $[ind.m^{-2}]$	3.01	0.23	a	3.37	0.24	a	2.63	0.24	a	3.30	0.26	ab	3.64	0.29	b	3.21	0.26	a	3.27	0.27	a	3.09	0.29	a
earthworm_M $[gm^{-2}]$	126.80	11.45	a	127.38	11.38	a	109.90	11.90	a	128.87	13.01	a	142.50	14.61	a	132.98	12.84	a	139.96	13.41	a	108.32	14.55	a
earthworm_abundance $[gm^{-2}]$	227.23	22.84	a	250.65	22.64	a	199.95	24.12	a	190.58	26.47	a	326.29	29.85	b	221.23	25.88	a	266.64	27.10	a	228.95	29.25	a
SOM																								
C_{org} [%]	4.75	0.23	a	4.85	0.24	a	4.55	0.21	a	4.93	0.23	a	4.92	0.25	a	4.42	0.24	a	4.86	0.24	a	5.13	0.27	a
TON [%]	0.33	0.02	a	0.35	0.02	a	0.32	0.01	a	0.33	0.02	a	0.36	0.02	a	0.34	0.02	a	0.36	0.02	a	0.31	0.02	a
p_1620 $[A.U.cm^{-1}]$	0.34	0.01	a	0.32	0.01	a	0.35	0.01	a	0.34	0.01	a	0.31	0.01	a	0.31	0.01	a	0.31	0.01	a	0.37	0.01	b
p_1022 $[A.U.cm^{-1}]$	1.16	0.05	a	1.24	0.05	a	1.35	0.05	b	1.25	0.06	b	1.00	0.06	a	1.20	0.06	a	1.26	0.06	a	1.14	0.06	a
p_2930 $[A.U.cm^{-1}]$	0.94	0.04	a	0.89	0.04	a	0.90	0.04	a	0.96	0.04	a	0.88	0.05	a	0.82	0.04	a	0.88	0.04	a	1.04	0.05	b
Soil quality indices																								
C_{mic}/C_{org} [%]	1.87	0.09	a	1.72	0.09	a	2.00	0.08	b	1.72	0.09	a	1.67	0.10	a	2.31	0.09	c	1.73	0.09	b	1.35	0.10	a
qCO_2 $[\mu gCO_2 - Cg^{-1}h^{-1}C_{mic}^{-1}h^{-1}]$	0.31	0.01	a	0.33	0.01	a	0.29	0.01	a	0.31	0.01	ab	0.36	0.02	b	0.25	0.01	a	0.39	0.01	c	0.32	0.02	b
C_{org}/clay	0.21	0.01	a	0.21	0.01	a	0.19	0.01	a	0.22	0.01	a	0.22	0.01	a	0.20	0.01	a	0.21	0.01	a	0.23	0.02	a
TBI																								
green tea decomposition	0.59	0.01	a	0.59	0.01	a	0.60	0.01	a	0.58	0.01	a	0.58	0.01	a	0.59	0.01	a	0.58	0.01	a	0.58	0.01	a
rooibos tea decomposition	0.30	0.00	a	0.29	0.00	a	0.30	0.00	a	0.30	0.00	a	0.29	0.01	a	0.29	0.00	a	0.30	0.00	a	0.30	0.01	a
Heavy metals																								
Sb $[mgkg^{-1}]$	2.30	0.61	a	1.52	0.62	a	1.84	0.60	a	1.18	0.65	a	2.70	0.72	a	2.01	0.66	a	2.26	0.68	a	1.45	0.75	a
As $[mgkg^{-1}]$	9.75	0.52	a	9.10	0.54	a	9.84	0.43	a	9.23	0.45	a	9.20	0.47	a	9.21	0.47	a	9.58	0.48	a	9.49	0.53	a
Co $[mgkg^{-1}]$	32.53	0.65	a	30.87	0.65	a	31.30	0.64	a	32.26	0.69	a	31.55	0.77	a	31.04	0.70	a	31.91	0.73	a	32.16	0.80	a
Cu $[mgkg^{-1}]$	79.33	9.22	a	76.84	9.60	a	73.20	7.26	a	76.22	7.44	a	84.83	7.66	a	76.70	7.70	a	78.00	7.79	a	79.56	8.47	a
Pb $[mgkg^{-1}]$	171.94	26.30	a	179.84	26.94	a	173.82	23.40	a	169.34	24.73	a	184.51	26.48	a	175.39	25.77	a	178.21	26.40	a	174.08	29.49	a
V $[mgkg^{-1}]$	81.30	2.20	a	77.37	2.25	a	81.33	1.96	a	79.79	2.07	a	76.89	2.22	a	77.83	2.16	a	80.43	2.21	a	79.76	2.47	a

Table S 3.12 – Description of CO_2 flux measurements

Soil respiration was measured after pre-incubating 30 g of soil (dry matter equivalent) for 7 d at 20°C in 100 ml wide-neck DURAN glass bottles (Schott AG, Mainz, Germany). All samples of one series were kept in moist plastic boxes to avoid drying. Bottles were left open to allow soil aeration and only closed during CO_2 flux measurements. Soil moisture was readjusted if necessary by adding H_2O_{demin} until 40-50 % water holding capacity was reached. After re-wetting, we waited 24 hours until CO_2 flux measurements were performed. Bottles were gently vented with ambient air to produce defined initial conditions in the head-space, prior to each measurement. For CO_2 measurement, the bottles were sealed with a lid containing a rubber septum and placed on an auto sampler (MPS 2XL, Gerstel AG, Sursee, Switzerland), equipped with a temperature controlled tray at 20 ± 0.8°C. To avoid partial vacuum, 5 ml of He was injected into the head-space prior to each gas sampling. CO_2 concentrations in the head-space were analysed for by a gas chromatograph (7890A, Agilent Technologies, Santa Clara, CA) equipped with a flame ionization detector (FID). CO_2 calibration curves (R^2 >0.99) were obtained by a threefold analysis of 3 standard gases (300, 2960 and 9000 ppm CO_2) before and after each measurement cycle. CO_2 flux calculations were based on the increase in CO_2 concentration in the head-space over 6 hours. The linearity of the enrichment was tested according to Krause et al. (2017).

Table S 3.13 – Description of soil organic matter characterisation

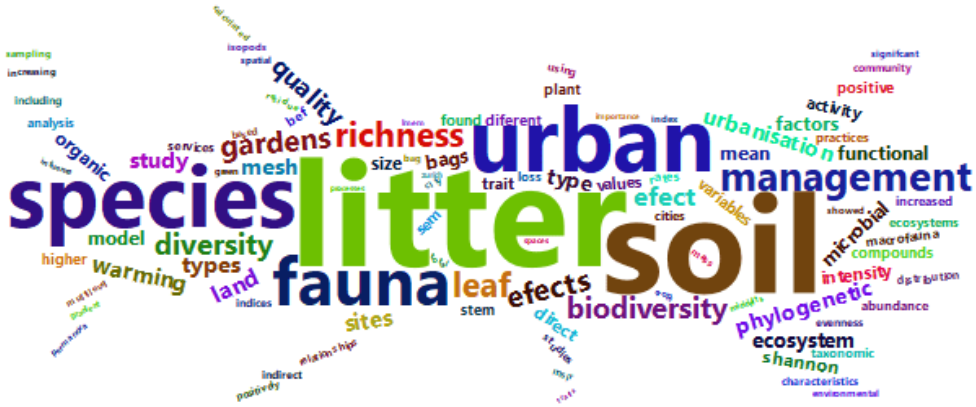
The biochemical composition and quality of soil organic matter (SOM) was characterized by diffuse reflectance Fourier transform mid-infrared spectroscopy (DRIFTS). Mid-infrared spectra of dried and ball milled soil samples were recorded according to Rasche et al. (2013). Each soil sample was analysed in triplicate from wavelengths 3950 to 650 cm-1 using solid, undiluted KBr as background. The obtained spectra were baseline corrected (OPUS version 7.0 Bruker Optik GmbH). Main functional groups of SOM compounds were calculated by peak area integration of particular spectral frequencies according to Rasche et al. (2014) & Demyan et al. (2012).

Table S 3.14 – Description of R packages used for various calculations.

Calculation	Description	Function	Package	Reference
Delaunay triangulation	Spatial matrix, representing the connections between the urban gardens	tri2nb	spdep	Bivand and Piras (2015)
MEM model selection	Selection with AICc	test.W	spacemakeR	Dray (2013)
PERMANOVA	Permutational multivariate analysis of variance	adonis	vegan	Oksanen et al. (2016)
Distance matrix selection	Selection of most appropriate distance matrix	rankindex	vegan	Oksanen et al. (2016)
NMDS	Nonmetric multidimensional scaling	metaMDS	vegan	Oksanen et al. (2016)
Cluster numbers	Optimal numbers of clusters	NbClust	NbClust	Charrad et al. (2014)
Fuzzy cluster analysis	Fuzzy cluster analysis	fanny	cluster	Maechler et al. (2017)
Variable fitting	Variable fitting on NMDS ordination	envfit	vegan	Oksanen et al. (2016)
LMEM	Linear mixed effect models	lmer	lme4	Bates et al. (2015)
CV	Coefficient of variation	cv	raster	van Etten (2012)
Least square means	Least square means comparison	cld & lsmeans	lsmeans	Lenth (2016)
Soil texture	Classification according to USDA taxonomy	soiltexture	soiltexture	Moeys (2016)
NMDS plots	Plotting NMDS plots	ggvegan	ggvegan	Simpson (2015)
MEM plots	Plotting MEM plots	s.value	ade4	Dray and Dufour (2007)
Maps	Spatial plots integrating google maps information	get_googlemap	ggmap	Kahle and Wickham (2013)
Plots	All other plots	ggplot2	ggplot2	Wickham (2009)
η^2	Effect size of categorical and numerical variables (η^2) Bakeman (2005)	etasq	heplots	Fox et al. (2007)
Bayesian inference	Bayesian inference posterior distributions with 95 % credible intervals (see Korner-Nievergelt et al. (2015))	sim	arm	Gelman and Su (2016)

4

Litter decomposition driven by soil fauna, plant diversity and soil management in urban gardens



Text mining: Most frequent 100 words from chapter 4 (shape: springtail), created with the "wordcloud2" R package (Lang 2016).

Litter decomposition driven by soil fauna, plant diversity and soil management in urban gardens

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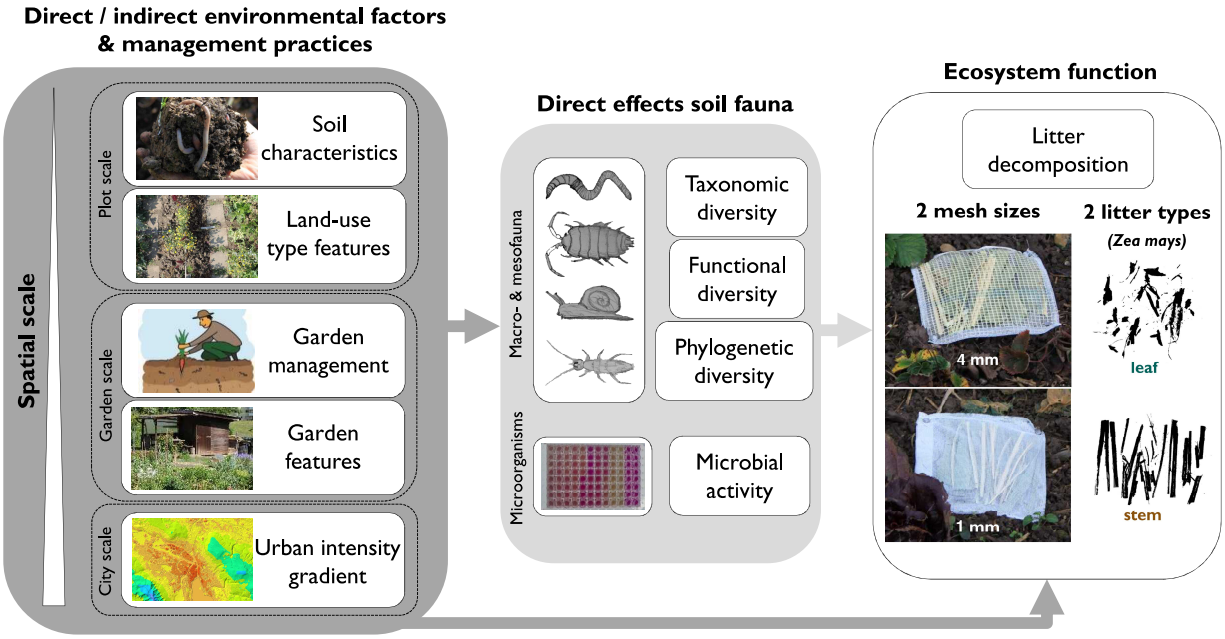
Keywords: Urban gardening, Litter bag decomposition, Biodiversity ecosystem functioning (BEF), Urban ecosystem services, Urban soil biodiversity, MidDRIFTS analysis, Urban warming

Science of the Total Environment
RESEARCH ARTICLE
Published: 21. December 2018
DOI: 10.1016/j.scitotenv.2018.12.235

HIGHLIGHTS

- Garden management affected both fauna diversity and litter decomposition
- Soil fauna species richness covaried positively with decomposition rates
- Plant diversity increased fauna diversity and microbial activity
- Urbanisation density was positively associated with litter decomposition
- MidDRIFTS analysis revealed variance in litter residue quality after decomposition

GRAPHICAL ABSTRACT



Graphical abstract: A priori conceptual structural equation model depicting direct pathways of soil fauna as well as direct and indirect pathways of soil characteristics, land-use features, garden management, garden features and urbanisation intensity on litter decomposition.

ABSTRACT

In the face of growing urban densification, green spaces in cities, such as gardens, are increasingly important for biodiversity and ecosystem services. However, the influences of urban green space management on biodiversity and ecosystem functioning (BEF) relationships is poorly understood. We investigated the relationship between soil fauna and litter decomposition in 170 urban garden sites along a gradient of urbanisation intensity in the city of Zurich, CH. We used litter bags of 1 and 4 mm mesh size to evaluate the contribution of soil meso- and macrofauna on litter decomposition. By using multilevel structural equation models (SEM), we investigated direct and indirect environmental effects and management practices on litter decomposition and litter residue quality. We evaluated the role of taxonomic, functional and phylogenetic diversity of soil fauna species on litter decomposition, based on a sample of 120 species (81007 individuals; 39 collembola, 18 earthworm, 16 isopod, 47 gastropod species). We found highest litter decomposition rates using 4 mm mesh size litter bags, highlighting the importance of soil macrofauna. Urban warming, a proxy for urbanisation intensity, covaried positively, whereas soil disturbances, such as intensive soil and crop management, were negatively correlated with decomposition rates. Interestingly, soil fauna species richness decreased, with the exception of gastropods, and soil fauna abundance increased with urban warming. Our data also show that plant species richness positively affected litter decomposition by increasing soil fauna species richness and microbial activity. A multivariate analysis of organic compounds in litter residues confirmed the importance of soil fauna species richness and garden management on litter decomposition processes. Overall, we showed, that also in intensively managed urban green spaces, such as gardens, biodiversity of plants and soil fauna drives key ecosystem processes. Urban planning strategies that integrate soil protecting management practices may help to maintain important ecosystem services in this heavily used urban environment.

4.1 Introduction

Anthropogenic activities have an unprecedented impact on ecosystems worldwide (Butchart et al. 2010). Globally, over 60 % of the world's ecosystems are degraded or managed unsustainably (UN FAO 2011), causing the loss of biodiversity across the globe (Ceballos et al. 2015). This may drastically accelerate the rates of change in ecosystem processes (Cardinale et al. 2011), thereby altering the productivity and regeneration capacity of ecosystems (Wardle et al. 2011).

One example of a global change driver is urbanisation. Nowadays, the majority of people live in cities (54 % in 2014) with an expected growth reaching 66 % within the next three decades (United Nations 2015a). Urban areas are expanding faster than any other land-use type (Hansen et al. 2005). This increasing urbanisation has a major influence on the environment (Grimm et al. 2008) but also on local processes (Groffman et al. 2014), by altering biogeochemistry, hydrology and biodiversity (Groffman et al. 2017).

Cities are unique ecosystems (Kaye et al. 2006) consisting of complex mosaics of different land-use types (Zhou et al. 2017). As anthropogenic ecosystems, they provide an ideal opportunity to investigate the influence of human activities on biodiversity and ecosystem services (Aronson et al. 2016). Overall, the expanding urbanisation is expected to reduce species diversity and abundance, increase biotic homogenisation and to negatively affect species interactions with likely negative consequences on key ecological processes (Foley 2005, McKinney 2008). On the other hand, depending on the intensity of urbanisation, cities can harbour a remarkably high biodiversity (Godefroid and Koedam 2007): even including endangered native species (Ives et al. 2016), which may even exceed the rural surroundings (Kühn et al. 2004). A possible explanation for this pattern is the high spatial and environmental heterogeneity in cities (e.g. Rebele 1994, Sattler et al. 2010). In this respect, urban green spaces can offer conservation opportunities (Mata et al. 2017) and benefits for humans, as biodiverse urban green spaces are known to improve well-being and health of residents (Keniger et al. 2013).

While green spaces such as gardens are becoming important refuges for native biodiversity in many cities (Goddard et al. 2010), soil sealing is steadily increasing due to the demand for infrastructure (Benton et al. 2003). Besides their role for biodiversity conservation, urban gardens also provide key ecosystem services (Zhu et al. 2018), which are otherwise negatively affected by urbanisation (Ziter 2016). These services include climate and water regulation along with recreation, health and social cohesion (Bell et al. 2016, Haase et al. 2014). However, there are still few studies about the benefits of urban gardens (Cabral et al. 2017), despite the fact that they cover large proportions of urban green spaces in many cities (Loram et al. 2007).

The importance of biodiversity in maintaining

ecosystem services is getting recognised (Hector and Bagchi 2007), however, our understanding of the underlying mechanisms remains limited. This is in part due to the lack of real world observations (Gossner et al. 2016), especially in human dominated ecosystems such as cities (Isbell et al. 2017b, Schwarz et al. 2017). Overall, there is substantial experimental evidence of a positive influence of biodiversity on the functioning of ecosystems (Cardinale et al. 2011, Duffy 2009). This positive influence is often derived from studies with productivity as an ecosystem service (Caruso et al. 2018, Vogel et al. 2013), but has also been found for ecosystem functions such as litter decomposition (Allan et al. 2013, Weisser et al. 2017).

Decomposition of organic matter is one of the central functions of ecosystems (Swift et al. 1979) and is mainly driven by environmental conditions such as climate or soil properties, litter quality, and the composition of decomposer species communities (Cadisch and Giller 1997, McClaugherty and Berg 2011, Swift et al. 1979). A loss or change of decomposer diversity and species composition is likely to alter decomposition dynamics (Hättenschwiler and Gasser 2005, Heemsbergen 2004), but its extent and consequences remain difficult to predict (García-Palacios et al. 2016a, Hättenschwiler et al. 2005, Hooper et al. 2005). The majority of studies has analysed the effect of different litter types on decomposition (Patoine et al. 2017) and has shown a mean positive effect of leaf litter species diversity on litter mass loss across biomes (Handa et al. 2014). Nielsen et al. (2011) explored the relationship between soil fauna and ecosystem functions relevant for C cycling. In 11 out of 15 studies, they reported a positive relationship between soil fauna species richness and decomposition. Yet, the role of soil fauna on litter decomposition, including not only taxonomic but also functional or phylogenetic metrics, remains mostly unknown (Patoine et al. 2017), especially in urban ecosystems (Schwarz et al. 2017), were environmental conditions and management practices can be profoundly different compared with rural areas (Gaston et al. 2010).

The complex urban soils are the foundation of a range of functions and services such as supporting (soil formation, nutrient cycling or habitat space), regulating (climate, floods and water), carrying or cultural services (Rawlins et al. 2015, Tresch et al. 2018a), which are essential for liveable and resilient cities (Elmqvist et al. 2015). Urban soils are influenced by several factors such as compaction, urban warming – the elevation in urban relative to non-urban temperatures (Oke 1995) – increased precipitation (Gilbert 1989), modified hydrology (Francis 2014) and increased deposition of pollutants and nitrogen (Kaye et al. 2006). Their soil properties and biogeochemical cycles are also altered directly by anthropogenic activities such as construction work (Lorenz and Lal 2009).

Nevertheless, in the case of urban garden soils, specific garden management practices can increase soil quality indices, such as organic matter content and biological activity, over the long term (Edmondson et al. 2014, Tresch et al. 2018a), if organic cultivation rules without chemical fertilisation and pesticides have been implemented (Bretzel et al. 2018). Thus, it seems probable that some urban soils contains higher organic carbon contents than those in rural landscapes (Edmondson et al. 2012).

While above-ground BEF relationships are often shaped by an interaction of local-scale (e.g. vegetation structure) and landscape-scale factors (Angold et al. 2006, Frey et al. 2018), our understanding of how such multi-scale factors affect the belowground BEF relationships is still limited (Lin and Egerer 2018). For example, recent studies of ecosystems services rarely addressed the high spatial heterogeneity and complexity of urban soils (Ziter and Turner 2018), and neither consider the variety of direct and indirect anthropogenic influences across spatial scales (Enloe et al. 2015), nor the role of different aspects of biodiversity (Schwarz et al. 2017). An assessment of BEF relationships in cities therefore requires integrated analytical tools, such as structural equation modelling (SEM), that allow for a causal understanding of direct and indirect at different spatial scales (Eisenhauer et al. 2015), especially including human components (Isbell et al. 2017b).

In this study, we chose litter decomposition as model ecosystem function, due to its importance in maintaining soil quality in urban gardens (Schram-Bijkerk et al. 2018) and because of the lack of studies analysing the effect of urbanisation on litter decomposition (Dorendorf et al. 2015). We investigated direct and indirect effects of environmental factors and management practices on litter decomposition along an urban intensity gradient, which was measured as the local temperature increase due to urban warming. The objectives of our study were to investigate the following three aspects: (i) the direct effects of abiotic and biotic factors on litter mass loss, (ii) the direct and indirect effects of soil characteristics, garden management and urban warming on soil fauna and litter decomposition and (iii) to analyse which factors influenced litter residue quality after decomposition. Overall, we hypothesised that factors

at plot and garden scale have a greater impact on soil biodiversity and litter decomposition than at the city scale, because of the dominant management influence on soil fauna and functioning (Lavelle et al. 2006).

4.2 Material and methods

4.2.1 Study area and site selection

The study took place in urban gardens in the city of Zurich, Switzerland (Figure 4.1). Zurich lies in the temperate climate zone with mean annual temperature of 9.3°C (1981-2010) and mean annual precipitation of 1134 mm (MeteoSwiss 2017). With an area of approximately 92 km^2 and a population of 0.4 million citizen it belongs to the globally most common city class (United Nations 2015a) and is therefore an ideal system to study BEF relationships in urban environments.

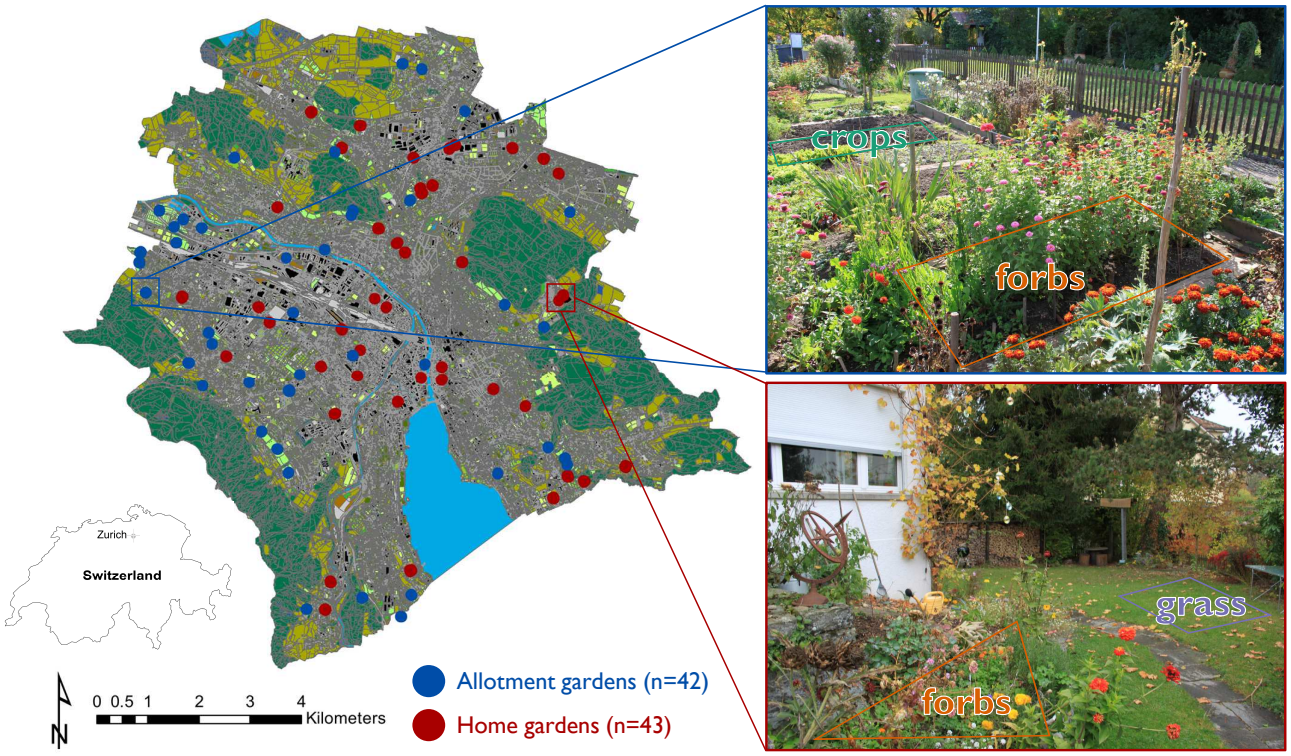


Figure 4.1 – Typical examples of home (red) and allotment (blue) gardens of Zurich. Within each of the 85 urban gardens, two sampling sites were chosen according to the main garden land-use types: crops, forbs and grass.

We selected 42 allotment and 43 home gardens (Figure 4.1), representing the two most common urban garden types worldwide (Lin et al. 2017). Allotment gardens are cultivated lots of land in an urban area, normally used for recreational purposes or the cultivation of fruits, vegetables and flowers (Bell et al. 2016). The allotment gardens in Zurich mainly belong to the city municipality. The first lots were established in 1907 (Bell et al. 2016), succeeding a history of self-supplying citizen gardens dating back to the 16th century (Christl et al. 2004). Home gardens are privately owned garden lots belonging to single-occupancy and terraced houses. They cover 25 % of the total urban green space of Zurich, while allotments only account for 7 % (Grün Stadt Zurich 2010). Gardens were selected according to a stratified sampling design based on an urban habitat map (Grün Stadt Zurich 2010) with three independent strata: (i) garden type (private vs. allotment), (ii) garden management (low vs. high vertical vegetation structure and proportion of native plant species) and an (iii) urbanisation intensity gradient, ranging from densely built-up areas to peripheral areas within the city margins of Zurich (Frey and Moretti 2019) . In each of the 85 urban gardens, we selected two sampling sites (2 m × 2 m) with contrasting soil disturbance. All sites (n=170) were associated to one of the common urban land-use types: crops (annual vegetables), forbs (perennial flowers and berries) and grass (lawn and meadows).

4.2.2 Litter decomposition experiment

Litter decomposition was assessed both quantitatively with litter bags and qualitatively with spectroscopy of litter residues. We used litter bags (18 cm × 18 cm; Finerty et al. (2016)) of two mesh sizes (1 mm and 4 mm) to evaluate the contribution of macrofauna to litter decomposition. A fine mesh (1 mm) was used on the bottom for both litter bag types to avoid loss of litter material. Litter bag contents were standardised by using 4 g of oven dried (40°C) maize (*Zea mays* L.) leaves with equal proportions of 2 ± 0.01 g leaf and 2 ± 0.01 g of stem material (i.e. central leaf vein). Maize was absent in all investigated gardens which avoided facilitation effects of decomposition. Leaves and stems of *Zea mays* L. contrasts in the ratio of carbon to nitrogen (leaf: 18 ± 0.3, stem: 71 ± 3) and also in the leaf tensile strength (leaf: 1.2 ± 0.06 MNm^{-2}, stem: 4.4 ± 0.3 MNm^{-2}) and thus in the palatability and accessibility for soil decomposer organisms. Litter traits were measured on ten random samples per litter type following Pérez-Harguindeguy et al. (2013) (Table S 4.1). In total 340 litter bags (2 mesh sizes × 2 sites × 85 gardens; Figure S 4.1) were placed on top of the soil for six months (December 2015 - May 2016) and the remaining litter was dried (40°C). Decomposition was expressed as percentage change in litter mass before and after decomposition. Litter residue quality after decomposition was assessed by the composition of functional organic compounds measured by diffuse reflectance Fourier transform mid-infrared spectroscopy (midDRIFTS) following Rasche et al. (2013). For measurement details see Tresch et al. (2018a). We applied midDRIFTS before and after decomposition revealing the biochemical quality of the organic residues (see Table 4.1).

Table 4.1 – Litter residue analysis of contrasting functional organic compounds, selected as midDRIFTS peak measurements modified after Kunlanit et al. (2014).

Label	Frequency [cm^{-1}]	Structural assignment	Quality
Labile_A	2800-3010	Aliphatic C-H stretching (Senesi et al. 2003, Stevenson 1994)	labile
Labile_B	1915-2200	Carbohydrate overtones of C-OH stretching (Janik et al. 2007)	labile
Labile_C	1094-1147	C-OH of aliphatic OH (Tatzber et al. 2010)	labile
Stable_D	1700-1772	C-O stretching of COOH and ketones (Rodriguez 2011, Stevenson 1994)	stable
Stable_E	1620-1700	Aromatic COO stretching (Demyan et al. 2012, Nault et al. 2009, Smidt and Meissl 2007)	stable
Stable_F	1537-1620	C=C of aromatic groups (Duboc et al. 2012)	stable
Stable_G	1401-1445	C-H and N-H aromatic amide II, COO- stretching of some aromatic organic acids e.g. malonic and or benzoic acids (Stevenson 1994, Tatzber et al. 2010)	stable
Stable_H	1296-1350	Benzoic acids and C-O of aryl ethers, C-O of phenolic groups (Tatzber et al. 2010)	stable

4.2.3 Soil fauna

Soil macrofauna

We sampled gastropods (i.e. snails), isopods (i.e. woodlice) and earthworms as major macrofauna litter decomposers (Briones 2014). Gastropods and isopods were sampled using a triplet of pitfall traps (70 mm in diameter) as described in Frey et al. (2016). Pitfall traps were filled with 0.2 % Rocima solution (Acima, Buchs, CH) and weekly emptied from May 25 to August 18, 2015. Gastropod identification followed (Hausser 2005), whereas isopods were identified by specialists (see acknowledgements). Earthworms were collected between mid-September and the end of October 2015 using a standardised method combining hand sorting (Bartlett et al. 2010) and mustard solution (0.6 %) extraction (Lawrence and Bowers 2002). The sampling sites were 0.3 m × 0.3 m with a depth of 0.3 m. Species identification was done according to Bouché (1977), Sims and Gerard (1999). All decomposers were sampled at the litter bag sites of 2 m × 2 m.

Soil mesofauna

Collembola are important micro-arthropods for decomposition in terrestrial ecosystems (Rusek 1998). They were collected from mid-November until the end of December 2015 during the initial phase of the decomposition experiment. Six replicated undisturbed soil cores (5 cm diameter, 8 cm length, Eijkelkamp, NL) were taken randomly from the topsoil (Querner and Bruckner 2010) within the 2 m × 2 m sampling sites. Mesofauna was extracted using a

high temperature and moisture gradient MacFadyen extractor with an increasing temperature gradient from 20 to 60 °C (cf. Table S 4.2). Three undisturbed soil cores: each of 175 cm^3, were pooled together for the extraction period, which lasted one week. Collembola were identified at the species level by experts (see acknowledgements). All soil macro- and mesofauna species were stored in 70 % ethanol and juveniles were not taken into account.

Biodiversity indices

Taxonomic diversity

There are two principle components of taxonomic diversity: species richness and species evenness, which is how evenly species are distributed within a community (Magurran and McGill 2011). The calculated indices are soil fauna abundance (N), species richness (S), Shannon diversity index ($D_{Shannon}$) and Shannon evenness ($E_{Shannon}$). Species richness was the total number of soil fauna species observed in each study site and N the sum of soil fauna abundance on the lowest level of temporal resolution. The abundance was manipulated for the total abundance of gastropods and isopods to represent one week instead of ten weeks due to the sampling of those invertebrates by dividing N with ten ($\frac{N}{10}$). The Shannon diversity index was calculated as $D_{Shannon} = -\sum_{i=1}^{S} p_i ln p_i$, where p_i represents the proportion of soil fauna abundance belonging to species i and the Shannon evenness was calculated as $E_{Shannon} = \frac{D_{Shannon}}{lnS}$.

Functional diversity

Functional diversity indices includes components of richness, evenness and divergences of trait values and their abundances (Villéger et al. 2008). The three components were taken into account by calculating the trait onion peeling (TOP; Fontana et al. (2016)) index, the sum of all convex hulls' areas of a community in the trait space, the trait even distribution index (TED; Fontana et al. (2016)), the regularity in the distribution of species, and the functional dispersion index (FDis; Laliberté and Legendre (2010)), which is the mean distance of species to the centroid of trait distribution, based on standardised trait values (mean=0, standard deviation=1). We used body size as a trait directly connected to the food resource and consumption rate (Bardgett and Wardle 2010) and the eco-behavioural trait vertical distribution, reflecting functional life forms of soil fauna species (Briones (2014); Table S 4.5).

Phylogenetic diversity

Four variables related to phylogenetic diversity (Paradis 2011) were calculated including phylogenetic diversity (PD), the sum of branch lengths, phylogenetic species variability (PSV), the mean of the phylogenetic correlations among species in the community, phylogenetic species evenness (PSE), which is PSV with relative species abundance, and phylogenetic species richness (PSR), PSV multiplied by S. They were calculated by building a phylogenetic tree ('rotl' package; Michonneau et al. (2016)) with branch lengths ('ape' package; Paradis et al. (2004)) based on the open tree of life project (Hinchliff et al. 2015) following (Paradis 2011).

Microbial activity

Soil microbial activity of each litter bag site was quantified by the multiple substrate-induced respiration (MSIR) rate measured with the MicroResp™ system (Campbell et al. 2003) following Sradnick et al. (2013). A range of 19 different C-substrates, including H_2O, were selected to present a spectrum of root exudates typically occurring in soil, comprising six amino acids, one amino sugar, four sugars, four carboxylic acids, two phenolic acids and one hemicellulose (Campbell et al. 1997). The MSIR was calculated as the sum of all C-substrates respiration values. Details about the C-substrates, calibration and measurement procedure are described in Table S 4.3.

4.2.4 Environmental factors and management practices

Soil characteristics

Soil characteristics at the litter bag sites were quantified as a comprehensive set of soil quality indices (Bünemann et al. 2018), including nine physical (bulk density (BD), pore space volume (PV), stable aggregates (SA), soil texture (clay, sand, silt), water holding capacity (WHC), penetration resistance (PR) and soil depth), nine chemical (pH, electrical conductivity (EC), nutrients (P, K, Mg, Fe, Cu, Mn, B)) and ten biological (basal respiration, bacteria, C_{min}, N_{min} , C_{mic}, N_{mic}, DOC, DON, SOC, TON) soil properties as well as nine heavy metals (Sb, As, Co , Cu, Pb, Ni, Zn, V, Ba) as described in Tresch et al. (2018b) (cf. Table 4.2).

Garden land-use type features

Six garden land-use type features expected to influence soil biodiversity and decomposition were as-

sessed on at each litter bag site. Plant species richness (S plants) was calculated as the total number of plant species per sampling site, using a floristic inventory of cultivated and spontaneously growing plants. The species list of the 600 plants and the identification methodology can be found in (Frey and Moretti 2019). The amount of sun hours measured with a solar compass at maximum vegetation stage in July 2015 at 0.3 m height and the proportion of bare or impervious soil by taking orthogonal photographs taken at 3 m height and classified with digital image classification (ImageJ). Mean inclination of the sites (slope) was measured with a digital elevation meter. Each site was grouped into one of the three land-use types: grass, forbs or crops.

Garden management and garden features

Garden management was assessed by using a questionnaire with 26 questions (cf. Table S 4.6) about the physical (e.g. frequency of lawn cutting) and chemical (e.g. fertiliser) soil management practices. Questions were asked for each land-use type separately. A management intensity index was created similar to Smith et al. (2006) by summing up all management variables, which were ordered from low to high intensity on a five level Likert scale. The time since the last major change in the garden was defined as the garden age, while the garden types (allotment or home garden) were given by the sampling design.

Urban intensity gradient

We used urban warming as a surrogate of urbanisation intensity, because it is a direct effect of the density and amount of impermeable surfaces (Figure S 4.7; Davidson and Janssens (2006)). Urban warming was defined as the deviation of mean night temperatures near the surface (ranging from 0 to + 6°C) from a local climate model by Parlow et al. (2010). For group comparisons (Table 4.3, Table 4.6), urban warming was grouped into three classes: class 1 containing gardens with 0 - 1°C mean night temperatures, class 2 with 2 - 3°C and class 3 with 4 - 5°C.

4.2.5 Data analysis

Litter decomposition model

Direct effects on litter decomposition were analysed with a linear mixed effect model (LMEM), which was chosen due to the nested structure of the sampling sites within the gardens. The response variable (i.e. litter mass loss) was transformed ($log(100 - x + 1)$) to approach independent and identically distributed residuals (Figure S 4.2). Before model selection, Pearson correlation analysis (Dormann et al. 2013) and the variance inflation factor (VIF<10; Borcard et al. (2011)) were used to reduce collinearity issues. We chose species richness (S) and evenness ($E_{Shannon}$) to represent taxonomic diversity since abundance (N) was correlated with evenness (r<-0.6; Figure S 4.4). Subsequently, TED was selected to represent trait evenness and PSV the phylogenetic diversity, since all other indices were highly correlated (r>0.7; Figure S 4.5) with species diversity or evenness. Goodness of fit statistics for LMEM were the widely applicable information criterion (WAIC), which is a Bayesian version of the AIC (Watanabe 2010) and variance explained (including random factor $R^2_{Conditional}$ and for fixed effects $R^2_{Marginal}$ only; Nakagawa and Schielzeth (2013)). These statistics, suggested inclusion of garden type, MSIR, representing the microbial activity and urban warming, in the final litter decomposition model (delta WAIC 41±13, delta $R^2_{Marginal}$

Table 4.2 – Variables expected to directly or indirectly affect litter decomposition in urban gardens, including potential positive ↑ or negative ↓ effects on decomposition. Management questions used for the calculation of garden management variables can be found in Table S 4.9. Additional information about soil characteristics can be found in Tresch et al. (2018b) and about plant characteristics in Frey and Moretti (2019).

Variables	Expected effect	Description
Direct/indirect environmental factors and management practices		
Plot scale		
Soil physical characteristics		
BD [g cm^{-3}]	↓	Soil bulk density
Depth [cm]	↑	Soil depth (up to max. 80 cm)
PR [MPa]	↓	Mean penetration resistance
PV [%]	↑	Soil pore space volume
SA [%]	↑	Soil stable aggregates
Soil texture (3) [%]		Soil texture clay, silt, sand
WHC [%]	↑	Soil water holding capacity
Soil chemical characteristics		
pH		Soil pH
EC [μS cm^{-1}]		Electrical conductivity
Nutrients (7) [mg kg^{-1}]	↑	Nutrient contents of P (phosphorus), K (kalium), Mg (magnesium), Fe (iron), Cu (copper), Mn (manganese), B (boron)
Soil biological characteristics		
Basal [μg CO_2-C $g^{-1}h^{-1}$]	↑	Basal soil respiration
Bacteria [gene copy numbers]	↑	Bacterial gene copy numbers of 16S qPCR rRNA
C_{min} [μg CO_2-C kg^{-1}]	↑	Carbon mineralisation (4 weeks respiration measurement)
N_{min} [mg kg^{-1}]	↑	Nitrogen mineralisation
C_{mic} [mg kg^{-1}]	↑	Microbial biomass carbon
N_{mic} [mg kg^{-1}]	↑	Microbial biomass nitrogen
DOC [%]	↑	Dissolved organic carbon
DON [%]	↑	Dissolved organic nitrogen
SOC [%]	↑	Soil organic carbon content
TON [%]	↑	Total organic nitrogen content
Soil heavy metals		
Heavy metals (9) [mg kg^{-1}]	↓	Sb (antimony), As (arsenic), Co (cobalt), Cu (copper), Pb (lead), Ni (nickel), Zn (zinc), V (vanadium), Ba (barium)
Land-use type characteristics		
Bare soil [%]	↓	Proportion of soil not covered with vegetation on plot level ($10m^2$)
Impervious [%]	↓	Proportion of sealed soils on plot level ($10m^2$)
Sun hours [h]	↑	Solar hours (solar compass at maximum vegetation stage in July 2015)
Slope [%]	↑	Mean inclination measured with a digital elevation meter (n=10)
S plants	↑	Plant species richness identified at plot level ($10m^2$)
Land-use types	↑ ($grass$)	Three main garden land-use types (grass, forbs, crops)
Garden scale		
Garden management		
Management index	↓	Management intensity index (sum of 26 management questions; Table S 4.6)
Disturbance	↓	Frequency of major soil disturbance (DiggingVeg, DiggingFlower, CareLawn)
Weeding	↓	Frequency of removing weeds in the garden (Weeds)
Pesticides	↓	Pesticides (PestLawn, PestFeg, PestFlower, PestTrees, WeedingHerbicide)
Removing leaves	↓	Removing leaves in the garden (Leaves)
Water	↑	Frequency of irrigation (WaterLawn, WaterVeg, WaterFlower)
Fertiliser	↑	Applying fertiliser (FertGrass, FertCrops, FertForbs)
Garden features		
Garden age [a]	↑	Time since last major change in the garden (e.g., exchange of soil)
Garden type	↑ ($home$)	Two main urban garden types (allotment and home gardens)
City scale		
Urban intensity gradient		
Urban warming	↓	Deviation of local mean night temperatures (0 to + 6°C)
Sealed area	↓	Proportion of sealed area around each garden (30, 50, 100, 250, 500 m radius)
Direct effects soil fauna		
Taxonomic diversity		
N	↑	Soil fauna abundance
S	↑	Soil fauna species richness
$D_{Shannon}$	↑	Shannon diversity index
$E_{Shannon}$	↓	Shannon evenness
Functional diversity		
TED	↓	Trait even distribution
FDis	↑	Functional dispersion index
TOP	↑	Trait onion peeling index
Phylogenetic diversity		
PD	↑	Phylogenetic diversity
PSE	↓	Phylogenetic species evenness
PSV	↑	Phylogenetic species variability
PSR	↑	Phylogenetic species richness
Microbial activity		
MSIR	↑	Multiple substrate-induced respiration rate

0.22±0.05 and delta $R^2_{Conditional}$ 0.32±0.08; Table S 4.7). The final LMEM was analysed with a Bayesian approach including means and 95 % credible intervals of the Bayesian inference posterior distributions following Korner-Nievergelt et al. (2015).

Multilevel structural equation model (SEM)

Direct and indirect effects of soil characteristics, garden management, and the urban intensity gradient on litter decomposition were assessed using a multilevel SEM (Shipley 2016) implemented in the 'piecewiseSEM' package (Lefcheck 2016). Leaf 4 mm was used as a litter decomposition response, as it is better degradable than stem material and includes all soil fauna communities of this study. Composite models of the SEM were LMEM (Pinheiro et al. 2018) with garden ID (two sites nested within one garden) as random factor. Basis set construction (see R script in the Supplementary Materials), goodness-of-fit tests and parameter estimations were conducted according to AICc and Fisher's C statistic ($p > 0.05$; Shipley (2016)). We checked for missing paths in the SEM with Shipley's d-separation test. Model assumption were tested (Figure S 4.12) and potential spatial autocorrelation patterns in the response variables were calculated with Moran's I autocorrelation indices (Popescu et al. 2012) and the spatial structure in the model residuals (Figure S 4.10) using semivariograms (Pebesma 2004).

NMDS ordination and PERMANOVA

Litter residues after decomposition were analysed by a multivariate analysis of midDRIFTS measurements (Table 4.2) with a non-metric multidimensional scaling (NMDS) and a permutational multivariate analysis of variance (PERMANOVA) with a distance matrix (Gower index) using the 'vegan' package (Oksanen et al. 2016). For the PERMANOVA, we used only significant variables ($p \leq 0.05$) from the multilevel SEM, and for the NMDS, only significant variables from the PERMANOVA were fitted (Figure 4.5).

All statistical analyses were performed using R 3.4.2 (R Core Team, 2017), a script with R codes used for the calculation of descriptive statistics, tables and diversity indices as well as the LMEM and SEM are provided in the Supplementary Materials.

4.3 Results

In the biodiversity assessment of urban gardens, we collected 39 collembola (13694 individuals), 18 earthworm (3128 individuals), 16 isopod (59650 individuals) and 47 gastropod (4535 individuals) species (Table S 4.5). Total soil fauna species richness was highest for forbs and lowest for crops and decreased with increasing urban warming class. Whereas soil fauna abundance was highest for grass, lowest for crops and increased with urban warming classes (Table 4.3). These results varied among the taxonomic groups, for instance $S_{Gastropods}$ increased while $S_{Earthworms}$, $S_{Collembola}$, $S_{Isopods}$ decreased with urban warming classes (cf. Table S 4.4).

Mean mass loss of leaf litter (1 mm: 61.2±2.0 %; 4 mm: 79.6±2.2 %) was significantly higher (Table S 4.9; Figure 4.2) compared to the more recalcitrant stems (1 mm: 40.1±1.0 %; 4 mm: 37.9±0.8 %). We observed a significant effect of the macrofauna community (4 mm mesh size) for leaf litter compared to those with 1 mm mesh size that exclude soil macrofauna species. No mesh size effect was observed for the stems.

4.3.1 Litter decomposition model

Garden land-use types showed the largest effect in the LMEM (Table S 4.8) on litter decomposition, irrespective of litter type and mesh size (Figure 4.3; Figure S 4.8). Mean decomposition rates of grass sites (Leaf 4 mm: 90.5±1.8 %, Stem 4 mm: 42.8±0.9 %, Leaf 1 mm: 69.6±2.2 %, Stem 1 mm: 46.0±1.0 %) were higher compared to forbs (Leaf 4 mm: 80.7±3.5 %, Stem 4 mm: 37.9±1.2 %, Leaf 1 mm: 56.2±3.5 %, Stem 1 mm: 39.1±1.6 %) and crops (Leaf 4 mm: 61.8±5.6 %, Stem 4 mm: 30.5±1.9 %, Leaf 1 mm: 45.1±5.3 %, Stem 1 mm: 30.0±3.1 %). Urban warming was positively related to higher decomposition rates irrespective to the litter type and mesh size. Soil fauna species richness (S) and microbial activity (MSIR) were positively related to mass loss of leaves in 4 mm and stems in 1 mm litter bags (Table S 4.8). None of the functional and phylogenetic diversity indices significantly explained decomposition in our study. Mean decomposition rates were influenced by garden management practices. For instance, major soil disturbance (i.e. digging) correlated with decreased decomposition rates in forbs irrespective of litter type and mesh size (Table S 4.9). The addition of compost led to increased leaf decomposition in 4 mm bags for grass sites and water application to increased decomposition of stems in 1 mm bags.

4.3.2 Multilevel structural equation model

The multilevel SEM (Figure 4.4) was used to test both direct and indirect effects of environmental

factors (Table 4.2) on the soil fauna community and on the ecosystem function litter decomposition. The SEM fit the criteria (Shipley 2016) that there are no missing relationships among unconnected variables (AICc=299.7, Fisher's C=124.2, P-value=0.96). Overall, 55 % of the variation in leaf litter decomposition of the 4 mm litter bags has been explained in the multilevel SEM. Soil fauna species richness had a direct positive influence on decomposition and was positively affected by plant species richness in the gardens, but negatively by soil antimony (Sb) content, explaining in total 39 % of the variation in species richness (Table 4.4). Garden soils with higher contents of microbial biomass (C_{mic}) and bacteria also had elevated microbial activity (MSIR) that positively affected litter decomposition. Moreover, plant species richness covaried positively with microbial activity, explaining 41 % of the variation.

4.3.3 Effects on litter residue quality

Several variables had a significant effect on the quality of litter residues after decomposition (Table 4.5). Land-use types showed the greatest effect on the composition of litter resides of functional organic compounds (PERMANOVA; R^2=0.08; P=<0.001). In the NMDS ordination (Figure 4.5) crop sites were associated with more labile litter quality compounds, while grass and forb sites were associated with more stable compounds. Soil fauna species richness (S; PERMANOVA; R^2=0.02; P=0.01), was positively associated with stable organic compounds. Furthermore, the management variable leaf removal (PERMANOVA; R^2=0.03; P=<0.001) as well as total organic nitrogen (PERMANOVA; TON; R^2=0.02; P=0.01) contributed to the separation of the litter residue quality after decomposition for the leaf 4 mm litter bags.

Table 4.3 – Descriptive statistics of biodiversity components. Soil fauna species richness (S) and abundance (N) are presented in Table S 4.4, and the species and trait list of soil fauna used for the calculation of the biodiversity components are shown in Table S 4.5. Presented values are mean values with standard errors. $D_{Shannon}$: Shannon diversity index, $E_{Shannon}$: Shannon evenness, FDis: Functional dispersion index, MSIR: Multi substrate-induced respiration rate, N: Abundance, PD: Phylogenetic diversity, PSE: Phylogenetic species evenness, PSR: Phylogenetic species richness, PSV: Phylogenetic species variability, S: Species richness, TED: Trait even distribution, TOP: Trait onion peeling index. All garden sites n=168, crops n=46, forbs n=52, grass n=70, allotment n=82, home n=86, Urban warming class 1 n=34, Urban warming class 2 n=114, Urban warming class 3 n=20.

	Taxonomic diversity				**Functional diversity**			**Phylogenetic diversity**				**Microbial activity**
	S	N	$D_{Shannon}$	$E_{Shannon}$	TOP	TED	FDis	PD	PSV	PSR	PSE	MSIR
All garden sites	20.3±0.4	162±10	1.79±0.1	0.6±0.01	21.7±0.4	0.94±0.002	0.85±0.02	4.42±0.05	0.68±0.01	8.6±0.2	0.35±0.01	96.6±1.2
Land-use types												
Crops	18.6±0.7	115±14	1.82±0.1	0.63±0.02	20.8±0.9	0.95±0.005	0.99±0.05	4.26±0.10	0.68±0.01	7.9±0.3	0.44±0.02	89.6±2.5
Forbs	21.9±0.8	162±22	1.67±0.1	0.54±0.03	22.9±0.8	0.94±0.003	0.83±0.05	4.59±0.08	0.66±0.01	9.1±0.3	0.33±0.02	99.3±1.9
Grass	20.2±0.6	192±14	1.85±0.1	0.62±0.02	21.3±0.6	0.93±0.004	0.77±0.03	4.41±0.06	0.68±0.01	8.5±0.3	0.31±0.02	99.2±1.7
Garden types												
Allotment	20.7±0.6	147±13	1.88±0.1	0.63±0.02	22.1±0.7	0.94±0.004	0.9±0.04	4.47±0.07	0.68±0.01	8.8±0.3	0.39±0.02	91.9±1.8
Home	19.9±0.5	177±14	1.69±0.1	0.57±0.02	21.3±0.6	0.94±0.003	0.8±0.03	4.38±0.06	0.67±0.01	8.3±0.2	0.31±0.02	101.1±1.5
Urban warming												
Class 1	21.3±1.0	120±13	1.97±0.1	0.66±0.02	22.3±1.0	0.95±0.005	0.99±0.05	4.47±0.11	0.68±0.01	9.0±0.4	0.42±0.03	93.7±2.9
Class 2	20.3±0.5	154±10	1.86±0.1	0.62±0.01	21.8±0.5	0.94±0.003	0.86±0.03	4.43±0.05	0.68±0.01	8.6±0.2	0.36±0.01	97.2±1.4
Class 3	18.8±1.2	274±46	1.08±0.1	0.37±0.05	20.0±1.0	0.92±0.007	0.56±0.07	4.31±0.11	0.65±0.01	7.5±0.4	0.19±0.02	97.8±3.5

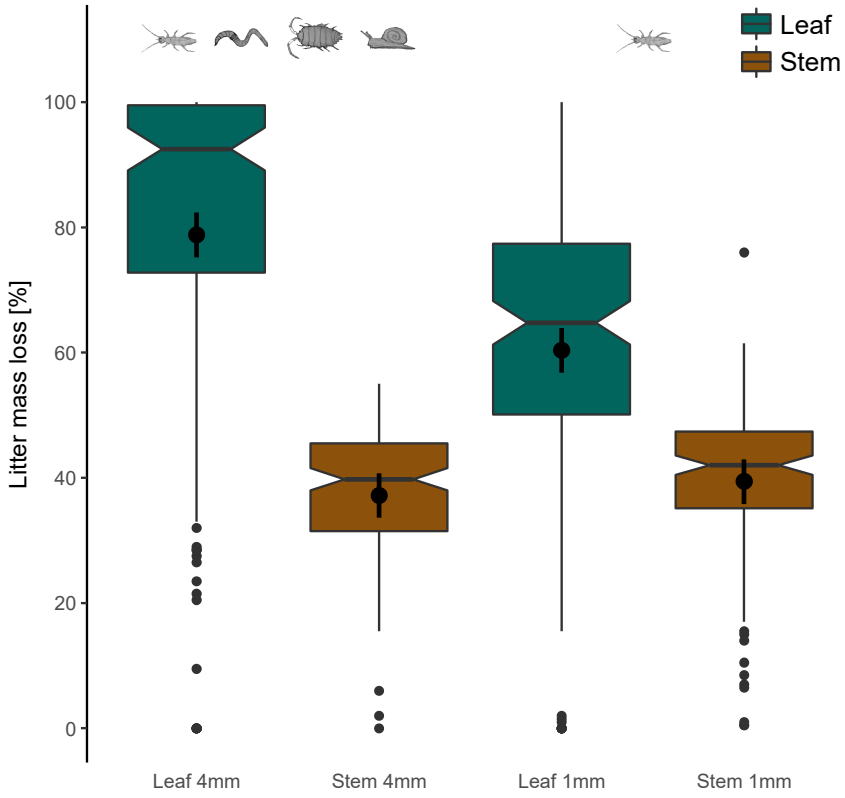


Figure 4.2 – Litter mass loss by litter type (leaf and stem of *Zea mays* L.) and mesh size (1 & 4 mm). Soil fauna drawings indicate sampled soil mesofauna (collembola) and macrofauna (earthworms, isopods, gastropods). Bold points are mean values of the simulated Bayesian inference posterior distribution with the 95% credible intervals as lines (cf. Table S 4.9). Colours correspond to litter types.

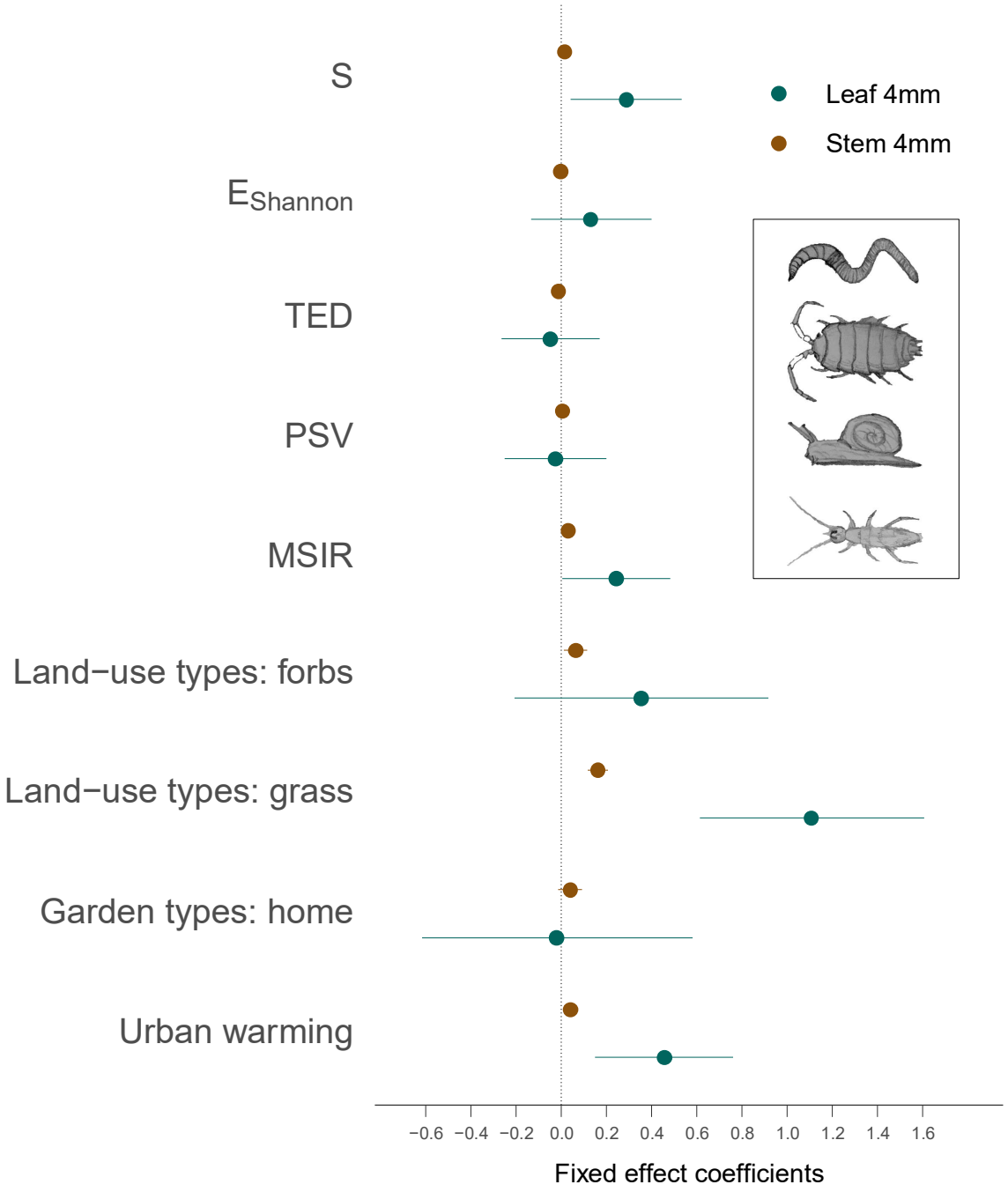


Figure 4.3 – Litter decomposition model fixed effect plots with 4 mm mesh size (see Figure S 4.8 for LMEM with 1mm mesh size). Points indicate mean values of simulated Bayesian inference posterior distribution with the 95% credible intervals as lines. Colours correspond to litter types.

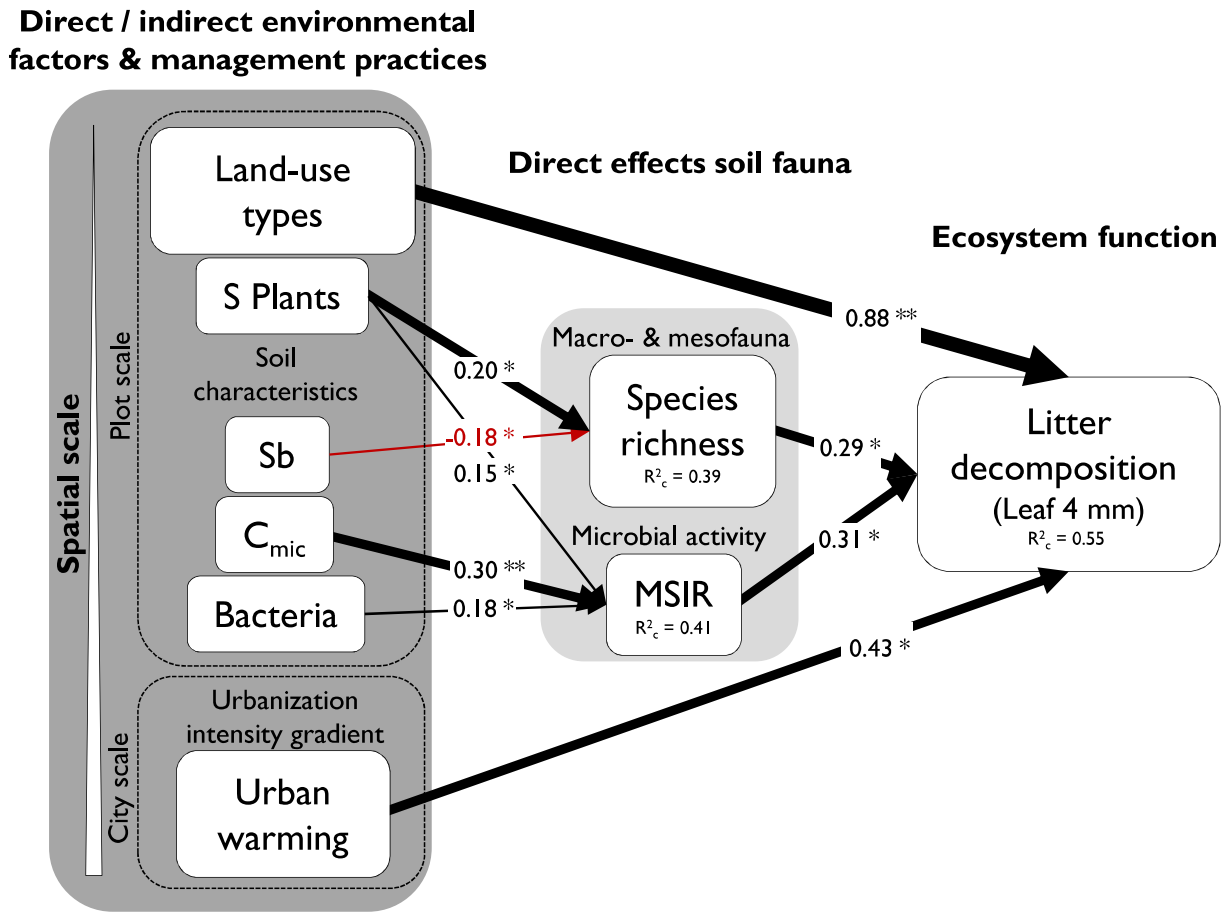


Figure 4.4 – Final SEM of direct and indirect effects of land-use features, soil characteristics and urbanisation intensity and direct effects of soil fauna on leaf litter decomposition. Arrows represent unidirectional relationships among variables. Black arrows denote positive and red arrows negative relationships. Variables not affecting decomposition significantly ($p>0.05$) are not included in this graphical representation (see Table 4.4). The thickness of paths has been scaled based on the magnitude of the standardised regression coefficient, given in the associated box. Conditional R^2, based on the variance of both the fixed and random effects, for component models are given in the boxes of response variables. C_{mic}: Microbial biomass carbon, MSIR: Multiple substrate-induced respiration of microorganisms, S plants: Plant species richness, Sb: Antimony content.

Table 4.4 – Multilevel SEM of litter decomposition (leaf 4 mm litter bags) investigating environmental factors on the soil fauna community and soil function decomposition (AICc=299.7, Fisher's C=124.2, P-value=0.96). Marginal R^2 based on fixed effects and conditional R^2 based on fixed and random effects. Significant paths are highlighted in bold. BD: Soil bulk density, C_{mic}: Microbial biomass carbon, $E_{Shannon}$: Shannon evenness, MSIR: Multiple substrate-induced respiration of microorganisms, N_{min}: Nitrogen mineralisation, PSV: Phylogenetic species variability, S: Soil fauna species richness, Sb: Antimony content, TED: Trait even distribution, TON: Total organic nitrogen, WHC: Water holding capacity.

Response	$R^2_{conditional}$	$R^2_{marginal}$	Predictor	Estimate	P	
Leaf 4 mm	0.55	0.24	**Land-use types: grass**	0.88 ± 0.3	0.002	**
			Urban warming	0.43 ± 0.2	0.01	*
			MSIR	0.31 ± 0.1	0.02	*
			S	0.29 ± 0.1	0.03	*
			$E_{Shannon}$	0.14 ± 0.1	0.35	
			Land-use types: forbs	0.22 ± 0.3	0.46	
			TED	-0.07 ± 0.1	0.57	
			PSV	-0.03 ± 0.1	0.79	
			Garden types: home	0.05 ± 0.3	0.89	
MSIR	0.41	0.39	**C_{mic}**	0.30 ± 0.09	0.002	**
			Bacteria	0.18 ± 0.07	0.02	*
			S plants	0.15 ± 0.07	0.04	*
			Bare soil	-0.12 ± 0.08	0.12	
			Sun hours	-0.11 ± 0.07	0.14	
			WHC	0.12 ± 0.08	0.14	
			BD	-0.11 ± 0.09	0.21	
			Management index	-0.06 ± 0.07	0.40	
S	0.39	0.19	**S plants**	0.20 ± 0.08	0.02	*
			Sb	-0.18 ± 0.08	0.03	*
			Bare soil	-0.15 ± 0.08	0.07	
			N_{min}	0.14 ± 0.08	0.08	
			MSIR	0.14 ± 0.08	0.09	
			Remove leaves	-0.12 ± 0.09	0.17	
$E_{Shannon}$	0.54	0.36	**Urban warming**	-0.46 ± 0.08	<0.001	***
			MSIR	-0.19 ± 0.07	0.008	**
			PSV	0.16 ± 0.07	0.02	*
			S	0.16 ± 0.07	0.02	*
			TED	-0.14 ± 0.07	0.04	*
			Management index	0.09 ± 0.07	0.21	
PSV	0.29	0.10	**Urban warming**	-0.24 ± 0.09	0.007	**
			Fertiliser	0.21 ± 0.09	0.02	*
TED	0.22	0.07	Land-use types: grass	-0.38 ± 0.2	0.06	
			S	0.11 ± 0.09	0.20	
			TON	0.10 ± 0.09	0.24	
			Urban warming	-0.10 ± 0.09	0.25	
			Land-use types: forbs	-0.01 ± 0.20	0.98	

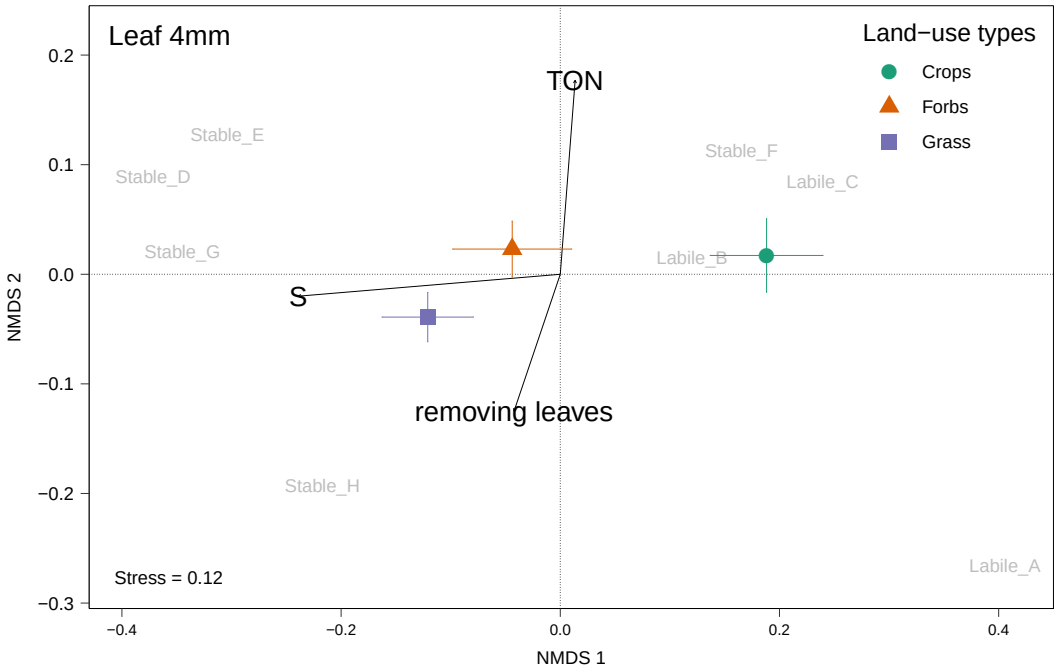


Figure 4.5 – Multivariate NMDS ordination of functional organic compounds of litter residues after decomposition (cf. Table 4.1). Leaf litter (4 mm mesh size) quality compounds are printed in grey. Variables significantly affecting the litter quality (cf. Table 4.5) are fitted on the ordination (arrows). Symbols and colours correspond to the garden land-use types (crops n=23, forbs n=27, grass n=26). Bold points correspond to mean values per land-use type and lines indicate standard errors. S: Soil fauna species richness, TON: Total organic nitrogen.

Table 4.5 – Multivariate analysis (PERMANOVA) of functional organic compounds of litter residues. Leaf litter (4 mm mesh size) quality compounds (n=76) were used in a Gower distance matrix and significant variables from the multilevel SEM as predictor variables. Significant variables affecting the litter quality are bold printed. BD: Soil bulk density, C_{mic}: Microbial biomass carbon, $E_{Shannon}$: Shannon evenness, MSIR: Multiple substrate-induced respiration of microorganisms, N_{min}: Nitrogen mineralisation, S: Soil fauna species richness, Sb: Antimony content, TED: Trait even distribution, TON: Total organic nitrogen, WHC: Water holding capacity.

Leaf 4mm		Df	F Model	R^2	P	
Soil Fauna	**S**	1	1.7	0.02	0.01	**
	$E_{Shannon}$	1	2.8	0.03	0.1	
	MSIR	1	0.5	0.01	0.7	
Soil characteristics	**TON**	1	1.6	0.02	0.01	*
	WHC	1	1.2	0.01	0.1	
	Sb	1	1.1	0.01	0.2	
	N_{min}	1	1.3	0.02	0.7	
	C_{mic}	1	0.8	0.01	0.1	
	BD	1	0.7	0.01	0.2	
	Bacteria	1	0.7	0.01	0.1	
Plot scale	**Land-use types**	2	3.4	0.08	< 0.001	***
	S plants	1	0.6	0.01	0.6	
	Bare soil	1	2.3	0.03	0.1	
	Sun hours	1	0.5	0.01	0.7	
Garden scale	**Remove leaves**	1	2.8	0.03	< 0.001	**
	Management index	1	1.2	0.01	0.3	
	Fertiliser	1	0.1	0.01	0.7	
	Garden types	1	0.2	0.01	0.6	
City scale	Urban warming	1	1.5	0.02	0.3	
	Residuals	55		0.66		

4.4 Discussion

One of the main challenges of sustainable urban development is reducing the rate of urban soil sealing (Artmann 2016), which is steadily increasing at the expense of highly contested green spaces such as gardens (Tappert et al. 2018). In European cities, green spaces are declining regardless of whether the urban population is shrinking or growing (Kabisch and Haase 2013). In this context, we aimed to contribute to the discussion on the importance of urban gardens for the biodiversity of a city and to investigate which and how biotic and abiotic factors drive BEF relationships. In the present study, we used urban gardens as model system to explore above and belowground BEF relationships in a human dominated environment and investigated direct and indirect effects on soil organisms and litter decomposition at different spatial scales. Our results showed that decomposition was highest when soil macrofauna was involved. This is in line with the results that soil fauna diversity increases decomposition across biomes found by the study of Handa et al. (2014). Overall, our results showed that the way gardeners manage land-use types and how the surrounding urban matrix is composed, can have an important effect on litter decomposition.

4.4.1 Soil fauna drives litter decomposition also in human dominated ecosystems

Environmental conditions such as climate or soil characteristics together with litter quality, are the main influencing factors of decomposition across biomes, with the decomposer community contributing substantially (García-Palacios et al. 2016b). In this study, we used four major taxonomic soil fauna groups and calculated indices of taxonomic, functional and phylogenetic biodiversity representing different facets of soil fauna biodiversity to assess direct effects on litter decomposition. Regarding the litter bags including all soil fauna, we observed higher decomposition rates for the better decomposable leaf litter type. Looking at macrofauna only, we observed generally higher litter decomposition rates for the better decomposable leaf litter type. This could be explained partially by the food preference of macrofauna, which will first feed on the most palatable available litter, and only consume litter of low nutritional value later when other resources are not available and once microorganisms have increased its palatability through their presence and the degradation of recalcitrant compounds such as lignin (Vos et al. 2011). In our study, taxonomic diversity of soil fauna species was the driving diversity aspect affecting litter decomposition on both the LMEM and the SEM (Figure 4.3,4.4). In relation the mass-ratio hypothesis, claiming that ecological processes are driven by the traits of the most abundant species within the community (Grime 1998), we tested the effect of soil fauna abundance on litter decomposition (Table S 411.), but no significant effects were detected and the model goodness-of-fit tests decreased. Moreover, functional and phylogenetic diversity did not explain litter mass loss, although it has been shown that species identity is important for the decomposition process, as functionally different groups of macrofauna can interact positively (Heemsbergen 2004). Nevertheless, functional diversity is usually highly correlated with taxonomic diversity in most BEF investigations (Gessner et al. 2010), but maybe not in very complex soil fauna communities. In our study, we used four taxonomic groups of soil fauna, which occur ubiquitously and are phylogenetically distant from each other. This may have resulted in similar phylogenetic and functional variation among soil fauna communities, because differences in species functional traits or phylogenetic indices were highest between the investigated taxonomic groups.

4.4.2 Garden land-use and management intensity affect decomposition mediated by soil fauna

Garden land-use management practices, such as soil tillage or planting vegetables are important factors of soil biodiversity in cities (Beninde et al. 2015), including less mobile soil invertebrates (Braaker et al. 2014). Likewise, in our study, management of land-use types was also the dominant factor for litter decomposition: regardless of litter type or mesh size. For instance, 83 % of all 85 gardeners used compost for their crops. The use of compost has been reported as a moderate and relatively cost-effective management practice against urban soil problems such as soil compaction, lack of organic matter, or heavy metal pollution (Lusk and Toor 2018). An adequate addition of good quality compost has been shown to have positive effects on several soil properties such as bulk density, porosity, aggregate stability, water holding capacity or infiltration (Cogger 2005). In our study, adding compost had a positive effect on decomposition rates, while major soil disturbance, such as digging or loosening the soil, reduced decomposition rates across leaf litter type and mesh size. The use of pesticides was not a significant factor affecting decomposition (Table S 4.9). This could be

influenced by several factors, such as the types and amounts of pesticides used, which were not investigated here. Nevertheless, a study of the effects of pesticides on soil fauna diversity, soil quality or food quality would be of great interest, as many gardeners are not aware of the negative consequences on soil and its diversity (Zaller 2018).

In addition to management practices, soil characteristics may also contribute to the effect of aboveground biota on decomposition processes in urban gardens. An example is a reciprocal litter decomposition experiment in the city of Lahti, Finland, in which soil type was one of the main factors affecting decomposition (Vauramo and Setälä 2011). In our study, we observed that several soil properties affected the soil fauna community. Soils with higher amounts of microbial biomass and bacteria had a higher microbial activity (Figure 4.4), thus indirectly increased litter decomposition (Table 4.4). Despite the importance of fungi in decomposition (e.g. Seastedt 1984, Kabuyah et al. 2012), we only found effects of bacteria on microbial activity (Figure S 4.13, Table S 4.10) increasing indirectly decomposition of the less recalcitrant litter material (Table S 4.1). In fact, similar results have been found by Girvan et al. (2004), who showed that soil management practices such as fertilisation or the use of pesticides affected bacterial but not fungal communities. Furthermore, the soil heavy metal antimony content showed a negative effect on soil fauna species richness and plant diversity (Figure S 4.11) had a positive effect on soil fauna species richness and microbial activity. This is in line with the study by Ebeling et al. (2018), in which higher plant species richness supported more diverse and complex arthropod communities: thus affecting ecosystem services in grassland ecosystems. Therefore, we can conclude that garden management, for instance how many plant species are planted, not only influences soil properties, but also indirectly influences litter decomposition via the change of the soil fauna communities.

4.4.3 Effects on litter residue quality

Litter residues were analysed to show effects of biotic or abiotic factors on the litter quality, complementary to the litter mass loss analysis. MidDRIFTS peak measurements were selected to represent contrasting functional organic compounds of the remaining litter after decomposition (Table 4.1). We showed that soil fauna not only had a significant positive effect on litter decomposition but also determine litter residue quality. Sites with higher soil fauna species richness had more stable organic compounds (Figure 4.5), while crop sites had more labile compounds left after the assessed decomposition period. Thus, a changed soil fauna community composition not only affected the mass loss but also the consumption of different organic compounds of the leaf litter. Interestingly, the management practice 'leaf removal' resulted in a distinct grouping of sites regarding the litter residue quality. This indicates that at sites, where leaves were removed in winter, soil fauna were either less abundant because of restricted food resources, had a different community structure, or preferred the organic material of the litter bags.

4.4.4 Effects of garden land-use type and urban warming on litter decomposition

We hypothesised that garden land-use type and garden scale features will have a larger effect on soil biodiversity and litter decomposition than city scale factors. Our results highlighted an influence of garden land-use type features and soil characteristics on soil fauna and decomposition. Surprisingly, urban warming strongly positively affected litter decomposition and also shaped the soil fauna community (Table 4.4). This demonstrated that soil invertebrates are also influenced by large scale urban intensity gradients, similarly to previous findings on mobile species, such as bees (Pardee and Philpott 2014). In this study, we found decreasing species richness but increasing abundance of soil fauna species across taxa, mainly because of the more abundant isopods in more urbanised gardens. Although most taxonomic groups of urban fauna occur at higher densities in rural environments than in non-rural ones, few species can be attributed as urban exploiters or even urban adapters (Evans 2010). This pattern of population density depends on the species identity. For instance, crustacea such as isopods are dependent on the availability of calcium, as they frequently replace their cuticle (Fabritius and Ziegler 2003), but also shelled gastropods rely on the calcium levels (Charrier et al. 2013). Carbonate in soils lead to moderate alkaline pH values, as found in this study with a mean pH value of 7.3 ± 0.02. The pH is expected to increase with urbanisation, due to calcium-rich materials used for construction (Kida and Kawahigashi 2015), but also due to precipitation depositions caused by anthropogenic emissions (Blume et al. 2016). In our study soil pH did not increase with urbanisation (Figure S 4.14, Table S 4.10). This can be explained by the overlying effect of garden management practices (e.g. liming), leading to increased pH values in annual vegetated sites. However, it is difficult to quantify the pure effect

of urban intensity on soil fauna due to the complex structure and functioning of urban ecosystems (McDonnell et al. 1997) and because other factors, such as management practices or pollution can also influence soil functions such as decomposition (Pouyat et al. 1997).

Studies investigating the effect of urbanisation on litter decomposition (Table 4.6) revealed mixed effects with six out of twelve studies indicating a positive effect on decomposition with increasing urbanisation, while two showed no trends and four showed negative effects. This uncertainty associated with the effect of urbanisation on decomposition is likely due to the large variety of anthropogenic disturbances associated with different definitions of urbanisation and the strong local impact of management (Enloe et al. 2015). The quality of urban litter can be reduced by environmental pollutants affecting plant growth and leaf senescence (Carreiro et al. 1999). Alternatively, urban warming may accelerate metabolic processes and thus decomposition rates (Pouyat and Carreiro 2003). Indeed, we found a positive association between urban warming and microbial activity (Table 4.3; Table S 4.9).

However, a comparison of decomposition studies across cities should be done with caution due to differences of the abiotic environment, experimental design or also in the site conditions such as contrasting soil types (Berg and McClaugherty 2003). Almost all investigations shown in Table 4.6 have been done on urban forests with litter bags of different tree species including mixtures of them. The main reason for a positive effect of urbanisation on decomposition were higher temperatures either in soil (Pouyat and Carreiro 2003, Pouyat et al. 1997) or air at urban sites (Nikula et al. 2010). Studies with negative effects of urbanisation on decomposition were mainly driven by heavy metals (Cotrufo et al. 1995, Inman and Parker 1978), as they are known to reduce decomposer activity (Bååth 1989). Moreover, urban soils in Kiel and Rostock, Germany, were found to have higher aromatic compounds which reduced decomposition but increased SOC with urbanisation (Beyer et al. 2001). Finally, non-native plants in urban forests can increase decomposition rates if they replace native plants which decompose more slowly (Ehrenfeld 2003). The positive effects of urban warming on aboveground litter decomposition confirmed the results of a former study with tea bags as a measure of belowground decomposition (Tresch et al. 2018a). Taken together, our results suggest that the increased decomposition with urbanisation intensity can be explained by combined effects of higher microbial activity, more soil moisture through watering or ecological gardening activities (e.g. cover crops, mulch or compost), and a changed soil fauna composition, such as more gastropod species and higher abundance of isopods (Table S 4.4).

Table 4.6 – Literature review of litter decomposition studies and urbanisation intensity. Main global climate groups according to Köppen-Geiger climate classification (Kottek et al. 2006), a: warm temperate, fully humid, hot summer, b:warm temperate, fully humid, warm summer; c: warm temperate, summer dry, hot summer; d: snow, fully humid, warm summer; e: snow, fully humid, hot summer. Tea bag index decomposition as reported in Tresch et al. (2018a) with mean values for urban (Urban warming class 3[1]), suburban (class 2[2]) and rural (class 1[3]) areas. Significant levels can be found in Table S 4.9.

Litter type	Litter bag size [cm]	Mesh size [mm]	Experiment	Habitat types	Litter mass loss			Effect	Main drivers	City	Climate	References
					Urban [%]	Suburban [%]	Rural [%]					
Acer saccharum	15×15	1.7	Litter bag	Forest	76	68	40	↑	Soil temperatures, earthworms	New York, US	a	Pouyat et al. (1997)
Red oak	15×15	1.7	Litter bag	Forest	74.4± 3.5	-	53.5± 3.3	↑	Soil temperatures, earthworms	New York, US	a	Pouyat and Carreiro (2003)
Populus tremula	12×14	1.5	Litter bag	Forest	19-21 % greater	-	-	↑	Temperatures, humus, pH, N deposition	Helsinki, FI	d	Nikula et al. (2010)
Fagus silvatica	12×15	2	Litter bag	Forest	88.7± 1.4	86.7± 1.1	86.1± 0.9	↑	Soil moisture	Basel, CH	b	Melliger et al. (2017)
Oak & pine mixtures	30.5×45.7	6	Litter bag	Forest	50.5± 8.5	48.8± 3.7	65.5± 6.1	-	Human population density, drought	Eastpoint, US	a	Enloe et al. (2015)
5 local tree species	15×15	1.4	Litter bag	Parks	-	-	-	-	Leaf litter quality	Hamburg, DE	b	Dorendorf et al. (2015)
Black oak	5.5×5.5	3	Litter bag	Forest	8	-	16	↓	HM contamination in urban litter	Chicago, US	e	Inman and Parker (1978)
Quercus ilex	5.5 ⌀	2	Lab. microcosm	Forest	29	40		↓	HM contamination	Naples, IT	c	Cotrufo et al. (1995)
Quercus rubra	Discs 6 mm ⌀	-	Lab. vials	Forest	25.9± 1.3	31.1± 1.0	34.4± 0.9	↓	Decreased litter quality in urban litter	New York, US	a	Carreiro et al. (1999)
Quercus prinus	15×15	1	Litter bag	Forest	62.1	65.4	66.2	↓	Soil moisture, SOM	Asheville, US	a	Pavao-Zuckerman and Coleman (2005)
Green tea	Tea bags	0.25	Tea bag index	Gardens	60.8± 1.3[1]	59.4± 0.5[2]	57.3± 0.7[3]	↑	Temperatures, microbial activity	Zurich, CH	b	Tresch et al. (2018a)
Zea mays leaves	18×18	4	Litter bag	Gardens	91.9± 2.8[1]	79.9± 2.8[2]	70.9± 5.3[3]	↑	See discussion section	Zurich, CH	b	Present study
Zea mays stems	18×18	4	Litter bag	Gardens	43.7± 1.5[1]	38.1± 1.0[2]	33.7± 2.1[3]	↑	See discussion section	Zurich, CH	b	Present study
Zea mays leaves	18×18	1	Litter bag	Gardens	73.4± 5.1[1]	59.4± 2.4[2]	53.6± 6.1[3]	↑	See discussion section	Zurich, CH	b	Present study
Zea mays stems	18×18	1	Litter bag	Gardens	52.2± 3.3[1]	40.0± 1.3[2]	34.3± 2.8[3]	↑	See discussion section	Zurich, CH	b	Present study

4.4.5 Study limitations and perspectives

Our study design maximised the numbers of sites under actual garden management in one city with the aim of investigating direct and indirect effects of biotic and abiotic factors on litter decomposition at different spatial scales, as suggested by Bradford et al. (2016). On the one hand, the use of two litter types, one easily decomposable (leaf parts of *Zea mays* leaves) and one more difficult to decompose (stem parts of *Zea mays* leaves) allowed for a standardised and constant litter quality (Pouyat et al. 1997). On the other hand, the effect of multiple litter types on decomposition (e.g. Hättenschwiler and Gasser 2005) could have contributed to a more general picture of litter decomposition since litter mixtures are more realistic representations of the decomposition process. Further, litter mixing can also influence decomposer organisms with complex interactions and regulate decomposition (Hättenschwiler and Gasser 2005). Our assumption that litter bags with a mesh size of 1 mm are sufficient to exclude smaller macrofauna might not have been optimal, since Pouyat and Carreiro (2003) found that juvenile earthworms contributed significantly to leaf decomposition in litter bags with 1.7 mm mesh size. Regardless of the mesh size, juvenile species are often a problem in decomposition studies since they can hardly be determined to the species level. In addition, other soil fauna taxa affecting decomposition directly such as mites (Siepel and Maaskamp 1994) or indirectly such as nematodes (García-Palacios et al. 2017) could have impacted leaf litter decomposition. Future studies should consider the variance of functional and phylogenetic indices within taxonomic groups and select traits based on their sensitivity to the focal environmental gradient and known effect to the target ecosystem function. Although body size and vertical stratification are reasonably connected with decomposition rates (Bardgett and Wardle 2010, Briones 2014), other traits related to the feeding habits of soil fauna species or habitat preferences would have been of great interest for their effects on decomposition. However, such traits are still missing for many meso- but also macrofauna species.

4.5 Conclusion

There is great private and public interest in how urban gardens enhance urban biodiversity in cities and how different management strategies are modifying this relationship (Ossola et al. 2018), for they deliver a range of ecosystem services to urban residents (Goddard et al. 2010). Our city-wide litter decomposition experiment revealed the importance of the interactions between the management of land-use types, urbanisation intensity and soil fauna dynamics. With a multilevel SEM, we highlighted direct and indirect effects of soil fauna , land-use, and soil characteristics on litter decomposition. For example, we showed that plant species richness indirectly influenced litter decomposition through increasing soil fauna species richness and microbial activity. This demonstrates for the first time that belowground BEF relationships in urban gardens are influenced by management and urbanisation intensity gradients. With this multi-indicator evaluation at different spatial scales, we emphasised the importance of both local and city scale factors on litter decomposition. The multivariate analysis of litter residue quality confirmed the importance of soil fauna species richness and land-use type management on litter decomposition and was therefore useful to go beyond litter mass loss to better understand decomposition processes. Future experiments on trait based BEF relationship in urban ecosystems are needed (Schwarz et al. 2017) to improve our mechanistic understanding about management impacts on urban green spaces, since robust knowledge about urban BEF relationships is essential to improve current and future practices in urban planning and management.

Acknowledgements

This study is part of an interdisciplinary project that focuses on biodiversity, soil quality, ecosystem services and human well-being of urban gardens (www.bettergardens.ch). It was supported by the Swiss National Science Foundation in frame of the Sinergia program (Grant no. CRSII1_154416). Further, we would like to thank the coordinator of the project BetterGardens Dr. Robert Home for his support and corrections. We are grateful for the help in the lab by Adolphe Munyangabe, Anton Kuhn, Lena Fischer (earthworms), Reto Henzmann (soil samples, earthworms), Bernhard Stehle (earthworms and microbial measurements), Dr. Lukas Pfiffner (earthworm identification), Selina Gugelmann (gastropod identification), Dr. Daniel Haefelfinger (litter decomposition), Dr. Scott Demian (midDRIFTS analysis) and Giulia Benazzi for their extraordinary support and help in the field or laboratory. Special thanks to Dr. Simone Fontana for providing the R-codes for TOP & TED calculations as well as to Dr. Jörg Salamon (collembola identification) and Ferenc Visilics (isopod identification). We acknowledge in particular the participation of the 85 gardeners, who provided access and let us

dig and sample soils in their gardens.

doi.org/10.1016/j.scitotenv.2018.12.235

Supplementary data

Supplementary data to this article including the raw data and data manipulation and statistical computations in an R project can be found online at

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Science of the Total Environment, Volume 658, 25 March 2019, Pages 1614-1629

4.6 Supplementary Tables and Figures

4.6.1 Supplementary Figures

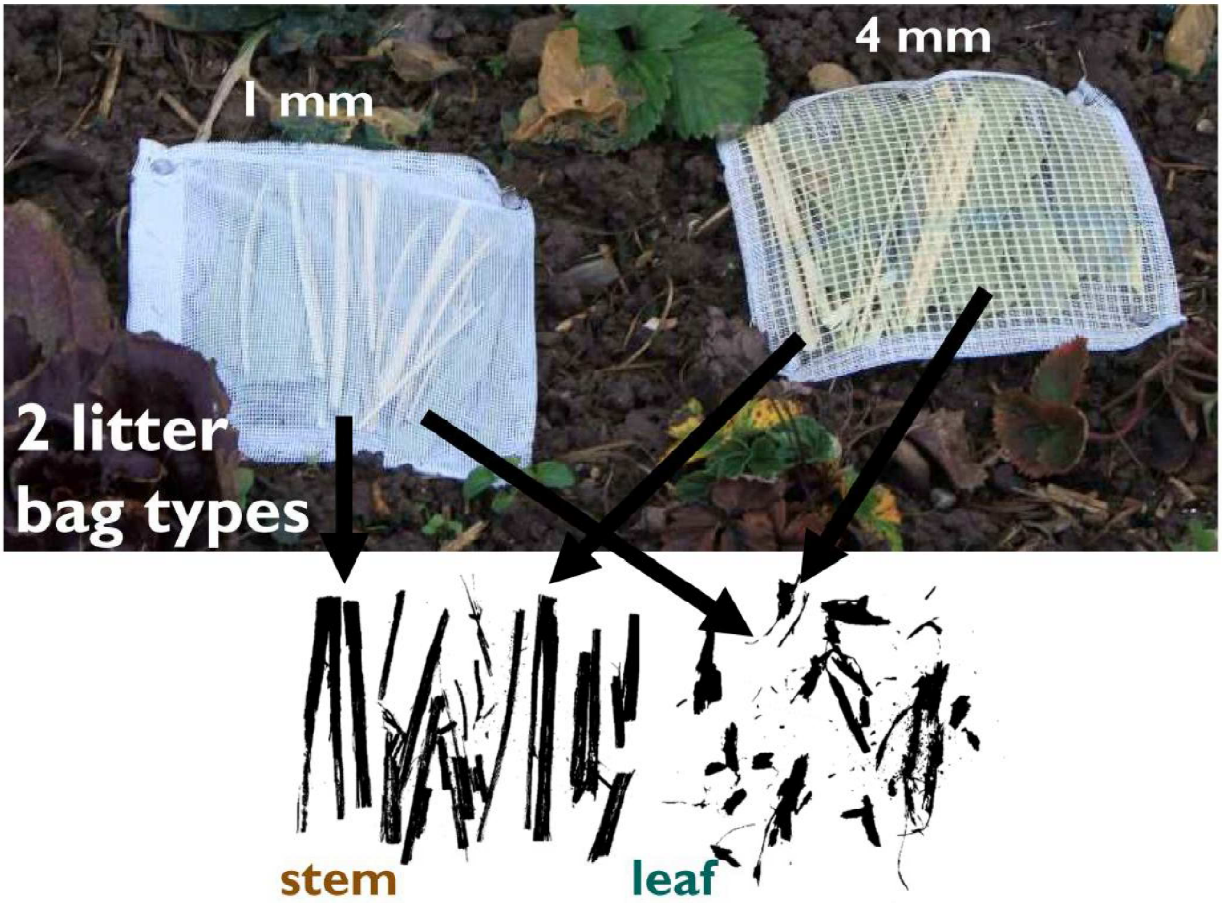

Figure S 4.1 – Litter bag types used for the decomposition study. Litter bags (18 cm x 18 cm) were constructed in accordance with Finerty et al. (2016), containing two mesh sizes of 1 mm and 4 mm, to evaluate the contribution of macrofauna decomposition. A fine mesh (1 mm) was used on the bottom for both litter bags to avoid loss of litter material. In addition, two types of litter sources were used to see effects of soil fauna on contrasting litter traits. Oven dried (40°C) leaf and stem material of *Zea mays* (only top 30 cm of the plant leaves) were used as litter materials. An amount of 2 ± 0.01 g of each leaf and stem pieces (16 ± 1 cm length) have been placed into the litter bags.

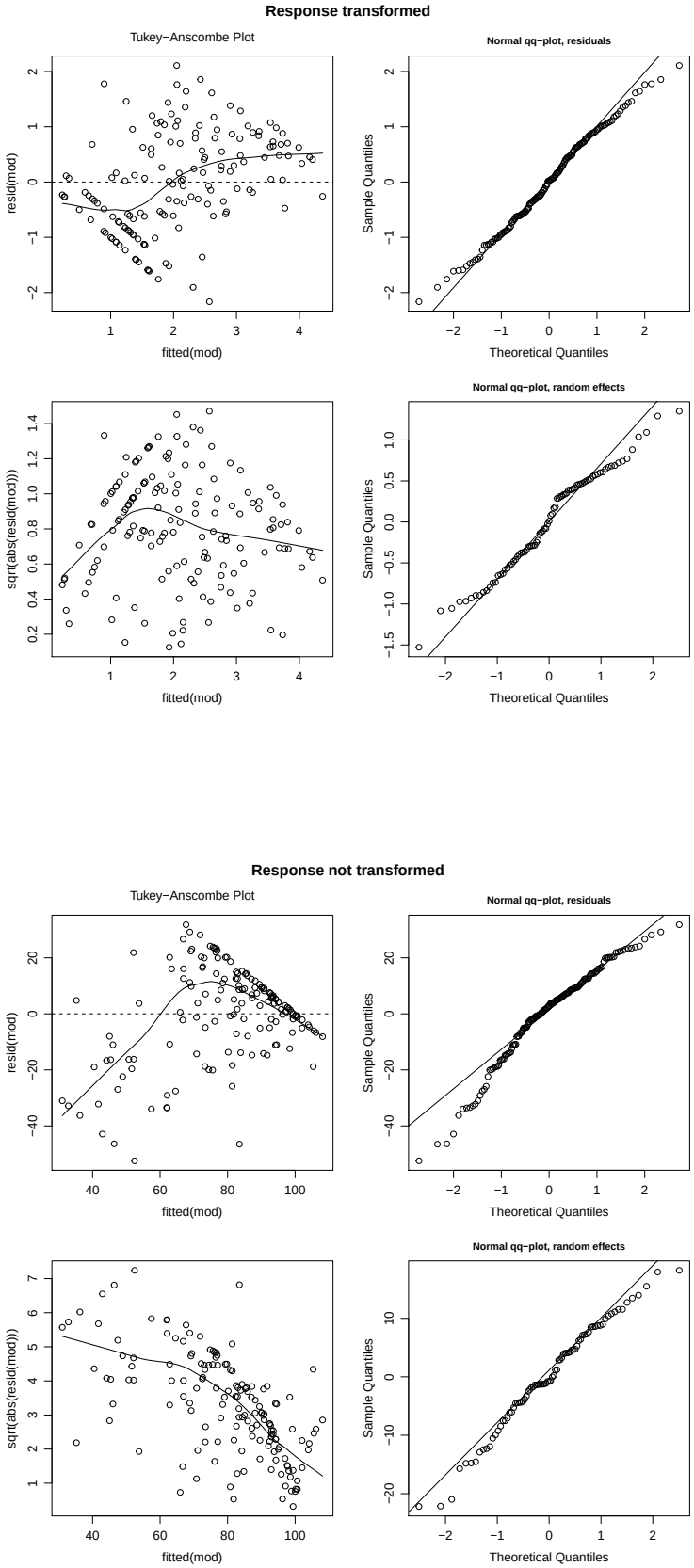

Figure S 4.2 – Diagnostic residual and random effect plots for the assessment of model assumptions. Transformed model: $lmer(log(100 - response + 1) \sim MSIR + S + E_{\text{Shannon}} + PSV + TED + urban\_warming + land - use\_types + garden\_types + (1|Garden\_ID), REML = F)$. Upper left: residuals versus fitted values. Upper right: Normal QQ plot of the residuals. Lower left: square-root of the absolute values of the residuals versus fitted values. Lower right: Normal QQ plot of the random effects.

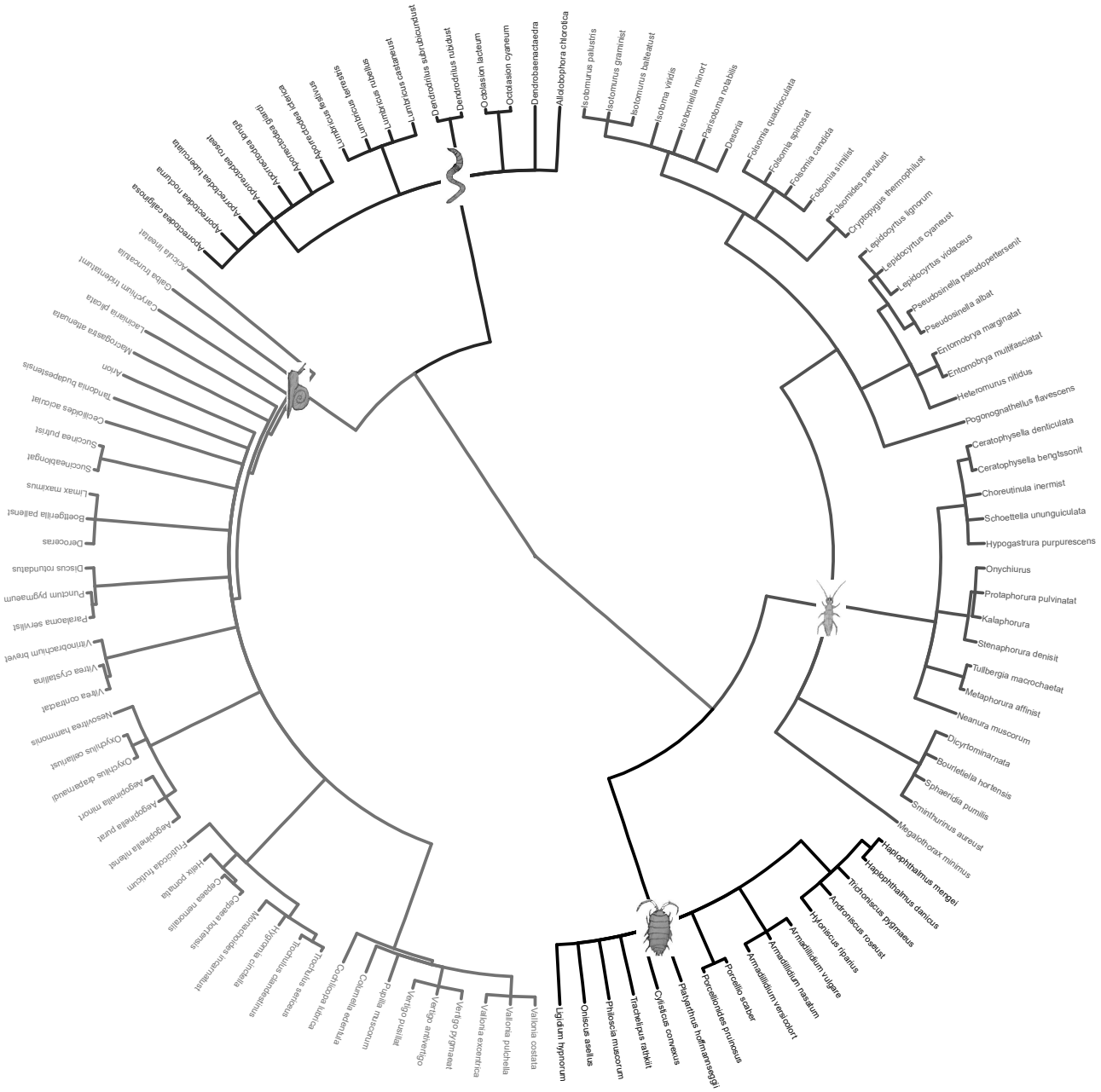

Figure S 4.3 – Phylogenetic tree of decomposer organisms based on the open tree of life project (Hinchliff et al. 2015). Pictures symbolise the four broad taxonomic groups: collembola (39 species), isopods (16 species), gastropods (42 species) and earthworms (18 species), representing the decomposer fauna in this investigation.

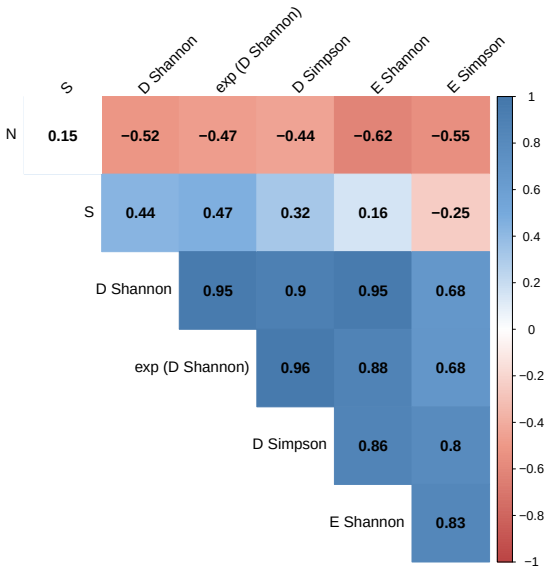

Figure S 4.4 – Pearson's correlation coefficient matrix of taxonomic diversity indices. Non-significant correlations are left blank (P<0.05) calculated with a modified version of the 'corrplot' package (Wei and Simko 2017). Species richness (S) and Shannon evenness (E Shannon) were selected to represent species richness and evenness in this study.

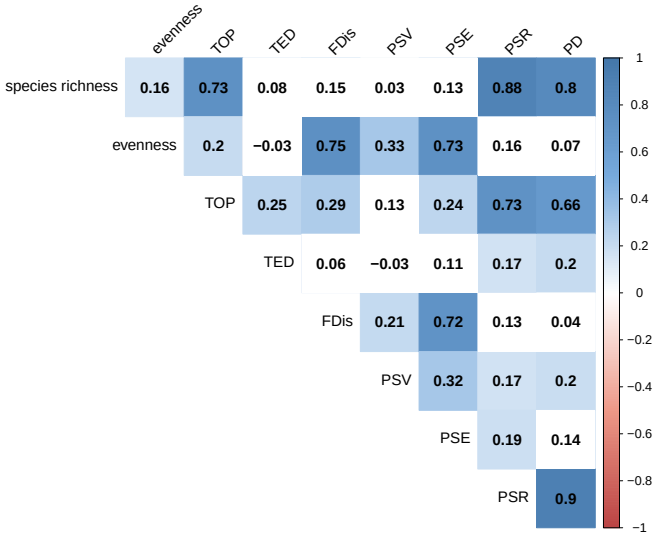

Figure S 4.5 – Pearson's correlation coefficient matrix of taxonomic (species richness and evenness), functional (TOP, TED and FDis) and phylogenetic (PSV, PSE, PSR and PD) diversity indices. Trait even distribution (TED) and phylogenetic species variability (PSV) were selected together with species richness and evenness, due to the lowest correlation coefficient in order to avoid collinearity issues (Dormann et al. 2013). Non-significant correlations are left blank (P<0.05).

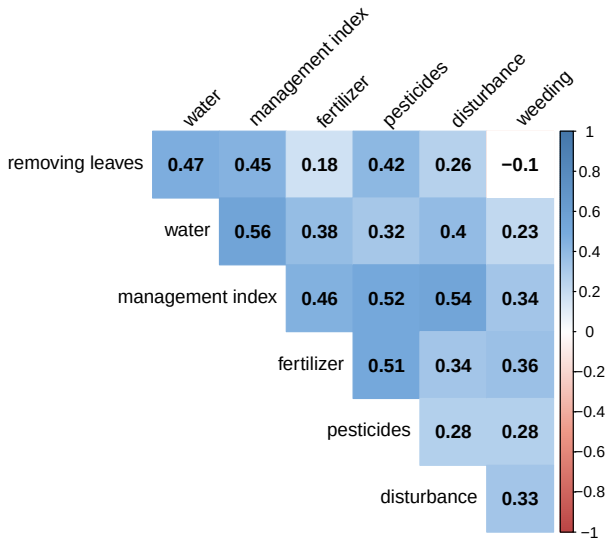

Figure S 4.6 – Pearson's correlation coefficient matrix of garden management variables. All management questions can be seen in Table S 4.6. Management variables has been asked individually per land-use type. Water = frequency of applying water (WaterLawn, WaterVeg, WaterFlower), management index = sum of all management variables ordered from low to high intensity each on a five level Likert scale, fertiliser = frequency of applying fertiliser (FertLawn, FertGrass, FertVeg, FertFlower), pesticides = frequency of applying pesticides (PestLawn,PestVeg,PestFlower,PestTrees,WeedingHerbicide), disturbance = frequency of soil disturbances (DiggingVeg, DiggingFlower, CareLawn), weeding = frequency of weeding (Weeds). Non-significant correlations are left blank (P<0.05).

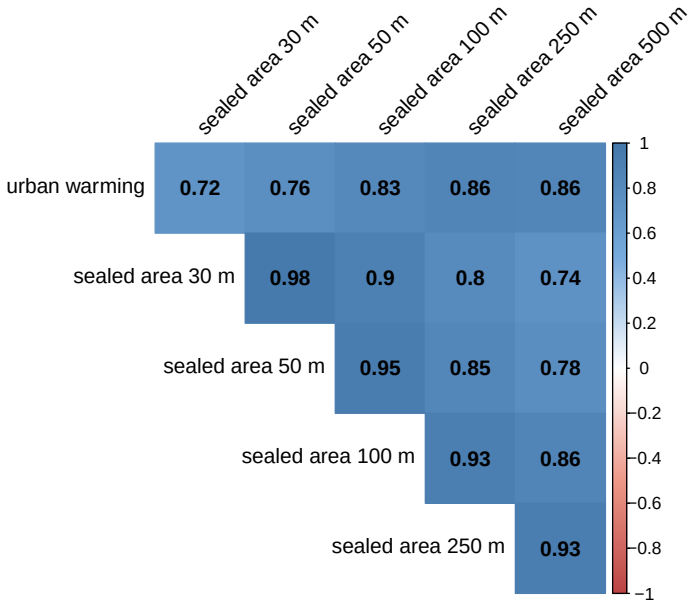

Figure S 4.7 – Pearson's correlation coefficient matrix of urban warming and sealed area. Urban warming is a measure of the local deviation of average night temperatures near surface in the city of Zurich. It is derived from a regional climate model by Parlow et al. (2010) and consists of six categories from 0 to + 6 °C. The sealed areas are the sum of sealed and built area around each garden with five radii (30, 50, 100, 250, 500 m) obtained in ArcGIS.

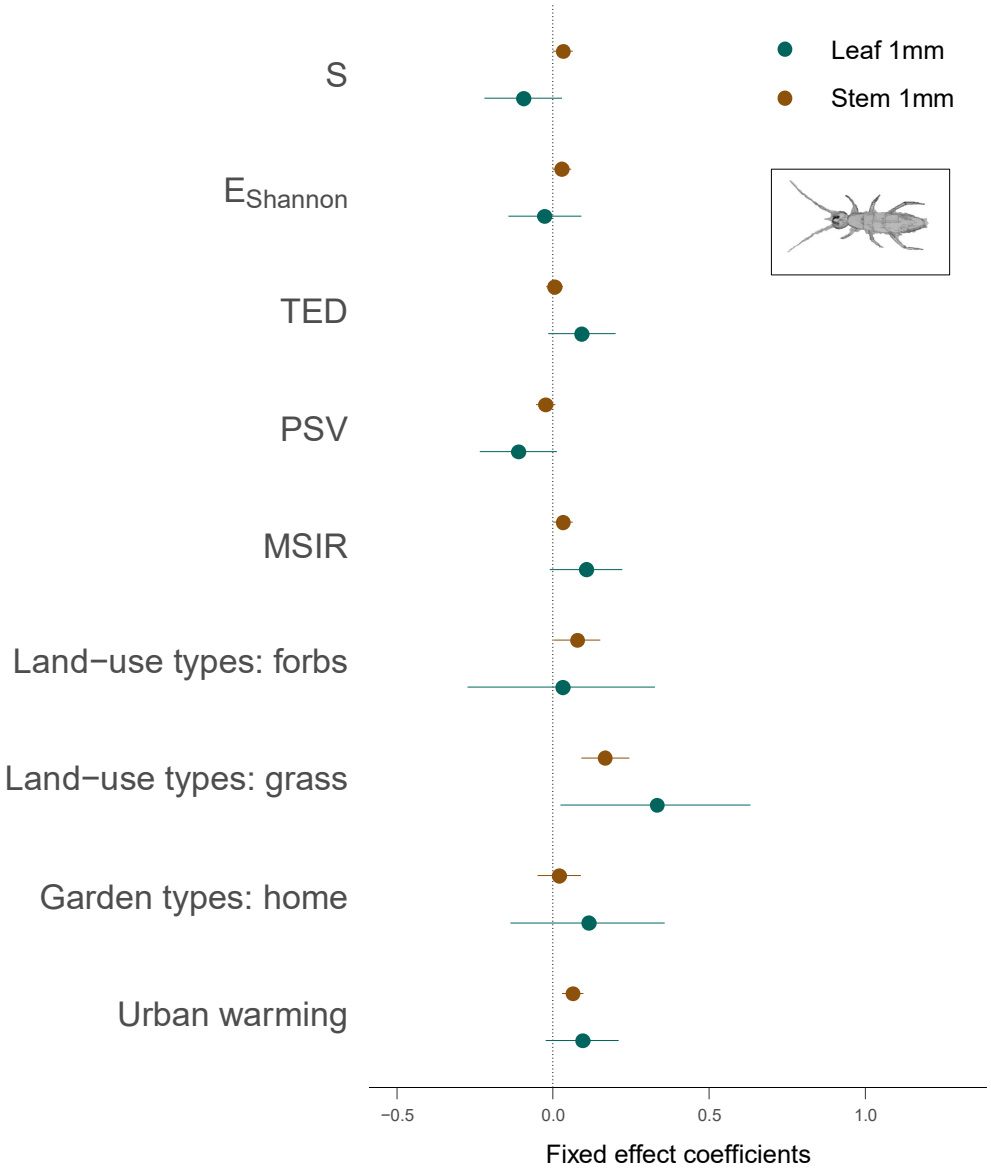

Figure S 4.8 – Litter decomposition model fixed effect plots with 1 mm mesh size. Points indicate mean values of simulated Bayesian inference posterior distribution with the 95 % credible intervals as lines. Colours correspond to litter types.

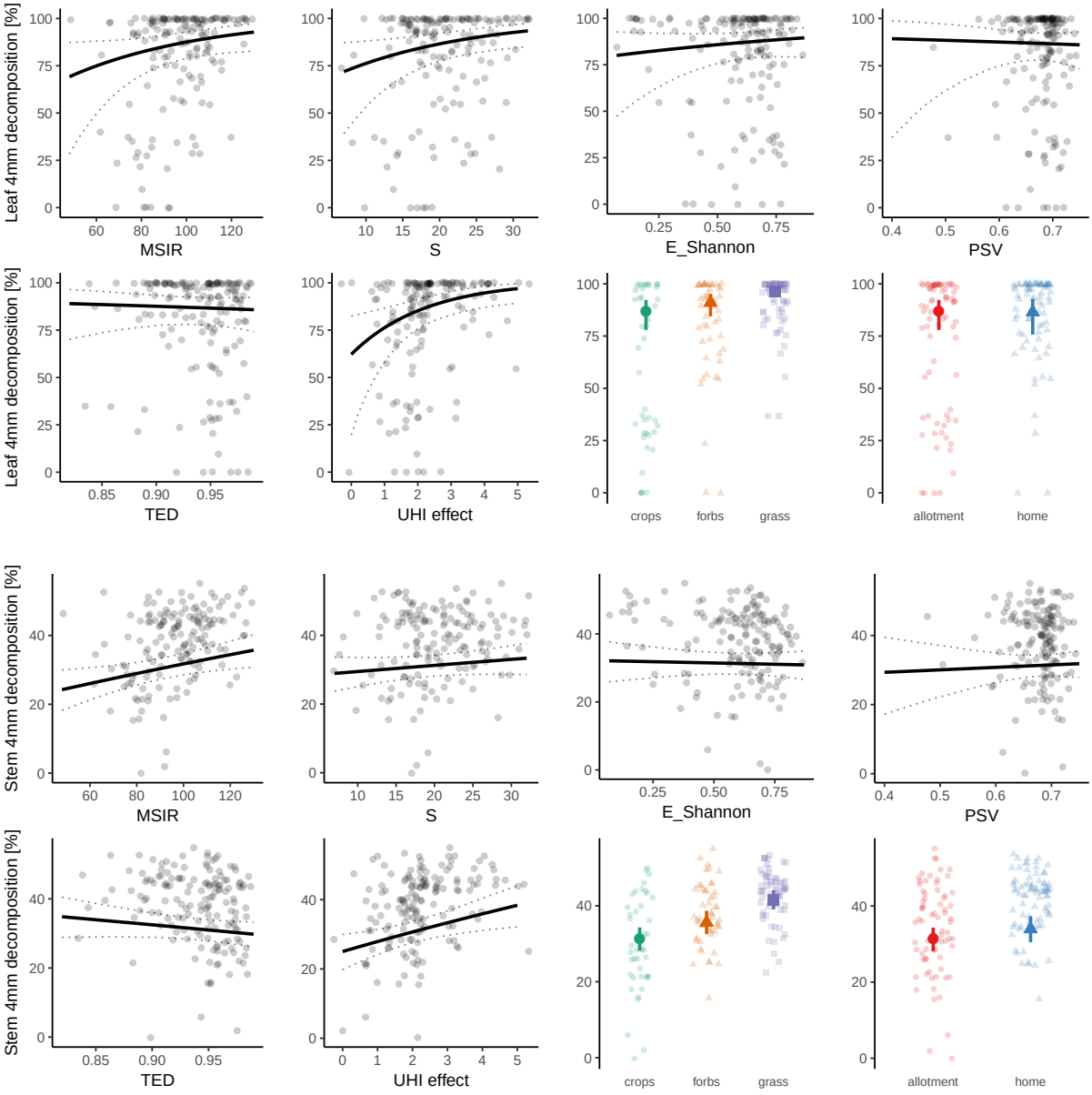

Figure S 4.9 – Effect plots of litter decomposition model showing the fixed effects microbial activity (MSIR), soil fauna species richness (S), species evenness (E_Shannon), phylogenetic species variability (PSV), trait even distribution (TED), urban warming, garden land-use types and garden types. Solid lines or bold points are fitted values of the simulated Bayesian inference posterior distribution taking into account the random effect of garden identity with the 95% credible intervals as dotted lines.

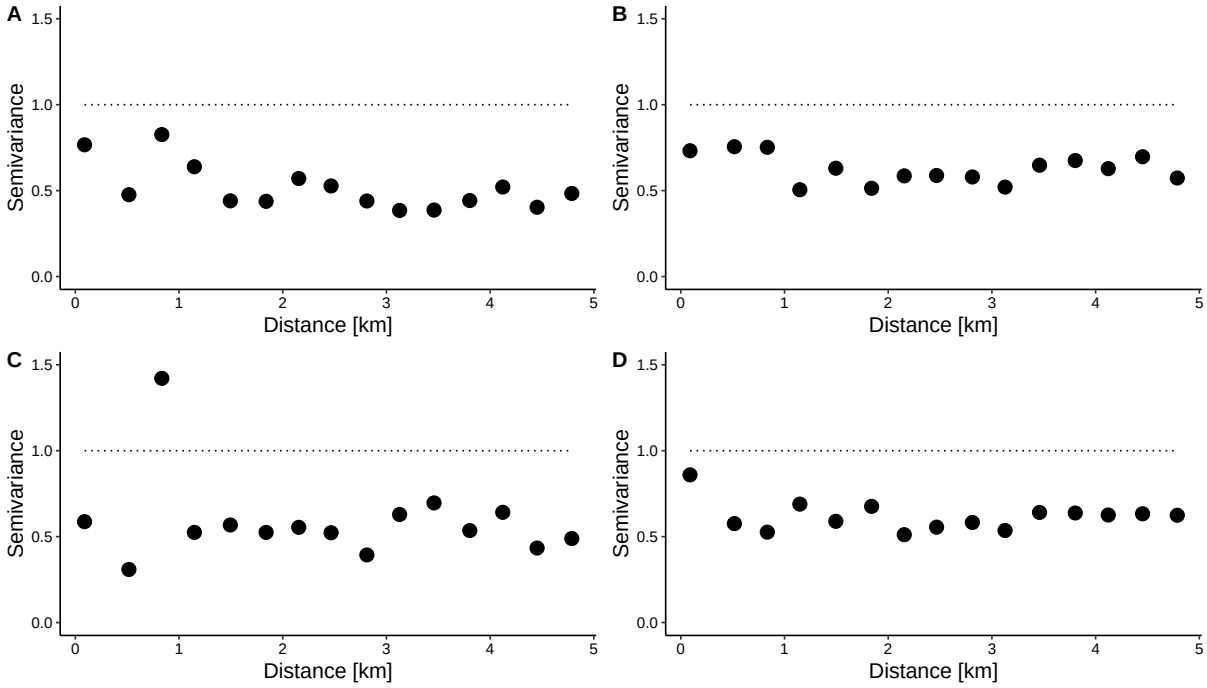

Figure S 4.10 – Semivariograms of LMEM residuals from the response variables of the decomposition model: leaf 4 mm (A), the model with the microbial activity: MSIR (B) and the model with plant species richness (C). Semivariances (0.5 times the mean squared differences between sites) were computed with the R package 'gstat' (Pebesma 2004). In all plots values are close to 1 and show no clear increase or decrease patterns of spatial autocorrelation, indicating that the residuals are not more similar or dissimilar to each other than expected by chance (Korner-Nievergelt et al. 2015). In addition, the calculated Moran's I autocorrelation index (Paradis 2018) for the response variable leaf 4 mm was not significant (p=0.26; observed=-0.01±0.004, expected=-0.007) as well as for the response variable MSIR (p=0.09; observed=-0.013±0.004, expected=-0.007) and the response variable plant species richness (p=0.65; observed=-0.008±0.004, expected=-0.007).

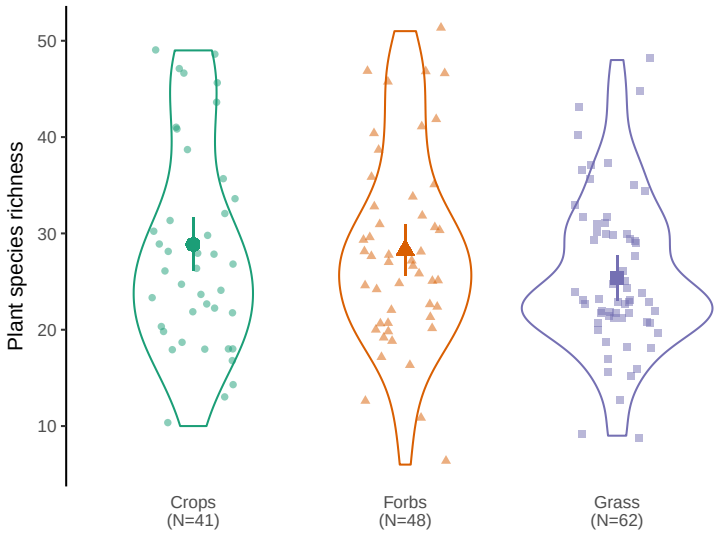

Figure S 4.11 – Plant species richness per garden land-use type assessed as the sum of all cultivated and spontaneously growing plants per urban garden study site. Sampling and methodology of identification including the complete species list can be found in (Frey and Moretti 2019). Bold points represent mean values of the simulated Bayesian inference posterior distribution (Korner-Nievergelt et al. 2015) of the LMEM with garden ID as random factor and garden land-use types as fixed effects. Lines indicate 95 % credible intervals. Estimated LMEM coefficients of fixed effects can be found in Table S 4.10.

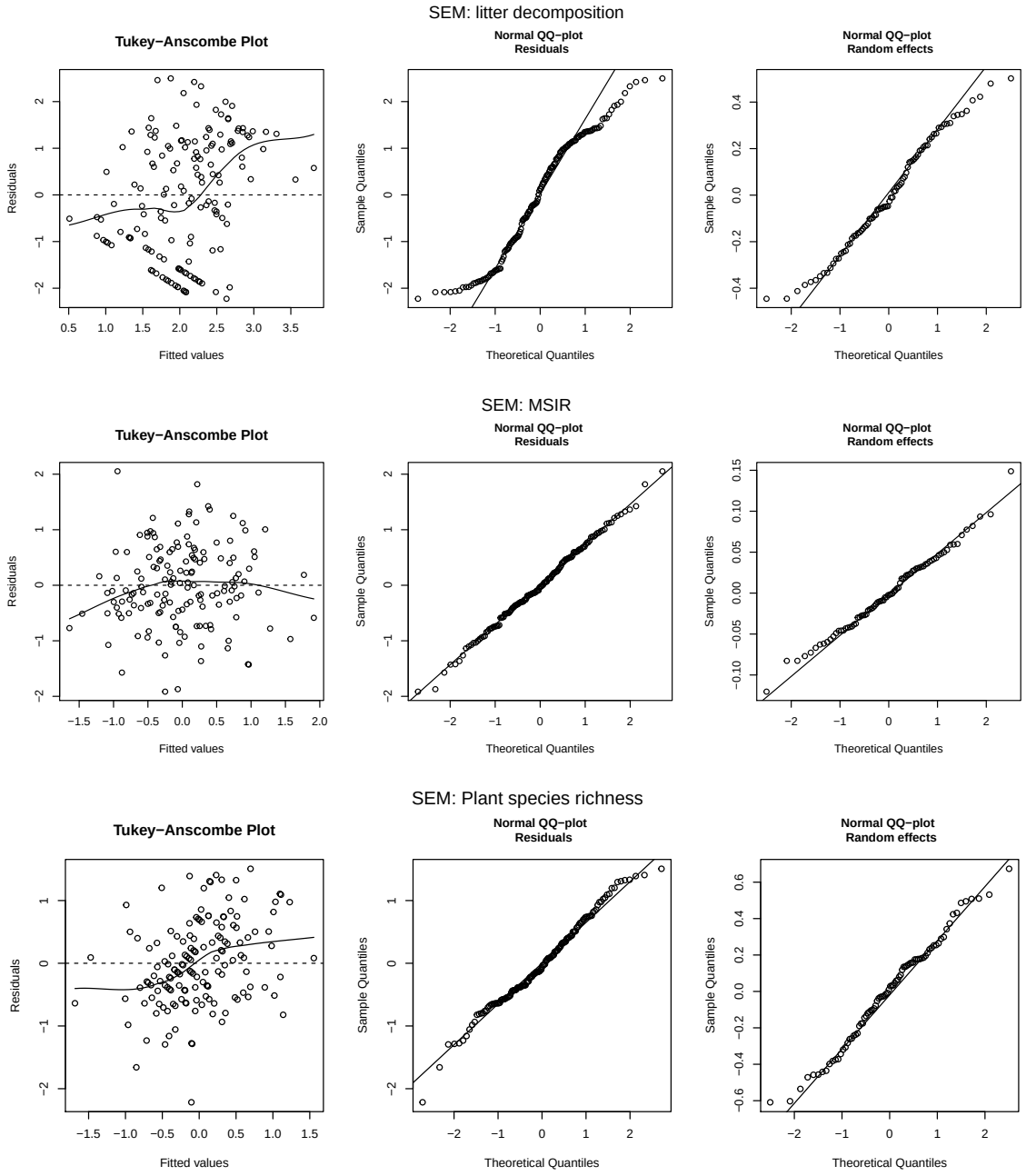

Figure S 4.12 – Residual plots for assessing model assumptions of the LMEM used in the SEM framework. Response variables for the LMEM with leaf 4 mm, microbial activity (MSIR) and plant species richness are plotted (see Table 4 for complete model compositions). Residuals have to be independent and identically distributed, hence they should scatter around zero in the Tukey-Anscombe plots (Korner-Nievergelt et al. 2015). A few measurements do not fit well to the model as recognisable in the QQ-plots of the residuals, however the majority of the observations seem to fulfil the model assumptions well and since we did not assume a non-linear effect of the assessed variables with the response variables, we accept the slight lack of model assumptions. In addition, we checked the assumptions that the random effects are normally distributed, which was the case in both response values.

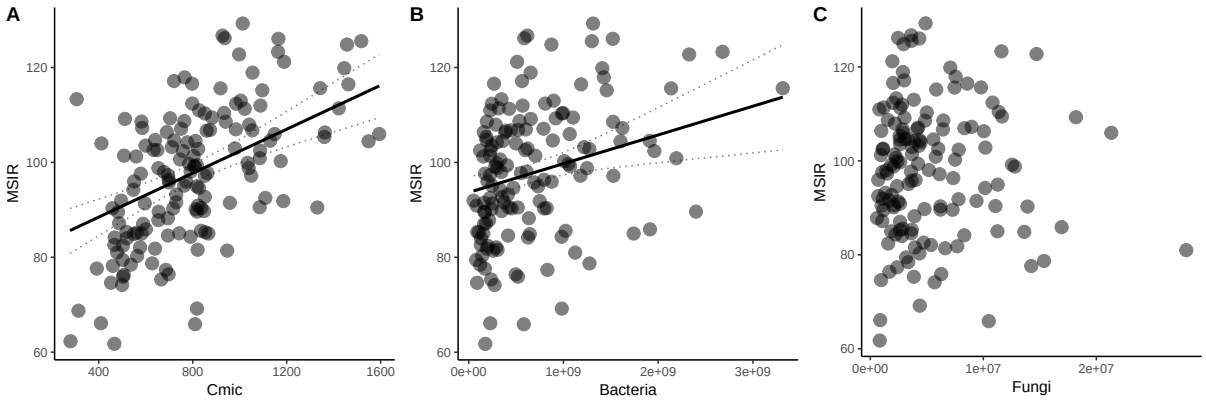

Figure S 4.13 – LMEM effect plots of microbial activity (MSIR) as response variable and microbial biomass (C_{mic}; A), bacterial (B) and fungal (C) qPCR gene copy numbers as fixed effects. Sampling and methodology of bacterial and fungal gene copy numbers can be found in (Tresch et al. 2018b). Lines indicate fitted values of the simulated Bayesian inference posterior distribution (Korner-Nievergelt et al. 2015) of the LMEM with garden ID as random factor. Dotted lines indicate 95 % credible intervals. Estimated LMEM coefficients of fixed effects can be found in Table S 4.10.

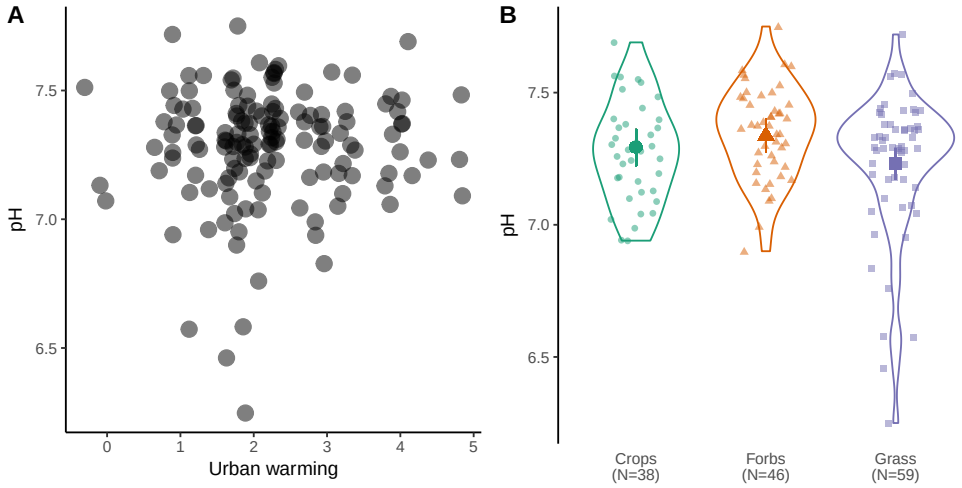

Figure S 4.14 – LMEM effect plots of soil pH as response variable and urban warming A) and urban garden land-use types B) as fixed effects. Solid line and bold points represent fitted or mean values of the simulated Bayesian inference posterior distribution (Korner-Nievergelt et al. 2015) of the LMEM with garden ID as random factor. Dotted lines and error bars indicate 95 % credible intervals. Estimated LMEM coefficients of fixed effects can be found in Table S 4.10.

4.6.2 Supplementary Tables

Table S 4.1 – Trait measurements of *Zea mays* leaf and stem litter used for the decomposition experiment. For each litter type ten randomised samples were measured. Differences in traits were investigated with a non-parametrical Wilcoxon rank sum test.

	Leaf	Stem	P	
N $[\%]$	2.4 ± 0.04	0.62 ± 0.02	<0.001	***
C $[\%]$	43 ± 0.06	44 ± 0.07	0.01	**
C to N ratio	18 ± 0.3	71 ± 3	<0.001	***
pH	6 ± 0.009	5.7 ± 0.01	<0.001	***
Labile A $[A.U.cm^{-1}]$	10 ± 0.6	11 ± 0.6	0.44	
Labile B $[A.U.cm^{-1}]$	1.6 ± 0.04	1.7 ± 0.04	0.31	
Labile C $[A.U.cm^{-1}]$	0.09 ± 0.005	0.14 ± 0.007	<0.001	***
Stable D $[A.U.cm^{-1}]$	0.31 ± 0.03	0.5 ± 0.05	0.01	*
Stable E $[A.U.cm^{-1}]$	0.6 ± 0.04	0.81 ± 0.05	0.01	*
Stable F $[A.U.cm^{-1}]$	0.47 ± 0.01	0.04 ± 0.008	<0.001	***
Stable G $[A.U.cm^{-1}]$	0.059 ± 0.003	0.16 ± 0.008	<0.001	***
Stable H $[A.U.cm^{-1}]$	0.066 ± 0.002	0.12 ± 0.004	<0.001	***
leaf tensile strength $[MegaNm^{-2}]$	1.2 ± 0.06	4.4 ± 0.3	<0.001	***

Table S 4.2 – Soil mesofauna extraction temperature values and time for the high temperature and moisture gradient MacFadyen extractor. Best extraction time and temperature values has been reviewed from the literature (Macfadyen 1953; 1961, Bieri et al. 1978).

Time $[d]$	1	2	3	4	5	6	7
Temperature $[°C]$	20	25	30	35	40	50	60

Table S 4.3 – 19 substrates used for the assessment of the Community level physiological profile (CLPP) based on the MicroResp™ technique (Campbell et al. 2003). We dissolved 18 substrates in H_2O_{demin} and added 25 µl aliquots to deliver 30 mg of C-substrate per g of soil water for each well. Each substrate was measured in five technical replicates. The absorbance of the detection plate is measured at 570 nm after 5 hours of incubation at 20°C in the dark. The detection plate contains a pH sensitive dye (Cresol Red) which is dissolved in a solution with 150 mM potassium chloride (KCl) and 2.5 mM sodium bicarbonate ($NaHCO_3$) in a matrix of 1% agarose gel. For the calibration equations 44 samples from five different soils together with four different quantities (10 g, 20 g, 30 g and 40 g) were amended with 0, 0.5, 2, 3, 5 and 10 mg of glucose or α-keto-glutaric acid per g soil. The substrates were dissolved in water so that 62.5 µl per g soil was added to each sample. Samples without substrates received the same amount of water. The calibration was obtained in 100ml Schott bottles containing 4 wells of breakable microstrips filled with the detection gel. These microstrips were measured immediately before and after the incubation on a plate reader (MRX II TC, Dynex, USA) at 570 nm. The bottles were sealed and CO_2 evolution was measured on a gas chromatograph (7890A, Agilent Technologies, USA). The difference in absorbance between the first and the second measurement is then plotted against the log of CO_2 evolution measured by the gas chromatograph. The linear fit between measured log(CO_2) concentrations $[\mu g CO_2 - C g^{-1} h^{-1}]$ was $y = -4.67 + 2.90$ with an $R^2$of 0.87.

Compound category	Substrate
Amino acid	gamma-aminobutyric acid
	alanine
	aspartic acid
	glutamine
	leucine
	cysteine
Amino sugar	glucosamine
Sugar	arabinose
	galactose
	glucose
	fructose
Carboxylic acid	ascorbic acid
	citric acid
	malic acid
	alpha-keto-glutaric acid
Phenolic acid	protocatechuic acid
	vanillic acid
Hemicellulose	xylan
Water	H_2O

Table S 4.4 – Descriptive statistics of biodiversity components per soil fauna taxa. Species and trait list used for the calculation of the biodiversity components are shown in Table S 4.5. Presented values are mean values with standard errors. S = species richness, N = abundance. All gardens n=168, crops n=46, forbs n=52, grass n=70, allotment n=82, home n=86, urban warming class 1 n=34, urban warming class 2 n=114, urban warming class 3 n=20.

	S	N	$S_{Earthworms}$	$S_{Isopods}$	$S_{Gastropods}$	$S_{Collembola}$	$N_{Earthworms}$	$N_{Isopods}$	$N_{Gastropods}$	$N_{Collembola}$
All gardens	20.3±0.4	162±10	2.9±0.1	4.1±0.1	6.4±0.2	7±0.2	7.2±0.6	70.3±8.4	2.6±0.2	79.7±6.1
Land-use types										
Crops	18.6±0.7	115±14	3.3±0.3	4.2±0.2	5.0±0.3	6.1±0.4	11.2±1.6	54.1±13.3	1.5±0.2	45.3±4.5
Forbs	21.9±0.8	162±22	3.0±0.2	4.1±0.2	8.2±0.4	6.6±0.4	5.8±0.8	103.5±20.7	3.6±0.3	47.2±7.4
Grass	20.2±0.6	192±14	2.5±0.2	3.9±0.2	5.9±0.3	7.9±0.3	5.5±0.8	56.3±9.1	2.5±0.2	126.5±11
Garden types										
Allotment	20.7±0.6	147±13	3.1±0.2	4.4±0.2	6.2±0.3	7.0±0.3	8.4±0.9	55.9±11.2	2.4±0.2	77.8±8.3
Home	19.9±0.5	177±14	2.7±0.2	3.7±0.2	6.6±0.3	7.0±0.3	6.0±0.8	84.1±12.3	2.7±0.2	81.6±8.9
Urban warming classes										
Class 1	21.3±1.0	120±13	3.3±0.3	4.4±0.3	6.4±0.5	7.2±0.5	8.6±1.4	35.8±9.1	2.9±0.4	71.4±8.8
Class 2	20.3±0.5	154±10	2.8±0.2	4.2±0.1	6.1±0.3	7.1±0.3	7.1±0.8	57.2±7.0	2.3±0.2	85.3±8.1
Class 3	18.8±1.2	274±46	2.5±0.3	2.3±0.3	7.8±0.9	6.1±0.8	5.3±1.1	203.8±47.2	3.3±0.5	62±15.5

Table S 4.5 – Species names and traits table. Species names were matched with Hinchliff et al. (2015), while stars indicate that only the genus was found on the open tree of life data base (Hinchliff et al. 2015). Number of collembola species = 39, number of earthworm species = 18, number of gastropod species = 47 and number of ispod species = 16.

Taxa	*Species*	Vertical distribution	Body size $[mm]$	Reference
Collembola	*Bourletiella hortensis*	surface	0.18	Ellers et al. (2018)
	Ceratophysella bengtssoni	mixed	0.13	Ellers et al. (2018)
	Ceratophysella denticulata	mixed	0.18	Ellers et al. (2018)
	Choreutinula inermis	mixed	0.15	Ellers et al. (2018)
	Cryptopygus thermophilus	mixed	0.10	Ellers et al. (2018)
	Desoria violacea *	mixed	0.24	Ellers et al. (2018)
	Dicyrtomina ornata	surface	0.30	Ellers et al. (2018)
	Entomobrya marginata	surface	0.20	Ellers et al. (2018)
	Entomobrya multifasciata	surface	0.20	Ellers et al. (2018)
	Folsomia candida	soil	0.30	Ellers et al. (2018)
	Folsomia quadrioculata	mixed	0.25	Ellers et al. (2018)
	Folsomia similis	soil	0.09	Hopkin (2007)
	Folsomia spinosa	soil	0.11	Hopkin (2007)
	Folsomides parvulus	soil	0.09	Ellers et al. (2018)
	Heteromurus nitidus	soil	0.30	Ellers et al. (2018)
	Hypogastrura purpurescens	mixed	0.23	Ellers et al. (2018)
	Isotoma viridis	mixed	0.60	Ellers et al. (2018)
	Isotomiella minor	soil	0.12	Ellers et al. (2018)
	Isotomurus balteatus	mixed	0.13	Hopkin (2007)
	Isotomurus graminis	mixed	0.30	Ellers et al. (2018)
	Isotomurus palustris	mixed	0.34	Ellers et al. (2018)
	Kalaphorura burmeisteri *	surface	0.31	Hopkin (2007)
	Lepidocyrtus cyaneus	surface	0.15	Ellers et al. (2018)
	Lepidocyrtus lignorum	surface	0.20	Ellers et al. (2018)
	Lepidocyrtus violaceus	surface	0.16	Ellers et al. (2018)
	Megalothorax minimus	soil	0.04	Ellers et al. (2018)
	Mesaphorura macrochaeta	soil	0.07	Hopkin (2007)
	Metaphorura affinis	soil	0.13	Ellers et al. (2018)
	Neanura muscorum	mixed	0.35	Ellers et al. (2018)
	Onychiurus granulosus *	soil	0.15	Ellers et al. (2018)
	Parisotoma notabilis	mixed	0.10	Ellers et al. (2018)
	Pogonognathellus flavescens	mixed	0.65	Ellers et al. (2018)
	Protaphorura pulvinata	soil	0.17	Ellers et al. (2018)
	Pseudosinella alba	soil	0.11	Ellers et al. (2018)
	Pseudosinella pseudopetterseni	soil	0.18	Ellers et al. (2018)
	Schoettella ununguiculata	mixed	0.17	Ellers et al. (2018)
	Sminthurinus aureus	mixed	0.10	Ellers et al. (2018)
	Sphaeridia pumilis	mixed	0.05	Ellers et al. (2018)
	Stenaphorura denisi	soil	0.13	Ellers et al. (2018)
Earthworms	*Allolobophora chlorotica*	soil	80.00	Blakemore (2008)
	Aporrectodea icterica	soil	90.00	Blakemore (2008)
	Aporrectodea caliginosa	soil	80.00	Blakemore (2008)
	Aporrectodea nocturna	mixed	150.00	Blakemore (2008)
	Aporrectodea tuberculata	soil	120.00	Blakemore (2008)
	Aporrectodea giardi	mixed	250.00	Blakemore (2008)
	Aporrectodea longa	mixed	170.00	Blakemore (2008)
	Aporrectodea ripicola *	mixed	170.00	Blakemore (2008)
	Aporrectodea rosea	soil	85.00	Blakemore (2008)
	Dendrobaena octaedra	surface	60.00	Blakemore (2008)
	Dendrodrilus rubidus	surface	60.00	Blakemore (2008)
	Dendrodrilus subrubicundus	surface	90.00	Blakemore (2008)
	Lumbricus castaneus	surface	60.00	Blakemore (2008)
	Lumbricus festivus	surface	95.00	Blakemore (2008)
	Lumbricus rubellus	surface	130.00	Blakemore (2008)
	Lumbricus terrestris	mixed	250.00	Blakemore (2008)
	Octolasion cyaneum	soil	140.00	Blakemore (2008)
	Octolasion lacteum	soil	160.00	Blakemore (2008)

Taxa	*Species*	vertical distribution	body size $[mm]$	Reference
Gastropoda	*Acicula lineata*	surface	3.75	Falkner et al. (2001)
	Aegopinella minor	mixed	10.00	Falkner et al. (2001)
	Aegopinella nitens	mixed	10.00	Falkner et al. (2001)
	Aegopinella pura	surface	3.75	Falkner et al. (2001)
	Arion	surface	80.00	Falkner et al. (2001)
	Boettgerilla pallens	soil	60.00	Falkner et al. (2001)
	Carychium tridentatum	mixed	1.25	Falkner et al. (2001)
	Cecilioides acicula	soil	53.10	Falkner et al. (2001)
	Cepaea hortensis	surface	22.50	Falkner et al. (2001)
	Cepaea nemoralis	surface	22.50	Falkner et al. (2001)
	Cepaea	surface	22.50	Falkner et al. (2001)
	Cochlicopa lubrica	mixed	10.00	Falkner et al. (2001)
	Columella edentula	surface	3.75	Falkner et al. (2001)
	Deroceras	surface	30.00	Falkner et al. (2001)
	Discus rotundatus	mixed	10.00	Falkner et al. (2001)
	Fruticicola fruticum	surface	19.40	Falkner et al. (2001)
	Galba truncatula	surface	2.75	Falkner et al. (2001)
	Helix pomatia	surface	22.50	Falkner et al. (2001)
	Hygromia cinctella	surface	10.00	Falkner et al. (2001)
	Laciniaria plicata	mixed	10.00	Falkner et al. (2001)
	Limax maximus	surface	150.00	Falkner et al. (2001)
	Macrogastra attenuata	mixed	17.50	Falkner et al. (2001)
	Monachoides incarnatus	mixed	13.10	Falkner et al. (2001)
	Nesovitrea hammonis	mixed	3.75	Falkner et al. (2001)
	Oxychilus cellarius	soil	10.00	Falkner et al. (2001)
	Oxychilus draparnaudi	mixed	13.10	Falkner et al. (2001)
	Oxychilus *	mixed	11.55	Falkner et al. (2001)
	Paralaoma servilis	mixed	1.25	Falkner et al. (2001)
	Punctum pygmaeum	mixed	1.25	Falkner et al. (2001)
	Pupilla muscorum	surface	3.75	Falkner et al. (2001)
	Succinea putris	surface	13.13	Falkner et al. (2001)
	Succinea oblonga	mixed	10.00	Falkner et al. (2001)
	Tandonia budapestensis	surface	35.00	Falkner et al. (2001)
	Trochulus clandestinus	mixed	10.00	Falkner et al. (2001)
	Trochulus sericeus	surface	10.00	Falkner et al. (2001)
	Trochulus	mixed	10.00	Falkner et al. (2001)
	Vallonia costata	mixed	2.50	Falkner et al. (2001)
	Vallonia excentrica	surface	1.88	Falkner et al. (2001)
	Vallonia pulchella	mixed	2.50	Falkner et al. (2001)
	Vallonia	mixed	2.29	Falkner et al. (2001)
	Vertigo antivertigo	surface	1.25	Falkner et al. (2001)
	Vertigo pusilla	mixed	1.25	Falkner et al. (2001)
	Vertigo pygmaea	mixed	1.25	Falkner et al. (2001)
	Vertigo	mixed	1.25	Falkner et al. (2001)
	Vitrea contracta	soil	2.50	Falkner et al. (2001)
	Vitrea crystallina	mixed	3.75	Falkner et al. (2001)
	Vitrinobrachium breve	soil	3.75	Falkner et al. (2001)
Isopoda	*Androniscus roseus*	soil	0.35	Vandel (1960)
	Armadillidium nasatum	surface	2.05	Vandel (1960)
	Armadillidium versicolor	surface	1.75	Vandel (1960)
	Armadillidium vulgare	surface	1.73	Vandel (1960)
	Cylisticus convexus	soil	1.35	Vandel (1960)
	Haplophthalmus danicus	soil	0.40	Vandel (1960)
	Haplophthalmus mengei	soil	0.30	Vandel (1960)
	Hyloniscus riparius	soil	0.80	Vandel (1960)
	Ligidium hypnorum	surface	0.87	Vandel (1960)
	Oniscus asellus	surface	1.63	Vandel (1960)
	Philoscia muscorum	surface	1.06	Vandel (1960)
	Platyarthrus hoffmannseggii	soil	0.30	Vandel (1960)
	Porcellio scaber	surface	1.51	Vandel (1960)
	Porcellionides pruinosus	mixed	1.05	Vandel (1960)
	Trachelipus rathkii	surface	1.40	Vandel (1960)
	Trichoniscus pygmaeus	soil	0.22	Vandel (1960)

Table S 4.6 – Management questions asked of all 85 participating urban gardeners of this study. Management intensity index was calculated as a scaled sum (divided by the number of questions) of all 26 garden management questions on a five level Likert scale. For the land-use type grass we considered nine questions: MowGrass, FstCutGrass, FertGrass, WaterGrass, CareGrass, PestGrass, FlowerIslands, Weeds, Leaves. For forbs ten questions: FertForbs, WaterForbs, PestForbs, DiggingForbs, ForkForbs, CutTrees, PestTrees, Leaves, DrySticks, Weeds and for crops eleven questions: FertCrops, WaterCrops, PestCrops, CropRotate, MixCult, Mulch, GreenFert, DiggingCrops, ForkCrops, DrySticks, Weeds. Higher factor levels indicate higher management intensity. Questions were originally asked in German.

PestGrass
How often do you use pesticides, fungicides or herbicides to protect your lawn?
Never (1)
Less than once per year (2)
1 to 3 times per year (3)
4 to 10 times per year (4)
More than 10 times per year (5)

PestForbs
How often do you use pesticides, fungicides or herbicides (without slug pellets) to protect your flowers?
Never (1)
Less than once per year (2)
1 to 3 times per year (3)
4 to 10 times per year (4)
More than 10 times per year (5)

PestCrops
How often do you use pesticides, fungicides or herbicides (without slug pellets) to protect your vegetables?
Never (1)
Less than once per year (2)
1 to 3 times per year (3)
4 to 10 times per year (4)
More than 10 times per year (5)

FertGrass
How often do you use fertilisers for your lawn?
Never (1)
4 to 5 times per year (2)
2 to 3 times per year (3)
once a year (4)
More than once a year (5)

FertCrops
How often do you use fertilisers for your vegetables?
Never (1)
2 to 3 times per year (2)
Once a year (3)
2 to 3 times per year (4)
More than three per year (5)

FertForbs
How often do you use fertilisers for your flowers?
Never (1)
2 to 3 times per year (2)
Once a year (3)
2 to 3 times per year (4)
More than three per year (5)

Weeds
How often do you remove most of the weeds in your garden?
Never (1)
Rarely (2)
Sometimes (3)
Often (4)
Very often (5)

PestTrees
How often do you use insecticides, fungicides or herbicides to protect our trees and shrubs?
Never (1)
Less than once a year (2)
1 to 3 times per year (3)
4 to 10 times per year (4)
More than 10 times per year (5)

Leaves
How often do you remove most of the leaves in your garden?
Never (1)
Spring (2)
Autumn (3)
Every 2 to 3 weeks (4)
Weekly in autumn (5)

MowGrass
How often do you mow your lawn per year?
1 to 2 (1)
3 to 4 (2)
5 to 8 (3)
9 to 20 (4)
over 20 (5)

MixCult
Do you follow the principle of mixed cultivation (planting different varieties of vegetables and/or flowers in the same cultivation plot)?
Never (5)
Rarely (4)
Sometimes (3)
Mostly (2)
Always (1)

FlowerIslands
Do you leave islands of flowers when you mow your lawn?
Never (5)
Rarely (4)
Sometimes (3)
Mostly (2)
Always (1)

WaterGrass
How often do you water your lawn?

Never (1)
When dry (2)
once a week (3)
twice a week (4)
More than twice a week (5)

WaterCrops
How often do you water your vegetable beds?
Never (1)
When dry (2)
once a week (3)
twice a week (4)
More than twice a week (5)

WaterForbs
How often do you water your flower beds?
Never (1)
When dry (2)
once a week (3)
twice a week (4)
More than twice a week (5)

CareGrass
How often do you scarify your lawn (including reseeding)
Never (1)
Every 6 to 10 years (2)
Every 4 to 5 years (3)
Every 2 to 3 years (4)
Annually (5)

DiggingForbs
How often do you till your soil in the flower beds?
Never (1)
Every 3 years or less (2)
Every two years (3)
Once per year (4)
More than once per year (5)

DiggingCrops
How often do you till your soil in the vegetable beds?
Never (1)
Every 3 years or less (2)
Every two years (3)
Once per year (4)
More than once per year (5)

DrySticks
Do you leave withered flowers and sticks during the winter in your garden?
Never (5)
Rarely (4)
Sometimes (3)
Mostly (2)
Always (1)

FstCutGrass
When is the first time point of cutting your lawn?

April (5)
May (4)
Start June (3)
End June (2)
After June (1)

Mulch
Do you use organic material (mulch) to cover your vegetable beds?

Never (5)
Rarely (4)
Sometimes (3)
Mostly (2)
Always (1)

CropRotate
Do you consider changing flower beds (crop rotation) for the vegetables grown annually?
Never (5)
Rarely (4)
Sometimes (3)
Mostly (2)
Always (1)

GreenFert
Do you grow plants for green manure?

Never (5)
Rarely (4)
Sometimes (3)
Mostly (2)
Always (1)

WeedingHerbicide
Do you use commercial herbicides?

No (0)
Yes (1)

ForkForbs
How often do you loosening your soil with a fork without turning it around (or milling)?
More than once per year (5)
Once per year (4)
Every 2 years or less (3)
Every 3 years or less (2)
Never (1)

ForkCrops
How often do you loosening your soil with a fork without turning it around (or milling)?
More than once per year (5)
Once per year (4)
Every 2 years or less (3)
Every 3 years or less (2)
Never (1)

CutTrees
How often do you cut most of your forbs and trees?

More than once per year (5)
Once a year (4)
Every 2 years (3)
Every 3 to 5 years(2)
Less than every 5 years (1)

Table S 4.7 – Model selection based on goodness of fit statistics for LMEM, the widely applicable information criterion (WAIC), a Bayesian version of the AIC (Watanabe 2010) and explained variance of the fixed effects $R^2_{Marginal}$ and including the random effect and the fixed effects $R^2_{Conditional}$. Model 1 included the selected biodiversity indices species richness (S), species evenness ($E_{Shannon}$), phylogenetic species variability (PSV), trait even distribution (TED) and the garden ID as random effect : $lmer(log(100 - response + 1) \sim S + E_{Shannon} + PSV + TED + (1|Garden\_ID), REML = F)$. Model 2 included garden land-use and garden type in addition to predictor variables of the previous model 1. For model 3 we included the multiple substrate-induced respiration rate (MSIR) as a measure for microbial activity. For model 4 we added urban warming. All fixed effects have been standardised (mean=0, standard deviation=1). R^2 based on fixed and random effects were calculated according to Nakagawa and Schielzeth (2013).

Litter type	models	WAIC	$R^2_{Marginal}$	$R^2_{Conditional}$
Leaf 4mm	model 1	561.6 ± 11.4	0.07	0.30
	model 2	529.8 ± 12.2	0.20	0.53
	model 3	528.0 ± 12.2	0.22	0.54
	model 4	523.0 ± 13.9	0.26	0.55
Stem 4mm	model 1	-137.0 ± 17.7	0.11	0.17
	model 2	-200.2 ± 15.9	0.37	0.61
	model 3	-203.4 ± 16.5	0.39	0.62
	model 4	-203.4 ± 16.5	0.43	0.63
Leaf 1mm	model 1	365.1 ± 27.6	0.11	0.11
	model 2	361.7 ± 31.3	0.16	0.19
	model 3	358.1 ± 29.6	0.19	0.24
	model 4	359.4 ± 29.4	0.19	0.24
Stem 1mm	model 1	-86.1 ± 26.00	0.13	0.13
	model 2	-129.8 ± 23.6	0.35	0.54
	model 3	-129.5 ± 24.0	0.35	0.54
	model 4	-139.2 ± 22.8	0.41	0.56

Table S 4.8 – Litter decomposition models for leaf and stem litter types and 1 and 4 mm mesh sizes. LMEM fixed effects were calculated using a simulated Bayesian inference posterior distribution. Bold numbers indicate significant fixed effects, with credible intervals not crossing zero. Number of observations for 4 mm litter bags n=154 and for 1mm litter bags n=122, due to missing litter bags from the garden sites. Garden ID was set as random factor. LMEM model: $lmer(log(100 - decomposition + 1) \sim MSIR + S + E_{Shannon} + PSV + TED + urban\_warming + land\_use\_types + garden\_types + (1|Garden\_ID))$. $E_{Shannon}$: Shannon evenness, MSIR: Multiple substrate-induced respiration of microorganisms, PSV: Phylogenetic species variability, S: Soil fauna species richness, TED: Trait even distribution.

	Leaf 4mm [log(g)]		**Stem 4mm [log(g)]**	
Fixed effects	50%	(97.5%; 2.5%)	50%	(97.5%; 2.5%)
MSIR	**0.24**	(0.48; 0.01)	**0.03**	(0.05; 0.01)
S	**0.29**	(0.53; 0.04)	0.01	(0.04;-0.01)
$E_{Shannon}$	0.13	(0.40;-0.14)	0.01	(0.02;-0.03)
PSV	-0.03	(0.20;-0.25)	0.01	(0.02;-0.02)
TED	-0.05	(0.17;-0.26)	-0.01	(0.01;-0.03)
Urban warming	**0.45**	(0.76; 0.14)	**0.04**	(0.07; 0.01)
Land-use types: forbs	0.35	(0.92;-0.22)	**0.06**	(0.11; 0.01)
Land-use types: grass	**1.10**	(1.60; 0.61)	**0.16**	(0.21; 0.12)
Garden types: home	0.00	(0.62;-0.61)	0.04	(0.09;-0.02)

	Leaf 1mm [log(g)]		**Stem 1mm [log(g)]**	
Fixed effects	50%	(97.5%; 2.5%)	50%	(97.5%; 2.5%)
MSIR	0.11	(0.23;-0.01)	**0.03**	(0.06; 0.01)
S	-0.10	(0.03;-0.22)	**0.03**	(0.06; 0.01)
$E_{Shannon}$	-0.03	(0.09;-0.15)	**0.03**	(0.06; 0.01)
PSV	-0.11	(0.01;-0.24)	-0.02	(0.01;-0.06)
TED	0.09	(0.20;-0.02)	0.01	(0.03;-0.02)
Urban warming	0.10	(0.22;-0.03)	**0.06**	(0.10; 0.03)
Land-use types: forbs	0.03	(0.33;-0.27)	**0.08**	(0.15; 0.01)
Land-use types: grass	**0.34**	(0.64; 0.03)	**0.17**	(0.25; 0.09)
Garden types: home	0.09	(0.34;-0.17)	0.01	(0.09;-0.06)

Table S 4.9 – Litter decomposition grouped by garden land-use types, management and urban warming classes (A-D). Tea bag index (TBI) decomposition is described in Tresch et al. (2018a). Question disturbance is related to the combined management answers (yes/no) of major soil disturbances ("DiggingVeg", "DiggingFlower", "CareLawn"); pesticides is related to the use of pesticides ("PestLawn","PestVeg","PestFlower","PestTrees","WeedingHerbicide"); compost is related to the use of compost ("FertLawnCompost","FertVegCompost","FertFlowerCompost") and water to the application of additional water ("WaterLawn", "WaterVeg", "WaterFlower"). Individual management questions can be found in Table S 4.6. Urbanisation intensity was investigated as urban warming classes. Decomposition [%] is given as a mean decomposition rate including standard error values. Differences were investigated with a non-parametrical Wilcoxon rank sum test and if more than two groups with a LMEM with garden ID as random effect and analysed with a Bayesian approach including means and 95 % credible intervals of the Bayesian inference posterior distributions following Korner-Nievergelt et al. (2015). Significant values are bold printed.

A

			Fixed effects [log(g)]		
Litter type	**N**	**Decomposition [%]**	**50%**	**97.5%**	**2.5%**
Leaf 4mm	69	79.59±2.21			
Stem 4mm	62	37.89±0.84	**-2.04**	-1.82	-2.28
Leaf 1mm	90	61.15±1.95			
Stem 1mm	65	40.07±0.96	**-0.63**	-0.51	-0.76
Leaf 1mm	90	61.15±1.95			
Leaf 4mm	69	79.59±2.21	**1.37**	1.61	1.12
Stem 1mm	65	40.07±0.96			
Stem 4mm	62	37.89±0.84	**-0.04**	-0.01	-0.07

B **Decomposition per land-use type**

	Leaf 4mm					Stem 4mm					Leaf 1mm					Stem 1mm					TBI green tea					TBI rooibos tea				
Land-use			Fixed effects [log(g)]					Fixed effects [log(g)]					Fixed effects [log(g)]					Fixed effects [log(g)]					Fixed effects [log(g)]					Fixed effects [log(g)]		
types	**N**	**Decomposition [%]**	**50%**	**97.5%**	**2.5%**	**N**	**Decomposition [%]**	**50%**	**97.5%**	**2.5%**	**N**	**Decomposition [%]**	**50%**	**97.5%**	**2.5%**	**N**	**Decomposition [%]**	**50%**	**97.5%**	**2.5%**	**N**	**Decomposition [%]**	**50%**	**97.5%**	**2.5%**	**N**	**Decomposition [%]**	**50%**	**97.5%**	**2.5%**
crops	31	61.79±5.57				31	30.45±1.88				29	45.08±5.33				28	29.98±3.06				39	57.86±0.67				39	29.28±0.53			
forbs	33	80.66±3.52	**0.7**	1.26	0.15	34	37.89±1.20	**0.11**	0.16	0.06	29	56.23±3.54	0.17	0.46	-0.12	27	39.09±1.55	**0.13**	0.2	0.05	45	58.64±0.60	0.01	0.06	-0.04	45	29.5±0.54	0.034	0.024	-0.017
grass	30	90.54±1.76	**1.34**	1.83	0.84	36	42.82±0.88	**0.19**	0.24	0.15	43	69.63±2.19	**0.56**	0.82	0.29	31	46.00±0.96	**0.25**	0.31	0.18	59	60.31±0.66	**0.06**	0.11	0.02	59	29.48±0.45	0.033	0.023	-0.017

C **Decomposition per urban warming class**

	Leaf 4mm					Stem 4mm					Leaf 1mm					Stem 1mm					TBI green tea					TBI rooibos tea				
Urban warming			Fixed effects [log(g)]					Fixed effects [log(g)]					Fixed effects [log(g)]					Fixed effects [log(g)]					Fixed effects [log(g)]					Fixed effects [log(g)]		
classes	**N**	**Decomposition [%]**	**50%**	**97.5%**	**2.5%**	**N**	**Decomposition [%]**	**50%**	**97.5%**	**2.5%**	**N**	**Decomposition [%]**	**50%**	**97.5%**	**2.5%**	**N**	**Decomposition [%]**	**50%**	**97.5%**	**2.5%**	**N**	**Decomposition [%]**	**50%**	**97.5%**	**2.5%**	**N**	**Decomposition [%]**	**50%**	**97.5%**	**2.5%**
Class 1	27	70.90±5.26				25	33.68±2.09				24	53.6±6.08				23	34.29±2.82				29	57.33±0.70				29	28.59±0.65			
Class 2	50	79.85±2.77	0.63	1.28	-0.01	55	38.05±0.99	**0.07**	0.13	0.01	61	59.4±2.38	0.02	0.31	-0.26	47	40.01±1.28	0.09	0.17	0.01	98	59.36±0.47	**0.05**	0.1	0.01	98	29.51±0.35	0.013	0.033	-0.07
Class 3	12	91.92±2.84	**1.32**	2.22	0.41	15	43.71±1.52	**0.15**	0.24	0.07	10	73.4±5.06	**0.49**	0.97	0.01	9	52.15±3.25	**0.32**	0.45	0.18	16	60.83±1.30	**0.09**	0.16	0.01	16	30.50±0.81	0.027	0.057	-0.03

D **Decomposition per garden management**

Management question	group	Leaf 4mm N	Leaf 4mm Decomposition [%]	Leaf 4mm P		Stem 4mm N	Stem 4mm Decomposition [%]	Stem 4mm P		Leaf 1mm N	Leaf 1mm Decomposition [%]	Leaf 1mm P	Stem 1mm N	Stem 1mm Decomposition [%]	Stem 1mm P	TBI green tea N	TBI green tea Decomposition [%]	TBI green tea P	TBI rooibos tea N	TBI rooibos tea Decomposition [%]	TBI rooibos tea P
disturbance	no	33	85.39±2.73	**0.04**	*	40	40.83±1.08	**0.01**	**	42	63.7±3.12	0.08	30	42.41±1.26	0.06	61	59.66±0.59	0.16	61	29.77±0.44	0.37
	yes	48	75.56±3.18			53	35.85±1.16			53	56.13±3.00		46	37.87±1.80		82	58.71±0.51		82	29.18±0.38	
pesticides	no	44	77.92±3.12	0.54		52	37.16±1.15	0.34		58	59.34±3.00	0.60	45	39.26±1.57	0.48	81	59.05±0.54	0.74	81	29.77±0.37	0.28
	yes	40	81.74±3.09			38	38.83±1.21			40	59.27±3.09		34	40.62±1.77		62	59.20±0.56		62	28.99±0.45	
compost	no	33	76.52±3.86	0.24		35	37.84±1.44	0.99		33	59.50±3.75	0.80	27	40.67±1.77	0.72	49	59.37±0.66	0.69	49	28.99±0.53	0.18
	yes	52	81.21±2.70			53	37.92±1.03			59	59.22±2.72		50	39.36±1.53		94	58.98±0.48		94	29.67±0.34	
water	no	8	88.69±4.14	0.97		8	38.62±2.22	0.94		6	60.67±6.75	0.60	5	43.00±1.87	0.78	8	59.65±1.39	0.56	8	29.52±1.10	0.99
	yes	66	79.08±2.32			62	37.85±0.87			75	59.24±2.29		58	39.61±1.24		135	59.08±0.40		135	29.43±0.30	

E **Decomposition per garden land-use type and management**

Management question	Land-use types	Group	Leaf 4mm N	Leaf 4mm Decomposition [%]	Leaf 4mm P		Stem 4mm N	Stem 4mm Decomposition [%]	Stem 4mm P		Leaf 1mm N	Leaf 1mm Decomposition [%]	Leaf 1mm P		Stem 1mm N	Stem 1mm Decomposition [%]	Stem 1mm P		TBI green tea N	TBI green tea Decomposition [%]	TBI green tea P	TBI rooibos tea N	TBI rooibos tea Decomposition [%]	TBI rooibos tea P
disturbance	crops	no	14	68.23 ± 8.44	0.45		13	35.33 ± 2.71	0.07		13	51.54 ± 8.95	0.26		12	35.58 ± 3.68	0.09		14	57.94 ± 0.88	0.67	14	30.32 ± 0.93	0.24
		yes	20	58.08 ± 7.33			23	27.63 ± 2.39			17	40.66 ± 6.57			17	26.16 ± 4.36			25	57.81 ± 0.94		25	28.70 ± 0.62	
	forbs	no	15	88.76 ± 3.20	**0.02**	*	16	41.57 ± 1.25	**0.01**	**	16	63.94 ± 4.55	**0.05**	*	14	43.19 ± 1.45	**0.01**	**	23	59.57 ± 0.79	0.10	23	29.49 ± 0.76	0.88
		yes	21	73.20 ± 5.75			21	34.50 ± 1.77			15	48.97 ± 4.85			15	35.24 ± 2.34			22	57.66 ± 0.87		22	29.52 ± 0.79	
	grass	no	13	92.88 ± 1.87	0.58		20	43.56 ± 1.59	0.33		17	71.05 ± 2.81	0.81		16	46.05 ± 0.97	0.82		24	60.74 ± 1.16	0.79	24	29.73 ± 0.67	0.62
		yes	21	89.07 ± 2.62			30	42.36 ± 1.04			31	68.73 ± 3.13			24	45.97 ± 1.46			35	60.01 ± 0.80		35	29.32 ± 0.61	
pesticides	crops	no	18	61.48 ± 7.89	0.98		20	29.96 ± 2.46	0.52		19	42.55 ± 7.27	0.65		21	29.07 ± 3.82	0.56		23	57.84 ± 0.98	0.79	23	29.43 ± 0.71	0.94
		yes	15	62.28 ± 7.50			14	31.22 ± 3.00			11	49.91 ± 7.12			10	31.73 ± 5.31			16	57.89 ± 0.89		16	29.07 ± 0.81	
	forbs	no	19	79.37 ± 4.67	0.46		21	36.87 ± 1.51	0.29		22	59.63 ± 4.57	0.06		21	39.89 ± 2.03	0.22		26	57.76 ± 0.67	0.15	26	30.17 ± 0.63	0.12
		yes	17	82.18 ± 5.44			20	39.09 ± 1.94			10	48.40 ± 4.47			9	37.25 ± 2.07			19	59.84 ± 1.05		19	28.58 ± 0.93	
	grass	no	20	88.91 ± 2.54	0.34		27	42.69 ± 1.33	0.80		26	70.87 ± 2.63	0.73		21	45.92 ± 1.38	0.59		32	60.97 ± 0.96	0.35	32	29.69 ± 0.60	0.68
		yes	18	92.52 ± 2.39			23	42.98 ± 1.12			23	68.08 ± 3.71			19	46.10 ± 1.34			27	59.52 ± 0.90		27	29.24 ± 0.68	
compost	crops	no	5	52.00 ± 20.5	0.89		5	26.40 ± 8.32	0.56		4	39.38 ± 16.7	0.65		4	26.88 ± 10.0	0.78		5	56.27 ± 1.59	0.31	5	29.10 ± 1.39	0.79
		yes	29	63.15 ± 5.77			27	31.01 ± 1.86			26	45.89 ± 5.71			25	30.43 ± 3.26			34	58.09 ± 0.74		34	29.31 ± 0.58	
	forbs	no	18	74.91 ± 6.10	0.17		20	37.00 ± 1.99	0.88		12	51.58 ± 6.56	0.50		10	38.54 ± 3.05	0.74		21	59.58 ± 0.97	0.11	21	29.03 ± 0.78	0.46
		yes	19	85.94 ± 3.57			23	38.70 ± 1.43			19	59.25 ± 3.99			19	39.45 ± 1.67			24	57.81 ± 0.71		24	29.92 ± 0.76	
	grass	no	18	83.17 ± 3.85	**0.02**	*	20	41.02 ± 1.49	0.09		20	68.24 ± 3.68	0.43		19	44.62 ± 1.17	0.23		23	59.84 ± 1.02	0.61	23	28.93 ± 0.84	0.20
		yes	18	95.20 ± 1.02			27	43.96 ± 1.07			29	70.52 ± 2.75			22	46.88 ± 1.38			36	60.60 ± 0.88		36	29.84 ± 0.50	
water	crops	no	0				0				0				0				0			0		
		yes	31	61.79 ± 5.57			31	30.45 ± 1.88			29	45.08 ± 5.33			28	29.98 ± 3.06			39	57.86 ± 0.67		39	29.28 ± 0.53	
	forbs	no	5	92.50 ± 5.14	0.38		5	40.40 ± 2.00	0.36		3	66.83 ± 13.5	0.47		3	44.50 ± 3.33	0.21		5	61.31 ± 1.45	0.07	5	30.56 ± 1.56	0.49
		yes	30	79.28 ± 3.84			33	37.59 ± 1.32			26	55.17 ± 3.68			25	38.55 ± 1.65			40	58.30 ± 0.64		40	29.37 ± 0.58	
	grass	no	3	82.33 ± 6.33	0.09		3	35.67 ± 5.09	0.14		3	54.50 ± 2.52	**0.03**	*	3	41.50 ± 2.02	0.14		3	56.88 ± 2.23	0.20	3	27.78 ± 0.89	0.34
		yes	28	90.96 ± 1.82			35	43.19 ± 0.88			41	70.52 ± 2.25			29	46.26 ± 1.00			56	60.49 ± 0.68		56	29.58 ± 0.47	

Table S 4.10 – Estimated LMEM coefficients of different response variables and fixed effect variables (A-C). Garden ID was set as random effect in all models. Given are the mean, the 2.5% and the 97.5% quantiles of the Bayesian posterior distribution. Bold numbers indicate significant fixed effects, with credible intervals not crossing zero (Korner-Nievergelt et al. 2015).

A

Response variables	Fixed effects	
	Forbs vs. Crops	Grass vs. Crops
Plant species richness	-0.28 (-4.0;3.4)	**-3.57 (-6.9;-0.2)**

B

Response variables	Fixed effects		
	C_{mic}	Bacteria	Fungi
MSIR [$\mu gCO_2\text{-}Cg^{-1}h^{-1}$]	**2.3e-02 (1.5e-02;3.1e-02)**	**6.1e-09 (1.9e-09;1.0e-08)**	-4.3e-07 (-9.4e-07;9.7e-08)

C

Response variables	Fixed effects		
	Urban warming	Forbs vs. Crops	Grass vs. Crops
pH	0.002 (-0.04;0.04)	0.044 (-0.05;0.13)	-0.061 (-0.14;0.02)

Table S 4.11 – Alternative SEM including the effect of soil fauna abundance, as a proxy for soil fauna biomass, on litter decomposition (leaf litter 4 mm). SEM model goodness-of-fit outputs decreased in comparison to the original SEM (Table 4), with an increased AICc and a decreased P-value: AICc=356.8, Fisher's C=175.9, P-value=0.25. Marginal R^2 based on fixed effects and conditional R^2 based on fixed and random effects. Significant paths are highlighted in bold. BD: Soil bulk density, C_{mic}: Microbial biomass carbon, $E_{Shannon}$: Shannon evenness, MSIR: Multiple substrate-induced respiration of microorganisms, N_{min}: Nitrogen mineralisation, PSV: Phylogenetic species variability, S: Soil fauna species richness, N: Soil fauna species abundance, Sb: Antimony content, TED: Trait even distribution, TON: Total organic nitrogen, WHC: Water holding capacity.

Response	$R^2_{conditional}$	$R^2_{marginal}$	Predictor	Estimate	P	
Leaf 4 mm	0.56	0.24	**Land-use types: grass**	0.93±0.3	0.002	**
			Urban warming	0.44±0.2	0.01	**
			MSIR	0.31±0.1	0.02	*
			S	0.31±0.1	0.02	*
			Land-use types: forbs	0.21±0.3	0.48	
			TED	-0.07±0.1	0.55	
			N	-0.089±0.2	0.58	
			$E_{Shannon}$	0.084±0.2	0.65	
			PSV	-0.031±0.1	0.80	
			Garden types: home	-0.044±0.3	0.89	
MSIR	0.41	0.39	**C_{mic}**	0.30 ± 0.09	0.002	**
			Bacteria	0.18 ± 0.07	0.02	*
			S plants	0.15 ± 0.07	0.04	*
			Bare soil	-0.12 ± 0.08	0.12	
			Sun hours	-0.11 ± 0.07	0.14	
			WHC	0.12 ± 0.08	0.14	
			BD	-0.11 ± 0.09	0.21	
			Management index	-0.06 ± 0.07	0.40	
S	0.39	0.19	**S plants**	0.20 ± 0.08	0.02	*
			Sb	-0.18 ± 0.08	0.03	*
			Bare soil	-0.15 ± 0.08	0.07	
			N_{min}	0.14 ± 0.08	0.08	
			MSIR	0.14 ± 0.08	0.09	
			Remove leaves	-0.12 ± 0.09	0.17	
$E_{Shannon}$	0.54	0.36	**Urban warming**	-0.46 ± 0.08	<0.001	***
			MSIR	-0.19 ± 0.07	0.008	**
			PSV	0.16 ± 0.07	0.02	*
			S	0.16 ± 0.07	0.02	*
			TED	-0.14 ± 0.07	0.04	*
			Management index	0.09 ± 0.07	0.21	
PSV	0.29	0.10	**Urban warming**	-0.24 ± 0.09	0.007	**
			Fertiliser	0.21 ± 0.09	0.02	*
TED	0.22	0.07	Land-use types: grass	-0.38 ± 0.2	0.06	
			S	0.11 ± 0.09	0.20	
			TON	0.10 ± 0.09	0.24	
			Urban warming	-0.10 ± 0.09	0.25	
			Land-use types: forbs	-0.01 ± 0.20	0.98	

5

Direct and indirect effects of urban gardening on aboveground and belowground diversity influencing soil multifunctionality

Text mining: Most frequent 100 words from chapter 5 (shape: leaf), created with the "wordcloud2" R package (Lang 2016).

Direct and indirect effects of urban gardening on aboveground and belowground diversity influencing soil multifunctionality

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Scientific Reports
RESEARCH ARTICLE
Published: 05. July 2019
DOI: 10.1038/s41598-019-46024-y

Abstract

Urban gardens are popular green spaces that have the potential to provide essential ecosystem services, support human well-being, and at the same time foster biodiversity in cities. We investigated the impact of gardening activities on five soil functions and the relationship between plant (600 spp.) and soil fauna (earthworms: 18 spp., springtails: 39 spp.) in 85 urban gardens (170 sites) across the city of Zurich (Switzerland). Our results suggest that high plant diversity in gardens had a positive effect on soil fauna and soil multifunctionality, and that garden management intensity decreased plant diversity. Indices of biological activity in soil, such as organic and microbial carbon and bacterial abundance, showed a direct positive effect on soil multifunctionality. Soil moisture and disturbance, driven by watering and tilling, were the driving forces structuring plant and soil fauna communities. Plant indicator values proved useful to assess soil fauna community structure, even in anthropogenic plant assemblages. We conclude that to enhance soil functions, gardeners should increase plant diversity, and lower management intensity. Soil protective management practices, such as applying compost, mulch or avoiding soil tilling, should be included in urban green space planning to improve urban biodiversity and nature's contribution to people.

5.1 Introduction

Maintaining functional and biodiverse urban green spaces is fundamental for liveable cities (cf. SDG 11 (United Nations 2015b)). Urban gardens are a major component of urban green spaces in many countries (Loram et al. 2008, Edmondson et al. 2014). They are heterogeneous in structure, but despite their relatively small size they provide critical habitat resources and increase the connectivity of urban landscapes (Soanes et al. 2019). Garden management creates diverse garden land-use types including perennially vegetated habitats such as lawns or annually vegetated habitats such as vegetable beds (Loram et al. 2008). These diverse microhabitats support urban biodiversity and have the ability to provide nature's contributions to people (Owen 1991, Goddard et al. 2010). The worldwide increase in human population is expected to take place mainly in urban areas (Martellozzo 2012), while growing cities often expand onto fertile agricultural soils, thus challenging the supply of fresh food in the future (Tan and Jim 2017). There is a great potential for producing food in urban gardens and at the same time to provide other ecosystem services (ES) in densely populated cities (Endreny 2018). It is estimated that urban farming delivers food for approximately 800 million people (Lee-Smith 2010), although the current global scale is difficult to assess (Siegner et al. 2018). However, hundreds of millions of citizens rely on urban agriculture for part of their nourishment (Redwood 2009). Nonetheless, urban garden soils are also important for regulating soil functions such as water storage (flood control (Bolund and Hunhammar 1999)), C and N storage (Edmondson et al. 2012), pollination (Samnegård et al. 2011), soil formation (Levin et al. 2017), pest control (Frey et al. 2018), or to decrease urban heat island intensity (Susca et al. 2011) and provide habitats for many species even in densely urbanised areas (Goddard et al. 2010). From a sociological perspective, urban gardens are important for recreation, well-being, and social interaction (Hofmann et al. 2018). Urban gardening has a long tradition in many countries around the world (Bardgett 2016). As a consequence of decades of beneficial soil management practices, such as the application of

compost (Cogger 2005), urban garden soils may not always be as poor in quality and potentially polluted as other urban soils (Edmondson et al. 2014, Tresch et al. 2018a). Despite the importance of gardens for urban biodiversity (Goddard et al. 2010), information on the ecological importance of allotment and domestic gardens is still scarce compared to public green spaces (Loram et al. 2008, Cabral et al. 2017). However, there is a large body of evidence that biodiversity drives ecosystem processes and related services in aboveground communities (Hector and Bagchi 2007), but the functioning of belowground biodiversity is much less understood (Wall et al. 2010). Although it has been shown that soil biodiversity is linked in multiple ways with aboveground biodiversity (Fierer et al. 2009, Buchholz et al. 2017), further investigation is needed to better understand these relationships. Garden soils are strongly influenced by human activities (Edmondson et al. 2014, Amossé et al. 2016, Tresch et al. 2018a), but they are also affected by the past land-use, the degree of disturbance or climate related drivers such as the urban heat island effect (Lorenz 2017). Soil functions are provided and controlled by a large variety of soil organisms (Bardgett and Van Der Putten 2014), also in urban soils (Amossé et al. 2016), where the frequency of soil disturbance is often high (Lorenz 2017). Changes in community composition of soil fauna in both alpha and beta diversity (Mori et al. 2018), for instance due to soil disturbance, can impair soil functions such as organic matter decomposition or nutrient retention (Wagg et al. 2014). The interactions of aboveground and belowground species, driving ecosystem functions, at least at the local scale (Dedeyn and van der Putten 2005), are mainly linked via plants (Wardle 2002). However, still very little is known about this relationship between aboveground and belowground diversity and associated soil functions (Morriën et al. 2017), especially for garden soils (Vauramo and Setälä 2011).

The ability of an ecosystem to provide multiple functions, so-called multifunctionality (Hector and Bagchi 2007), can be calculated as indices based on the functions of interest (Byrnes et al. 2014). Such measures of multifunctionality (i.e. the averaging approach), have been used to analyse a wide range of ecosystem drivers (Manning et al. 2018), such as soil characteristics (Mori et al. 2016), habitat diversity (Alsterberg et al. 2017), climate (Delgado-Baquerizo et al. 2016), or management practices in agriculture (Allan et al. 2015) and even in constructed ecosystems such as green roofs (Lundholm 2015). Here, we focus on five independent measurements for calculating soil multifunctionality ranging from aboveground (Wall et al. 2008) and belowground (Buchholz et al. 2017) litter decomposition, to nutrient supply for plant growth (Van Eekeren et al. 2010) and water regulation, such as water storage capability (Egerer et al. 2018a).

Research on urban garden soils has recently received increased attention (Smetak et al. 2007, Edmondson et al. 2014, Amossé et al. 2016, Joimel et al. 2017, Frey et al. 2018), especially with regard to human health and well-being (Lorenz 2017). However, our understanding of the complex interactions between management practices, soil biodiversity and soil functioning is still scarce (Setälä et al. 2016). In this study, we focus on gardening activities in the two most dominant garden types of Zurich (CH), allotment and domestic gardens, and assess the interactions between aboveground diversity of plants and belowground diversity of soil fauna. We investigated earthworms (Oligochaeta: Lumbricidae), representing soil macrofauna species and springtails (Hexapoda: Collembola), representing soil mesofauna species, as indicators for soil functioning (Gobat et al. 2004) and assessed the impacts of urban gardening on soil multifunctionality. Earthworms are generally described as ecosystem engineers (Gobat et al. 2004), due to their impact on soil structure and quality, at least in temperate soils (Blouin et al. 2013). They are important indicator organisms for soil functions (Le Bayon et al. 2017), soil disturbance, and management practices (Gobat et al. 2004). It has been shown that also in urban ecosystems such as parks or urban gardens, they are sensitive indicators of anthropogenic management intensity (Smetak et al. 2007). Springtails are a key group of microarthropods (Gobat et al. 2004) and can be used as indicators of sustainable land-use, soil quality (Buchholz et al. 2017), or the use of pesticides (de Lima e Silva et al. 2017). Moreover, they are used to assess soil functionality (dos Santos et al. 2018) and the impact of environmental factors (Hopkin 1997) on soil biodiversity. In addition, we assessed soil microfauna by biological soil measurements, such as basal respiration, microbial biomass and gene copy numbers of bacteria and fungi (Table 5.1).

The overall objective of our study was to investigate impacts of garden management practices (management intensity index, garden land-use types) on aboveground plant diversity and belowground diversity of soil fauna, and their direct and indirect effects on soil multifunctionality. We hypothesised (cf. a priori structural equation model (SEM) Figure 5.1) that (i) intensive soil management will re-

duce the diversity of both plant and soil fauna and negatively affect soil multifunctionality (arrows 1 & 2). We assumed that (ii) aboveground and belowground diversity are also linked in urban garden ecosystems and therefore expected that a higher diversity of plants will have a positive effect on both soil fauna and soil multifunctionality (arrows 3). Furthermore, we expected that (iii) soil fauna diversity (arrow 4a) and biomass (arrow 4b) will have a direct positive effect on soil multifunctionality. Additionally, we assumed an influence of (iv) soil characteristics and (v) urbanisation on soil multifunctionality (arrows 5 & 6).

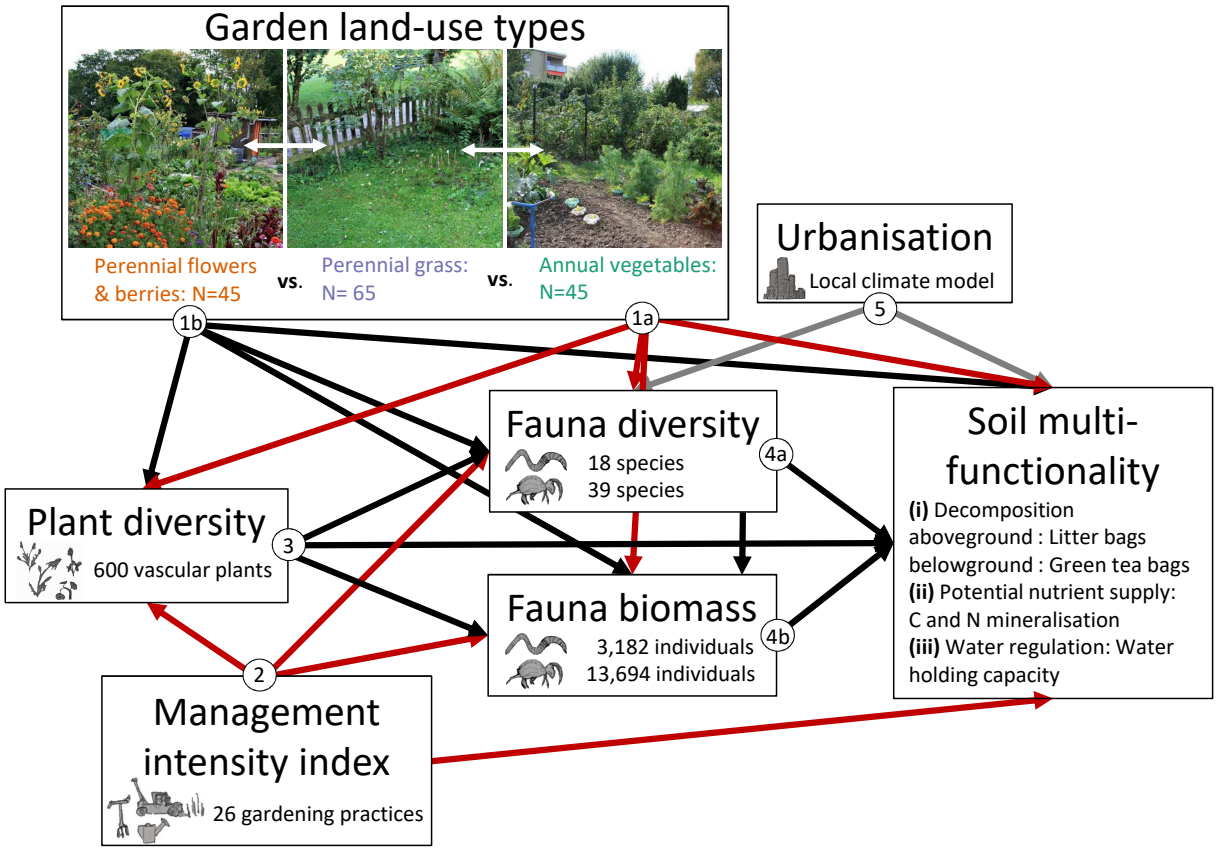

Figure 5.1 – A priori SEM model with hypothesised direct and indirect effects of urban gardening on soil multifunctionality. Expected positive relationships are given in black and negative ones in red, grey arrows represent both positive and negative effects. We expected that annual vegetables (arrows 1a) will negatively influence plant and soil fauna as well as soil multifunctionality compared to perennial grass sites, while perennial flowers (arrows 1b) will show positive effects. Management intensity (arrows 2) is expected to negatively affect plant diversity and soil fauna as well as soil multifunctionality. Higher plant diversity (arrows 3) is hypothesised to have a positive effect on soil fauna and soil multifunctionality. Soil fauna diversity and biomass (arrows 4a & 4b) are also expected to have a positive effect on soil multifunctionality. Urbanisation (arrows 5) might have a positive or negative effect on soil fauna and soil multifunctionality. Expected effects of soil characteristics (arrows 6) can be found in Figure S 5.9.

In a second step, we analysed soil fauna community structure. We expected that frequently disturbed soils would have the lowest species diversity within (alpha diversity) and among (beta diversity) garden sites, including a high community evenness and beta diversity mainly driven by species loss (nestedness) rather than species replacement (turnover). For the plant community, we expected highest alpha and beta diversity for garden sites with high planting activities, including a high species turnover component for beta diversity. Furthermore, we investigated impacts of management practices on soil fauna community composition and on soil fauna disturbance indices.

5.2 Results

Urban gardening effects on soil fauna and soil multifunctionality

The SEM based on our a priori expectations (Figure 5.1) of urban gardening effects on aboveground and belowground diversity and soil multifunctionality met the criteria of Fisher's C statistic (Shipley 2000) (Fisher's C= 30.7; p= 0.80; AICc= 286.8). The model included one significant missing path (Lefcheck 2016) between PC1 and soil fauna diversity (SEM; 0.18; p=0.03). With the inclusion of this path the overall model fit of the SEM improved (Fisher's C= 24.3; p= 0.93; AICc= 288.3), with marginal differences in the AICc (1.5). Overall, the strongest relationships in the SEM originated from garden land-use types (Table 5.3), influencing plant diversity, fauna biomass and soil characteristics (PC2) and soil multifunctionality both in indirect and direct ways (Figure 5.2 & S10). The strongest effects on soil multifunctionality came from soil PC1 (SEM; -0.61; p=<0.001), represented by lower loads of C_{mic}, C_{org}, bacteria, Fe and K, but higher soil bulk density values (Figure S 5.4). Soils with increased C_{mic}, C_{org}, bacteria, Fe and K, but lower bulk density values thus covaried with higher soil multifunctionality. *Annual vegetable* sites showed lower soil multifunctionality values (SEM; -0.40; p= 0.03) compared to *perennial grass* sites. Moreover, we found positive effects of plant diversity (SEM; 0.17; p= 0.01), and fauna biomass (SEM; 0.17; p= 0.02) on soil multifunctionality. Taken together, both significant and non-significant effects explained 74 % of the total variation of soil multifunctionality. In addition, we also identified several indirect effects on soil multifunctionality (Figure 5.2 & S 5.10, Table S 5.7). We found that plant diversity had a positive indirect effect on soil multifunctionality mediated by increased fauna diversity and fauna biomass. Plant diversity itself was positively affected by *flower & berry* sites (SEM; 0.37; p= 0.04) and negatively by management intensity (SEM; -0.22; p= 0.01), explaining 39 % of the variation in plant diversity. A similar pattern was found in high beta diversity values (Table S 5.5) for the plants (0.97 ± 0.001), dominated by a high turnover component (0.92 ± 0.001) and low nestedness component (0.02 ± 0.001), indicating the high variability between garden plots. Moreover, the management intensity indirectly negatively affected soil multifunctionality by decreasing fauna biomass and plant diversity (Figure 5.2, Table 5.3). Plant and soil fauna beta diversity and fauna phylogenetic diversity were not included in the final SEM (cf. Table S 5.7, Figure S 5.7) due to a large increase in the AICc (588.5) and because it explained only 3 % more variance in soil multifunctionality (Figure S 5.8).

Urban gardening effects on soil fauna community composition

We investigated the effects of management practices, plant ecological indicators (Table 5.2), soil characteristics (Table 5.1), garden land-use types, and urbanisation on the community composition of both earthworms and springtails (Table 5.4). Both soil fauna groups were strongly affected by the plant ecological indicator nutrients (PERMANOVA EW; F= 5.9; p=<0.001, COL; F= 5.4; p=<0.001), that represented the requirements of nutrient-rich soils for plants that are predominantly found in *vegetable* sites (Figure 5.3). Earthworm communities were further affected by the penetration resistance of the soil (PERMANOVA EW; F= 5.3; p=<0.001), favouring anecic species such as *L. terrestris*. Furthermore, plants indicating humus rich soils (PERMANOVA EW; F= 3.0; p=0.01) favoured the two most abundant earthworm species (Table S 5.1, Figure 5.3) *A. chlorotica* and *A. caliginosa*. Those soils were further associated with higher contents of Mg (PERMANOVA EW; F= 2.1; p=0.04) and K (PERMANOVA EW; F= 2.7; p=0.01), resulting from higher soil disturbance (PERMANOVA EW; F= 2.4; p=0.02) predominantly in *vegetable* sites. In summary, the NMDS ordination was driven by endogeic species *A. chlorotica* (NMDS; R^2= 0.30; p=<0.001) and *A. caliginosa* (NMDS; R^2= 0.13; p=<0.001), anecic species *L. terrestris* (NMDS; R^2= 0.17; p=<0.001), and both endogeic (NMDS; R^2= 0.20; p=<0.001) and anecic (NMDS; R^2= 0.19; p=<0.001) juveniles. Although the garden land-use type was not a significant factor affecting the earthworm community composition, the most abundant species can be allocated to specific land-use types (Figure 5.3, Table S 5.1). In *vegetable* sites we primarily found endogeic species such as *A. chlorotica* (64.5 %), which is tolerant to disturbances (Amossé et al. 2016) and a pioneer species (Schomburg et al. 2018), endogeic juveniles (41.9 %) and *A. caliginosa* (45.1 %). *Perennial grass* sites were dominated by anecic species such as *L. terrestris* (52.5%) and anecic juveniles (47.6 %), probably due to deeper and more compacted soils. As expected, we found the lowest earthworm diversity ($D_{Simpson}$) in *vegetable* sites (1.85 ± 0.2) and the highest in *grass* sites (2.57 ± 0.2). The earthworm diversity was driven by endogeic and anecic species but not by epigeic species (Table S 5.5). Additionally, we found the lowest

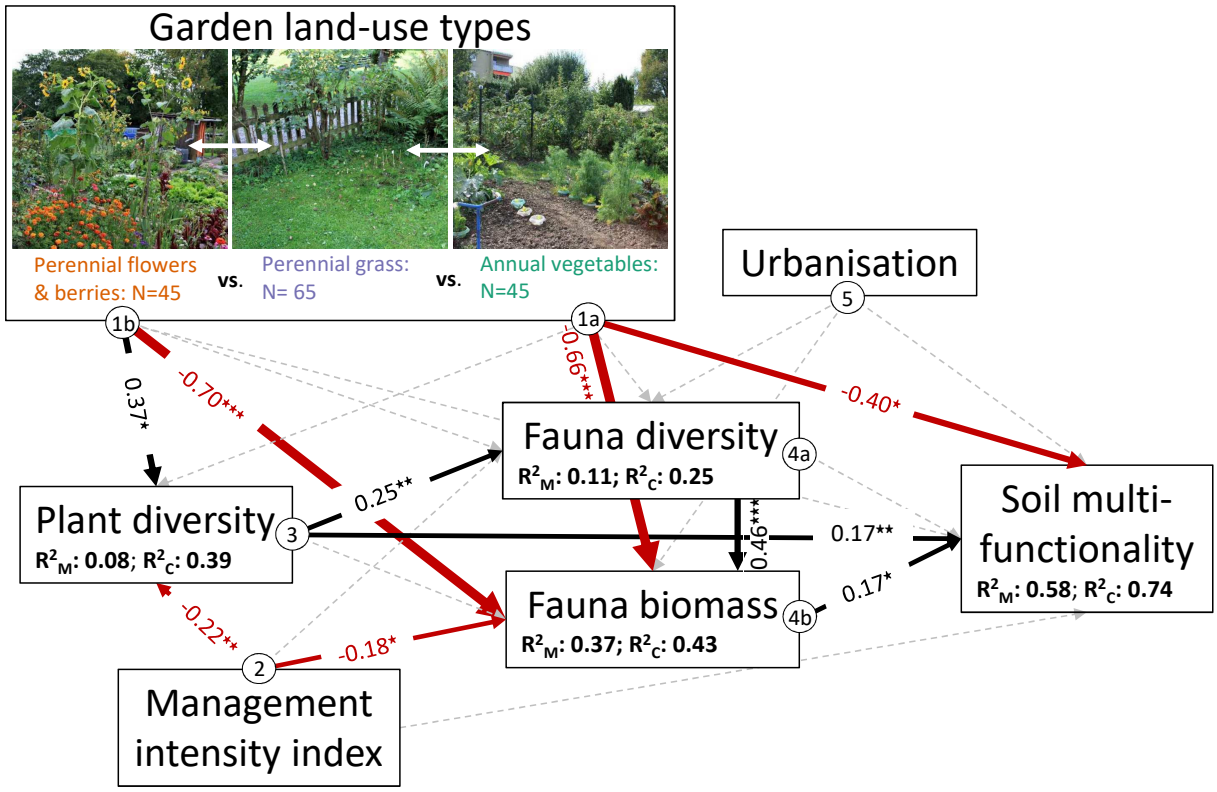

Figure 5.2 – Final most parsimonious SEM connecting garden management (land-use types, management intensity), urbanisation, plant and soil fauna diversity with soil multifunctionality (AICc=288.3, Fisher's C=24.3, P=0.93). Arrows represent unidirectional relationships among variables. Black arrows denote significantly (p<0.05) positive and red arrows significantly negative relationships (Table 5.3). Dashed grey arrows represent non significant relationships (p>0.05). The thickness of paths has been scaled based on the magnitude of the standardised regression coefficient. Conditional R^2s, based on the variance of both the fixed and random effects, as well as marginal R^2s, based on the fixed effect parts for each component models are given in the boxes of the response variables. Soil multifunctionality consists of five measurements related to important soil functions. Soil characteristics are included in Figure S 5.10.

beta diversity (β_{JAC}) for earthworms in *vegetables* and the highest in *grass* sites (Figure S 5.5, Table S 5.5). However, evenness ($E_{Simpson}$) was not highest in *vegetable* sites but in *grass* sites and *flower & berry* sites, where we also observed higher nestedness components (β_{JNE}).

Springtails were affected by the plant ecological indicators moisture (PERMANOVA COL; F= 3.7; p=<0.001) and moisture variability (PERMANOVA COL; F= 3.7; p=<0.001). These indicate moist soil or alternating soil moisture conditions, favouring species that were more abundant in *grass* sites such as *S. aureus* (81.1 %) or *P. alba* (67.3 %) on soils with high penetration resistance (PERMANOVA COL; F= 2.3; p= 0.01) and C_{mic} (PERMANOVA COL; F= 2.1; p= 0.02). All three life forms of springtails were present in the species that had the biggest effect on the community composition (Table S 5.4). *B. hortensis* showed the largest effect (NMDS; R^2= 0.29; p=<0.001), and was most often found in *vegetable* sites (75.5 %), where also *C. thermophilus* (NMDS; R^2= 0.13; p=<0.001) was often present (56.7 %), correlating with potassium loads (PERMANOVA COL; F= 2.5; p= <0.001). Other species driving the community composition of springtails were mostly found in *grass* sites, such as *P. notabilis* (85.4 %), representing the most abundant (22.9 %) springtail species (13,435 individuals) in this survey. Moreover, we found eight springtails (marked with stars in Table S 5.1) which were not included yet in the Fauna Europaea species list, with two new records for Switzerland (*I. balteatus* and *I. graminis*) according to the available literature and expert opinion (c.f Table S 5.1). The separation of *flower & berry* sites in the NMDS was mainly driven by *C. denticulata* (NMDS; R^2= 0.08; p=<0.001). Moreover, we found a clear effect of garden land-use type (PERMANOVA COL; F= 2.8; p=<0.001), but also two significant effects of specific garden management practices: applying water (PERMANOVA COL; F= 2.5; p= <0.001) and weeding (PERMANOVA COL; F= 1.8; p= 0.04). Weeding was more attributed to *vegetable* sites and applying water to *grass* sites in the NMDS ordination, whereas *flower and berry* sites were associated with a higher degree of urbanisation (urbanisation;

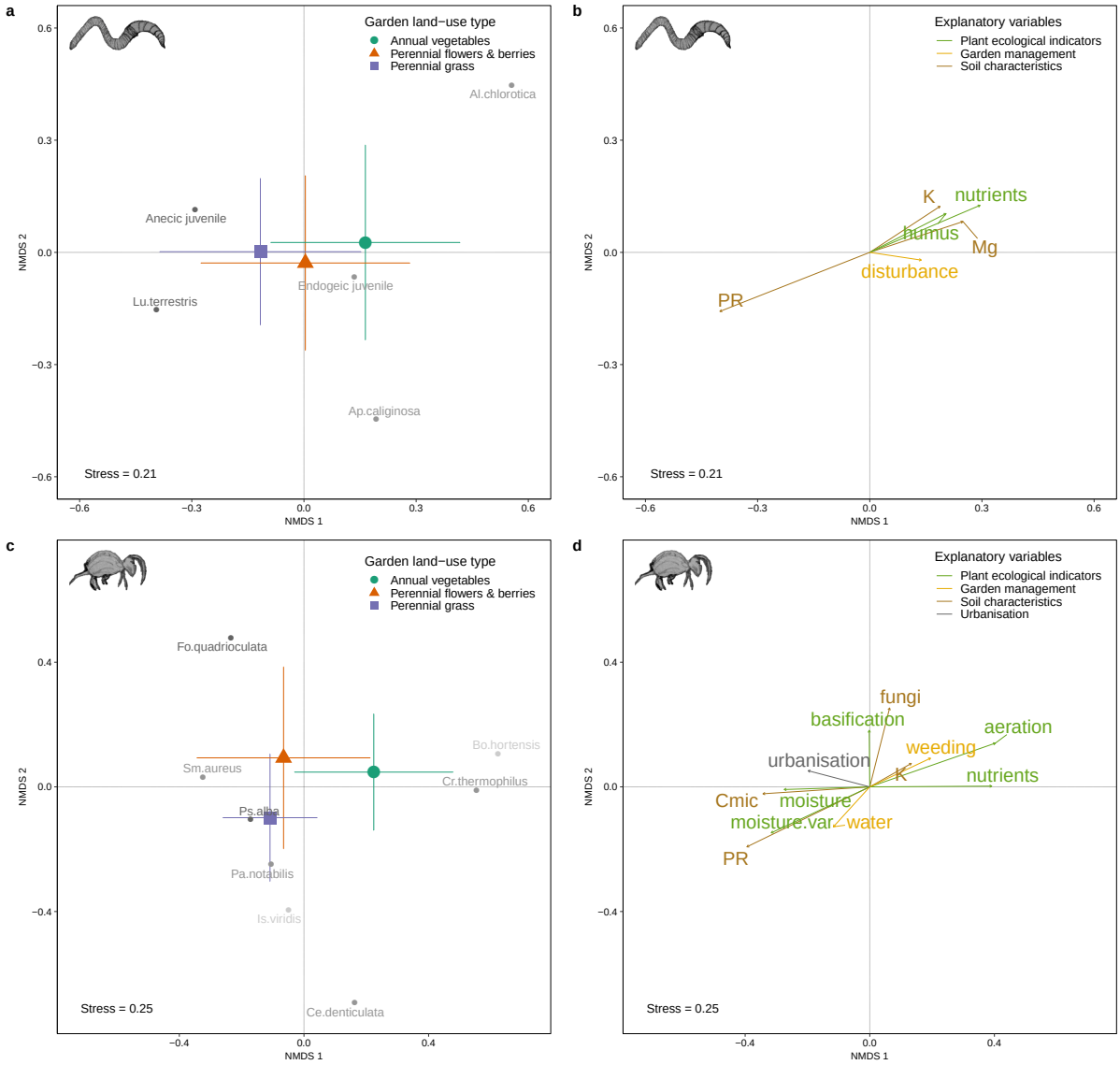

Figure 5.3 – Soil fauna community structure analysis of earthworms (a,b) and springtails (c,d). Soil fauna species are coloured in grey, corresponding to three ecological categories (Table S 5.1). Only species with a significant ($p<0.001$) effect (Table 5.4) and only significant explanatory variables ($p<0.05$) from the PERMANOVA model (Table S 5.1) were fitted. Garden land-use types include standard deviation bars.

PERMANOVA COL; F= 1.8; p= 0.04) and with more alkaline soils (basification; PERMANOVA COL; F= 2.0; p= 0.02). As expected, we found lower mean values for the springtail diversity ($D_{Simpson}$, Table S 5.2) in *vegetable* sites (3.3±0.2) compared to *grass* sites (3.8±0.1). Beta diversity (β_{JAC}) was highest for *flower & berry* sites with a high turnover in comparison to the nestedness component (Figure S 5.5, Table S 5.5). Springtail evenness ($E_{Simpson}$) was highest in *flower & berry* sites, where we also found the highest nestedness component (β_{JNE}), probably due to hemiedaphic and euedaphic species being more similar in *flower & berry* sites.

Additionally, we found differences between garden land-use types in soil fauna disturbance indices. The collembolan ecomorphological index and the earthworm anecic to endogeic ratio were lowest in *vegetable* sites (Table S 5.2). The acari to collembola ratio was lowest in *grass* sites and the fungal to bacterial ratio was highest in *vegetable* sites.

5.3 Discussion

Worldwide, there is a growing interest of city administrations in the socio-economic and ecological benefits of urban gardens (Redwood 2009, Goddard et al. 2010, Siegner et al. 2018, Frey et al. 2018, Cabral et al. 2017, Lorenz 2017, Lin et al. 2018). We investigated impacts of garden management practices on aboveground and belowground diversity and inter-

linked soil functions. The SEM (Figure 5.2, Table 5.3) revealed direct effects on soil multifunctionality and indirect effects mediated by soil fauna. Overall, our results showed that the largest effects on soil multifunctionality were caused by specific soil characteristics. Soils showing high biological soil quality indices such as organic and microbial carbon and bacteria increased the potential for soil multifunctionality. This probably originates from organic gardening practices such as the application of compost, due to the correlation with increased potassium loads and with decreased bulk density values (Table 5.3, Table 5.1), also influencing soil quality (Tresch et al. 2018a). The second strongest effect on soil multifunctionality was caused by the cultivation of vegetables and legumes in *annual vegetable* sites (hypothesis (i), Figure 5.1 arrows 1a), probably due to the frequent soil disturbance and the unprotected open soils in comparison to *perennial grass* sites. The cultivation of flowers and berries increased plant diversity (Figure 5.1 arrows 1b), but decreased soil fauna biomass compared to *grass* sites. Urban gardens with higher plant diversity (hypothesis (ii), Figure 5.1 arrows 3) increased soil multifunctionality directly, and indirectly through increasing fauna diversity and thus fauna biomass. The general pattern of enhanced soil multifunctionality with increased plant diversity is in line with results found in other ecosystems such as croplands, shrublands, grasslands, and forests, where plant diversity increased ES such as pollination, C storage, pest control, and productivity (Isbell et al. 2017a, Chen et al. 2018). Contrary to our expectations, we found no significant direct effect of management intensity on soil multifunctionality, but more intensively managed sites decreased plant diversity and fauna biomass. A similar relationship of management intensity and decreased diversity has been observed in urban lawns. Lerman et al. (2018) showed that mowing only every three weeks instead of every week increased the numbers of flowers by 2.5 times and thus the abundance and diversity of bee populations. Although Tresch *et al.* showed that aboveground (Tresch et al. 2019) and belowground (Tresch et al. 2018a) organic matter decomposition increased with urbanisation, there was no significant effect of urbanisation (hypothesis (v), Figure 5.1 arrow 5), on soil multifunctionality.

The structure of earthworm and springtail communities were influenced by plant ecological indicators (Figure 5.3, Table S 5.4), representing the living conditions of plants. Interestingly, springtails were more affected by plant ecological indicators than earthworms, highlighting the dominant influence of plants on springtails (Gobat et al. 2004). As expected, we found a lower alpha diversity of earthworms and springtails in *vegetable* sites, likely due to the high soil disturbance. Beta diversity was constantly high in both soil fauna and plant communities, driven by high turnover and low nestedness components. The plant community composition was shaped by the high species turnover between the garden sites, with highest dissimilarities for *flower & berry* sites. As expected, these significant differences originated most likely from planting and other garden management practices leading to site specific community compositions. The beta diversity values for both fauna communities were lowest in *vegetable* sites and peaked for earthworms in *grass* sites and for springtails in *flower & berry* sites, reflecting the different ecological strategies of earthworms and springtails.

Earthworms are important indicators for soil functioning (Blouin et al. 2013, Le Bayon et al. 2017). Functional groups of earthworms have been used to detect impacts of cultivation in different soils such as pastures, orchards or forest soils, while the ratio of anecic to endogeic species was used as an indicator of contaminated soils (Pérès et al. 2011), or soil disturbance (Fournier et al. 2012). In the frequently disturbed *vegetable* sites we found the lowest values for the earthworm anecic to endogeic ratio and the collembolan ecomorphological index, indicating a decreased soil biological quality due to soil disturbance (Joimel et al. 2017). Other studies have reported the highest value of the collembolan ecomorphological index in urban vegetable gardens and forest sites (Joimel et al. 2017). However, the highest value found in forest sites (2.3) was still considerably lower than the average we found in urban gardens of Zurich (5.8 ± 0.1), with a high number of euedaphic springtails such as *P. pulvinata*, *F. quadriculata*, and *I. minor*. Springtail abundance often increases from agricultural to forest sites (Sousa et al. 2006). Here they increased form *vegetable* to *grass* sites by a factor of 4.3. Besides soil disturbance, the increased abundance in grass sites could be explained by the higher plant cover of the perennially vegetated sites (Sousa et al. 2006) and because grass strips offer a variety of microhabitats for soil mesofauna species (Joimel et al. 2017). In contrast, Joimel et al. (2017) found higher mean densities in vegetable beds than forest or grassland sites, underlining the quality of urban gardens for soil fauna biodiversity and soil quality. In addition, the increased organic matter content in urban garden soils (Edmondson et al. 2014) can be an important factor for the high soil fauna diversity, since the input of organic matter in garden soils can

be higher than in agricultural fields (Joimel et al. 2016). Moreover, high management intensity is known to decrease soil mesofauna diversity (Cluzeau et al. 2012). We found a lower acari to collembola ratio in *grass* compared to *flower & berry* sites, which is in line with the dominance of acari in frequently disturbed arable or vineyard soils (Joimel et al. 2017). Additionally, we found a higher springtail biomass for *grass* sites, while earthworm biomass was at a similar and comparably high level in all urban garden land-use types. For instance, earthworm abundance (227.4±15.5) was considerably higher than mean reference values for biological soil quality indicators found in grass or cropland soils (Krüger et al. 2018). Referring to all micro-organisms (Cluzeau et al. 2012), C_{mic} peaked for grass sites, while the mean value (780.9±21.3) was higher than in cropland soils (341 mg kg^{-1}), but lower than in grassland soils (1249 mg kg^{-1}) found in Belgium (Krüger et al. 2018) or Switzerland (Oberholzer and Scheid 2007) (2077 mg kg^{-1}). This pattern of C_{mic} reflected management practices such as fertilisation or tillage (Cluzeau et al. 2012). Additionally, the composition of soil microorganism communities is an important driver for soil functioning (Morriën et al. 2017). For instance, a shift in fungal composition or activity can increase carbon uptake and nutrient cycling (Morriën et al. 2017). Both soil disturbance (Parfitt et al. 2010) or lower plant diversity (Eisenhauer et al. 2017) can result in decreased fungal to bacterial ratios. While increasing fungal to bacterial ratios can be expected from desert to temperate grassland and forest soils, assuming that grassland soils are more bacteria dominated than forest soils (Fierer et al. 2009). Here, we found an increased fungal to bacterial ratio for *vegetable* sites, due to the increased fungal and decreased bacterial gene copy numbers in those sites (cf. Table S 5.9, Figure S 5.13). This might be related to the input of compost on the *vegetable* sites or the increased plant diversity compared to *grass* sites.

The intuitive and rather simple concept of multifunctionality (Hector and Bagchi 2007) and its reduction to one single metric, such as the averaging approach (Byrnes et al. 2014), needs to be examined critically. For example, the functions and methods to measure them must be carefully selected (Allan et al. 2015). The strength of the biodiversity ecosystem multifunctionality relationships depends on the number of included functions, which was generally stronger when more functions were considered (Meyer et al. 2018). Another point is that the aggregation of multiple functions into one single metric can obscure information about potentially contrasting single functions (Bradford et al. 2014). The highest correlations among the soil functions (cf. Figure S 5.14) were found between C_{min} and N_{min} (r= 0.45, p= <0.001), both used to calculate soil nutrient supply, and between C_{min} and WHC (r= 0.43, p= <0.001). All other correlations (r<0.27) claimed a certain independence of the selected soil functions. The moderately positive correlation of all components to soil multifunctionality is required, because negative correlations among functions can be a limitation for multifunctionality assessments (Meyer et al. 2018). However, this multifunctionality assessment framework could also be used in future studies to assess the impact of managed urban green spaces on nature's contributions to people in cities.

With this city-wide assessment of the effect of urban gardening practices on aboveground and belowground diversity of plants and soil fauna, we demonstrated the potential impacts of gardeners' decisions on the quality and functioning of the soil and implications on the biodiversity of a city. In conclusion, our study suggests that a higher plant diversity can directly or indirectly increase soil multifunctionality by enhancing soil fauna diversity and biomass. In a previous study, intensive garden management decreased soil quality indices (Tresch et al. 2018a). Here we demonstrated that a high garden management intensity indeed also declined plant diversity and soil fauna biomass, with negative impacts on soil multifunctionality. In addition, we analysed drivers shaping soil fauna community structure of earthworm and springtail species. We showed that both were affected by plant ecological indicators, soil characteristics, and management practices such as the frequency of soil disturbance or applying water. We conclude that increasing plant diversity together with soil protective management practices have the potential to increase soil functions as well as foster biodiversity, and to create more biophilic (Lin et al. 2018) urban gardens, supporting human well-being and the ecological value of urban green spaces. Even though soil is a key resource in cities, it has not been integrated in most urban green space plans (Teixeira da Silva et al. 2018), thus we recommend that urban gardens including ecological management practices should be integrated in future green city strategies.

5.4 Methods

Study design and gradients

This study took place in 85 urban gardens of the city of Zurich, Switzerland (Figure S 5.1). We selected

gardens based on three independent criteria (Tresch et al. 2018a, Young et al. 2019): (i) the type of garden (domestic N= 43 vs. allotment; N= 42 Figure S 5.2), (ii) the management intensity (such as intensively managed vegetable or flower beds or extensively managed meadows), and (iii) the degree of urbanisation, ranging from densely built-up to peripheral areas within the city boundaries. In each garden two sampling plots (2 m x 2 m) with different land-use management were selected (Table S 5.12), belonging to one of the following three categories: annual vegetable beds (*vegetables*; N= 47), perennial flowers and berries (*flowers & berries*; N= 52) or perennial lawn and meadows (*grass*; N=71), reflecting the most dominant garden land-use types in Zurich and in many other cities.

Garden management practices were assessed using a questionnaire with 26 management questions, specific for each land-use type, ranging from the frequency of lawn cutting to fertiliser application or weeding (Table S 5.3). Garden management intensity was assessed as the sum of 26 management questions. In addition, five common management practices (disturbance, fertiliser, pesticides, water, weeding; Table S 5.2) were used in the community composition analysis. Urban warming was used as a proxy for urbanisation due to the correlation with the amount of built-up and paved area for different radii (30 - 500 m) around the gardens (Tresch et al. 2019). It has been assessed as the deviation in local mean air temperatures at night near the surface based on a local climate model (Parlow et al. 2010), showing temperatures increased of up to 5°C for urbanised gardens.

Aboveground diversity

Plant diversity was assessed by a floristic inventory (Frey and Moretti 2019) of cultivated and spontaneously growing plants on each sampling plot (N=170). Based on this inventory of 600 plant species, we calculated plant alpha diversity as the total number of plant species per sampling plot and plant beta diversity as the mean of the pairwise Jaccard dissimilarity comparisons between each focal plot and all other plots (Villéger et al. 2013). Additionally, we used a six-point ordinal scale (Frey and Moretti 2019) to calculate community weighted mean values of seven plant ecological indicator values (Landolt et al. 2010) (Table S 5.2), reflecting the plant environmental requirements (Wildi 2016).

Belowground diversity

Earthworms were collected in a smaller subplot of 0.3 m x 0.3 m within the 2 m x 2 m sampling plots by a combined hand sorting and mustard extraction method (Tresch et al. 2018a). Earthworms were stored in 70 % ethanol (Schomburg et al. 2018), identified to the species level, and classified into three ecological categories (Table S 5.1): epigeic species (living in the litter layer, with little burrowing activity), endogeic species (living in the soil, with horizontal burrows) and anecic species (living in large and deep vertical burrows).

Springtails and mites were sampled with six undisturbed soil cores (5 cm diameter, 8 cm length, Eijkelkamp, NL) randomly taken in the 2 m x 2 m sampling plots (Tresch et al. 2019). Springtails were identified to the species level including life forms according to ecological and functional traits (Table S 5.1): epedaphic species (living in the upper litter layer), hemiedaphic species (living at the interface between litter and soil) and euedaphic species (soil-dwelling species).

We defined soil fauna diversity as the average proportional species richness across soil macrofauna (earthworms) and mesofauna (springtails) species following Allan et al. (2014). Soil fauna beta diversity was calculated as the average proportional species beta diversity of earthworms and springtails, while the individual measures of beta diversity per soil fauna group were computed as mean pairwise Jaccard dissimilarities, similarly to the plant beta diversity. Soil fauna biomass was calculated as the average proportion of biomass per m^2 of soil, with measured earthworm biomass [gm^{-2}] on an individual basis (including gut contents) and estimated springtail biomass (conversion factor of 5 µg for each springtail Petersen and Luxton (1982)).

Soil fauna disturbance indices

The adaptation of soil fauna to management practices was assessed with four soil fauna disturbance indices: the collembolan ecomorphological index (Joimel et al. 2017), the acari to collembola ratio (Parisi et al. 2005), the fungal to bacterial ratio (Fierer et al. 2009), and the earthworm anecic to endogeic ratio (Pérès et al. 2011).

Soil characteristics

Soil characteristics were assessed with a combination of three physical, six chemical and four biological soil measurements (Table S 5.1), representing

the most commonly used soil quality indicator measurements (Bünemann et al. 2018). The microbial community information of bacterial (16S) and fungal (18S) gene copy numbers were used to calculate the fungal to bacterial ratio. Measurement details can be found in Table S 5.11 and Tresch et al. (2018b).

Soil multifunctionality

Similar to other studies (Schuldt et al. 2018) we used the averaging approach (Byrnes et al. 2014) to calculate soil multifunctionality. It calculates the mean value across standardised soil functions for each sampling plot. In total, we used five measurements (Table S 5.12), which are related to important soil functions, for the computation of soil multifunctionality. The three assessed key soil functions are (i) aboveground and belowground litter decomposition, (ii) soil nutrient supply, and (iii) soil water storage and regulation. The soil function litter decomposition aboveground was measured by standardised leaf litter mass loss (*Zea mays* L.) in 4 mm mesh sized litter bags (Tresch et al. 2019), while belowground litter decomposition was measured by the net mass loss of green tea bags, buried in 8 cm soil depth (Tresch et al. 2018a). The supply of nutrients in the soil was assessed by the mineralisation rates of N (N_{min}) and C (C_{min}), and the capacity of the soil for water regulation, was measured by the water holding capacity (WHC).

Data analysis

Soil fauna diversity and biomass were calculated by taking species richness per taxonomic group, applying a standardisation for each taxonomic group scaled to a range from 0 to 1 ($f(x) = (x_i - x_{min})/(x_{max} - x_{min})$) and then averaging the values for each plot (Allan et al. 2014). Aboveground and belowground beta diversity were calculated as mean pairwise Jaccard dissimilarities comparing each focal plot to all other sampling plots (Villéger et al. 2013) using the R package 'betapart' (Baselga and Orme 2012). Soil multifunctionality was computed by scaling each of the five measurements of soil functions to a range from 0 to 1 (Schuldt et al. 2018) and deriving mean values across the standardised soil functions according to the averaging approach (Byrnes et al. 2014). Community weighted means of plant ecological indicators were calculated with the R package 'FD' (Laliberté and Legendre 2010).

We fitted a piecewise structural equation model (SEM), with the 'piecewiseSEM' package (Lefcheck 2016), to infer relative importance of direct and indirect effects of urban gardening, plant diversity, urbanisation and soil characteristics on soil fauna and soil multifunctionality. To address multicollinearity and reduce the amount of variables we applied a PCA for the soil characteristics and used the first four PCA axes, explaining 64.2 % (Table S 5.1; Figure S 5.4) of the variation (Kaiser-Guttman criteria). We used Shipley's d-separation test to identify missing paths in the SEM and the AIC_C for model comparison. We used linear mixed effect models (LMEM; lme(nlme; Pinheiro et al. (2018)) with the garden as random effect for each SEM component and reported standardised (scaled by mean and variance) path coefficients, as well as marginal R^2 and conditional R^2 based on fixed and random effects (Lefcheck 2016) (Table 5.3). Model assumptions were tested (Figure S 5.11) and potential spatial autocorrelation patterns were calculated with Moran's I autocorrelation indices and the spatial structure in the model residuals using semivariograms (Figure S 5.12).

We applied individual LMEM with garden identity as random effect and land-use types as response variables to assess changes in fauna and plant diversity and soil fauna disturbance indices. We checked for normal distribution, autocorrelation, and heteroscedasticity of the model residuals and applied a transformation ($log(x + 1)$) in the cases of: earthworm biomass, anecic to endogeic ratio, acari to collembola ratio and springtail biomass. We reported means and 95 % credible intervals of the Bayesian inference posterior distribution based on 10,000 independent simulations (Korner-Nievergelt et al. 2015). Soil fauna community structure was further analysed using a permutational multivariate analysis of variance (PERMANOVA, 10,000 permutations) with a Hellinger transformed Euclidean distance species matrix of earthworms (EW) and springtails (COL) and a non-metric multidimensional scaling (NMDS) using the 'vegan' package (Oksanen et al. 2017). For the NMDS only significant variables from the PERMANOVA were fitted. Data management and statistical analyses are provided as an R project using R 3.4.2 (R Core Team, 2017).

Table 5.1 – Soil characteristics describing the soil quality of urban garden sites used as explanatory variables in the SEM. The first four PCA axes scores (PC1-PC4; Figure S 5.4) were used (Kaiser-Guttman criteria) as explanatory variables in the SEM (Figure 5.2), explaining 64.2 % of the total variation.

Variables	Description	PC1	PC2	PC3	PC4
Physical soil characteristics					
BD [g cm^{-3}]	Soil bulk density	0.39	-0.31	0.06	-0.15
PR [MPa]	Penetration resistance	0.03	0.39	0.03	-0.31
SA [%]	Soil stable aggregates	-0.26	0.44	-0.04	0.15
Chemical soil characteristics					
Fe [mg kg^{-1}]	Iron content	-0.41	-0.17	0.29	0.07
K [mg kg^{-1}]	Potassium content	-0.35	-0.33	0.03	-0.05
Mn [mg kg^{-1}]	Manganese content	0.01	0.13	0.41	-0.04
Mg [mg kg^{-1}]	Magnesium content	-0.13	-0.24	-0.44	0.18
P [mg kg^{-1}]	Phosphorus content	-0.22	-0.34	0.41	0.09
pH	Soil pH	0.18	-0.07	-0.49	-0.04
Biological soil characteristics					
C_{mic} [mg kg^{-1}]	Microbial biomass carbon	-0.37	0.35	-0.13	-0.07
C_{org} [%]	Soil organic carbon content	-0.39	-0.20	-0.33	0.18
Bacteria [gene copies]	16S bacterial gene copy number	-0.31	0.05	-0.11	-0.60
Fungi [gene copies]	18S fungal gene copy number	-0.03	-0.26	-0.04	-0.64
	Eigenvalue	2.7	2.4	1.7	1.5
	Explained variance [%]	20.8	18.4	13.3	11.7

Table 5.2 – Garden management practices based on the gardener survey (Table S 5.3) and plant ecological indicator values reflecting the plant environmental requirements (Wildi 2016). Plant ecological indicator values are calculated as community weighted means of plant species found on each sampling plot.

Variables	Description
Management practices	
Disturbance	Frequency of soil disturbance
Fertiliser	Frequency of fertiliser application
Management intensity	Garden management intensity gradient
Pesticides	Frequency of pesticide application
Water	Frequency of water application
Weeding	Frequency of weeding
Plant ecological indicator values	
Aeration	Supply of oxygen in the soil (from poor (0) to good (1))
Basification	Soil content of H^+-ions (from acid (0) to alkaline (1))
Humus	Dark organic matter content (humus) (from little (0) to high (1))
Moisture	Soil moisture during the growing season (from dry (0) to wet (1))
Moisture variability	Alternating soil moisture (from less (0) to often (1) alternating)
Nutrients	Soil nutrient availability (from low (0) to high (1))
Root depth	Depth of soil root penetration (from shallow (0) to deep (1))

Table 5.3 – Final most parsimonious structural equation model (SEM; AICc= 156.3, Fisher's C= 24.3, P= 0.93) indicating direct and indirect effects on soil multifunctionality from garden land-use types, garden management, plant and soil fauna diversity, soil fauna biomass, soil characteristics and urbanisation. R^2_M is based on fixed effects and R^2_C on fixed and random (garden ID) effects. Total estimates of indirect pathways are given in Table S 5.6.

Response	R^2_C	R^2_M	Predictor	Estimate ± SE	P	
Soil multifunctionality	0.74	0.58	Soil PC1	-0.61±0.06	<0.001	***
			Plant diversity	0.17±0.06	0.01	**
			Fauna biomass	0.17±0.07	0.02	*
			Vegetables	-0.40±0.20	0.03	*
			Soil PC3	0.12±0.06	0.08	
			Urbanisation	0.11±0.07	0.12	
			Soil PC2	0.12±0.08	0.15	
			Management intensity	0.08±0.06	0.22	
			Flowers & berries	-0.08±0.10	0.61	
			Soil PC4	-0.03±0.06	0.69	
			Fauna diversity	-0.01±0.06	0.83	
Fauna diversity	0.25	0.11	Plant diversity	0.25±0.09	0.005	**
			Soil PC1	-0.18±0.08	0.03	*
			Urbanisation	-0.15±0.09	0.09	
			Vegetables	-0.26±0.20	0.17	
			Management intensity	0.08±0.09	0.37	
			Flowers & berries	-0.02±0.20	0.92	
Fauna biomass	0.43	0.37	Fauna diversity	0.46±0.07	<0.001	***
			Flowers & berries	-0.70±0.20	<0.001	***
			Vegetables	-0.66±0.20	<0.001	***
			Management intensity	-0.18±0.07	0.01	*
			Plant diversity	0.09±0.07	0.20	
			Urbanisation	0.04±0.07	0.56	
Plant diversity	0.39	0.08	Management intensity	-0.22±0.08	0.01	**
			Flowers & berries	0.37±0.20	0.04	*
			Vegetables	0.29±0.20	0.09	
Soil PC1	0.48	0.02	Urbanisation	-0.12±0.10	0.24	
			Vegetables	0.17±0.20	0.30	
			Management intensity	-0.05±0.08	0.55	
			Flowers & berries	0.04±0.20	0.81	
Soil PC2	0.60	0.44	Vegetables	-1.60±0.10	<0.001	***
			Flowers & berries	-0.79±0.10	<0.001	***
			Management intensity	0.05±0.06	0.47	
			Urbanisation	-0.04±0.07	0.61	
Soil PC3	0.76	0.01	Vegetables	0.16±0.10	0.16	
			Management intensity	0.07±0.07	0.32	
			Flowers & berries	-0.02±0.10	0.86	
			Urbanisation	-0.01±0.10	0.90	
Soil PC4	0.47	0.02	Vegetables	0.22±0.20	0.18	
			Urbanisation	-0.09±0.10	0.40	
			Flowers & berries	-0.04±0.20	0.83	
			Management intensity	-0.01±0.08	0.93	

Table 5.4 – PERMANOVA of earthworms (PERMANOVA EW; left) and springtails (PERMANOVA COL; right) and management practices, plant ecological indicators, soil characteristics and garden characteristics as explanatory variables. SA: Soil stable aggregates, BD: Soil bulk density, PR: Penetration resistance.

		Earthworms				Springtails			
	Df	F	R^2	P		F	R^2	P	
Management practices									
Management Intensity	1	1.1	0.01	0.37		1.3	0.01	0.18	
Water	1	1	0.01	0.43		2.5	0.01	< 0.001	**
Fertiliser	1	1.8	0.01	0.09		1.4	0.01	0.14	
Pesticides	1	0.6	0.01	0.75		1.3	0.01	0.19	
Disturbance	1	2.4	0.01	0.02	*	1.2	0.01	0.24	
Weeding	1	1.6	0.01	0.11		1.8	0.01	0.04	*
Plant ecological indicators									
Moisture	1	1.5	0.01	0.15		3.7	0.02	< 0.001	***
Moisture Variability	1	1.9	0.01	0.06		3.7	0.02	< 0.001	***
Basification	1	1.4	0.01	0.18		2	0.01	0.02	*
Nutrients	1	5.9	0.04	< 0.001	***	5.4	0.03	< 0.001	***
Humus	1	3	0.02	0.01	**	0.6	0.01	0.82	
Aeration	1	1.2	0.01	0.29		1.8	0.01	0.05	*
Root depth	1	0.6	0.01	0.82		0.7	0.01	0.74	
Soil characteristics									
Physical measurements									
SA	1	1.9	0.01	0.06		1.8	0.01	0.05	
PR	1	5.3	0.03	< 0.001	***	2.3	0.01	0.01	**
BD	1	1.1	0.01	0.36		0.7	0.01	0.81	
Chemical measurements									
Mg	1	2.1	0.01	0.04	*	1.3	0.01	0.2	
P	1	1.1	0.01	0.32		0.6	0.01	0.81	
Fe	1	1.4	0.01	0.17		0.7	0.01	0.81	
K	1	2.7	0.02	0.01	*	2.5	0.01	< 0.001	**
pH	1	0.8	0.01	0.55		1.1	0.01	0.36	
Mn	1	0.7	0.01	0.72		0.7	0.01	0.79	
Biological measurements									
C_{org}	1	0.8	0.01	0.6		0.9	0.01	0.52	
C_{mic}	1	0.6	0.01	0.73		2.1	0.01	0.02	*
Fungi	1	0.8	0.01	0.62		1.8	0.01	0.04	*
Bacteria	1	0.7	0.01	0.71		0.8	0.01	0.63	
Garden characteristics									
Land-use type	2	1	0.01	0.41		2.8	0.03	< 0.001	***
Urbanisation	1	1.4	0.01	0.17		1.8	0.01	0.04	*
Residuals	119		0.72				0.69		

Acknowledgements

We are grateful to Adolphe Munyangabe and Anton Kuhn (soil measurements), Lena Fischer (tea bag index and earthworm extraction), Stefan Grubelnig and Reto Henzmann (soil and soil fauna sampling), Dr. Lukas Pfiffner (earthworm identification), Dr. Daniel Haefelfinger (litter bag decomposition), and Dr. Joerg Salamon (collembola identification) for their extraordinary support and help in the field or laboratory. In particular, we thank Dr. Robert Home for his support in finalising the manuscript and the 85 participating gardeners of this study for granting access to their gardens and their interest in promoting biodiversity in cities. We gratefully acknowledge the financial support for this interdisciplinary project BetterGardens provided by the Swiss National Science Foundation in frame of the Sinergia program (CRSII1_154416).

Author Contributions

AF, MM, RL, PM, DF, ST conceived and designed the research, ST performed both field and laboratory work with the help of BS and DF. ST analysed the data. All authors reviewed the manuscript.

Additional information

Supplementary information: The supplementary material for this article contains the raw data and a script with R codes used for the calculations.

Competing Interests: The authors declare that they have no competing interests.

5.5 Supplementary Tables and Figures

5.5.1 Supplementary Figures

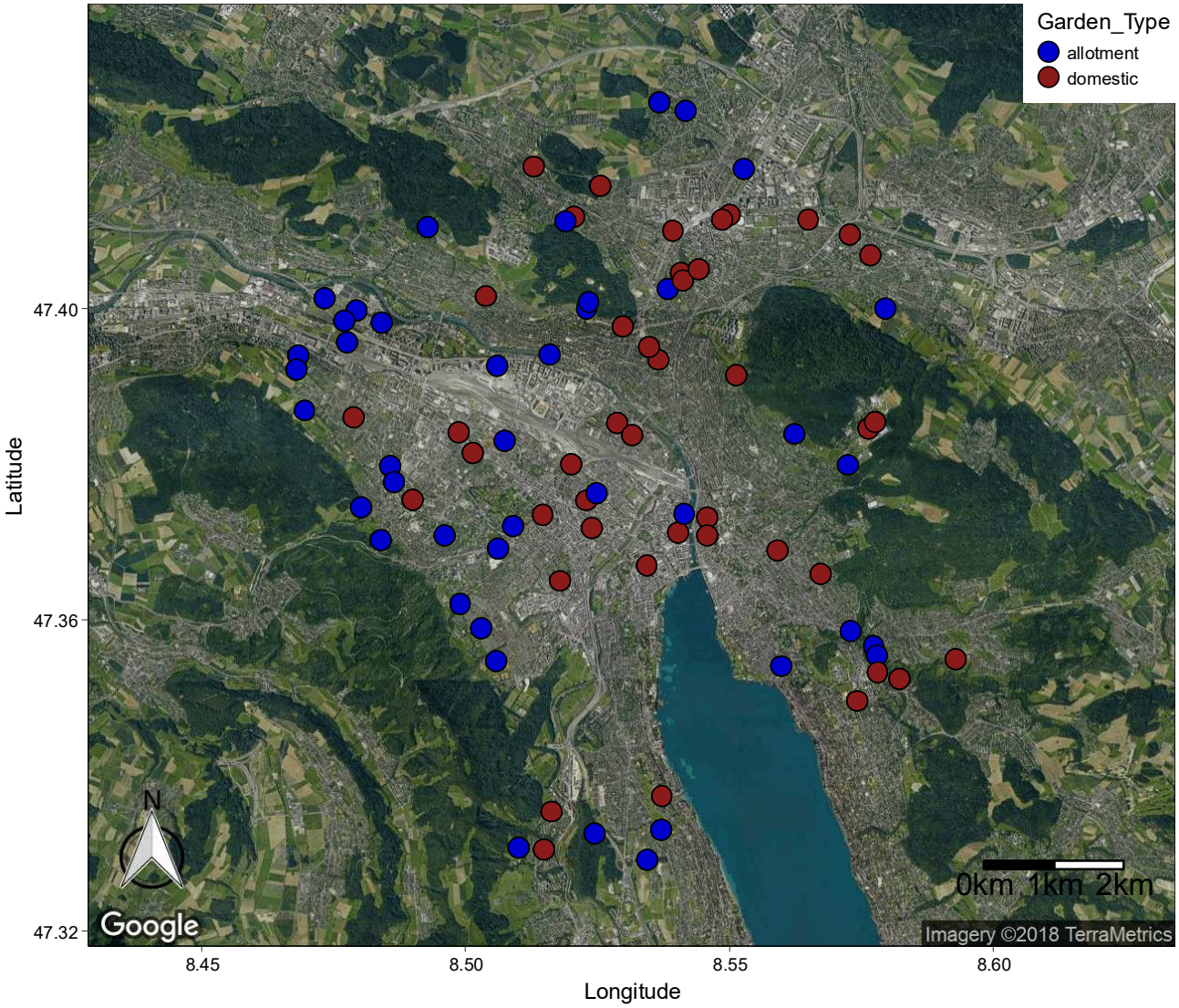

Figure S 5.1 – Urban gardens sampled in the city of Zurich. Allotment gardens are displayed in blue (N= 42) and domestic gardens in red (N= 43). Gardens were selected according to the garden type (domestic vs. allotment), the management intensity (extensive vs. intensive garden management), the degree of urbanisation (densely urbanised garden sites vs. peripheral areas). More information on the garden selection can be found *et al.*Tresch et al. (2018a) and Frey and Moretti (2019). This figure has been produced using the R package 'ggmap' (Kahle and Wickham 2013).

Figure S 5.2 – Example of allotment and domestic gardens in the city of Zurich. Within each garden two sampling plots (2 m x 2 m) with different garden land-use management were selected. Each of this sampling plots were later associated with one of the following garden land-use types: annual vegetable beds (vegetables; N= 47), perennial flowers and berries (flowers & berries; N= 52) or perennial lawn and meadows (grass; N= 71). This garden land-use types, rather than the two garden types, have been shown to contain the major differences in soil quality (Tresch et al. 2018a) and soil function decomposition (Tresch et al. 2019).

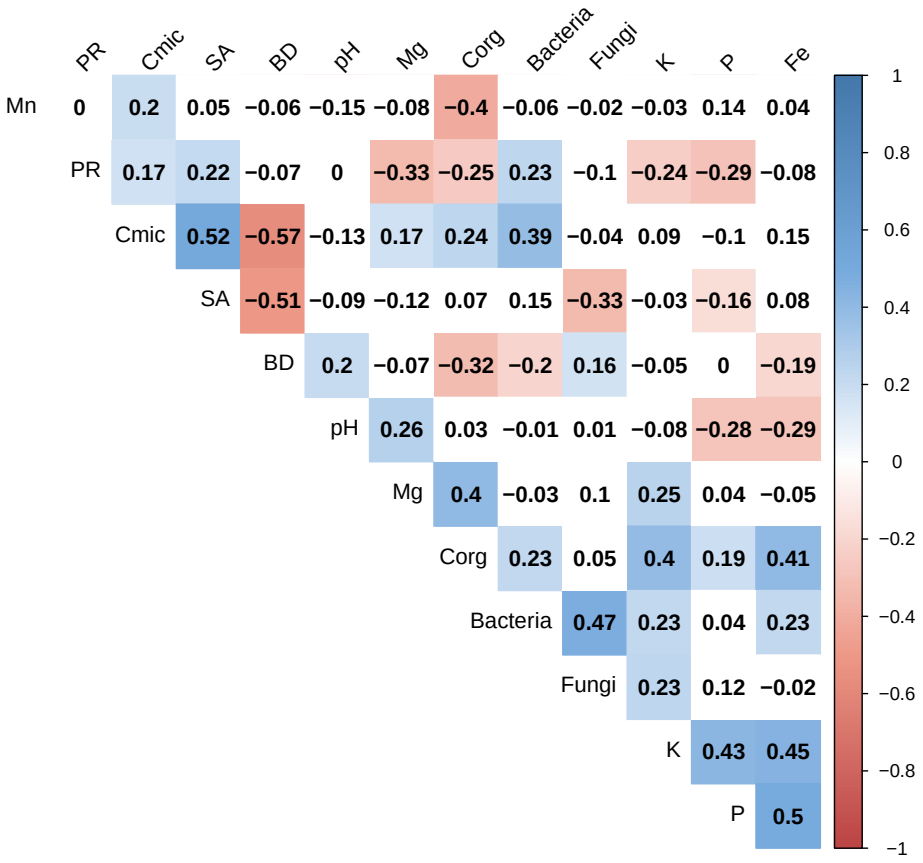

Figure S 5.3 – Pearson correlation matrix of selected soil characteristics based on the soil quality assessment of Tresch et al. (2018a). Only measurements with a very high goodness of fit statistic ($p<0.001$; see Table S 5.1 Tresch et al. (2018a)) for the NMDS ordination, characterising the differences in soil quality between the urban gardens of Zurich have been selected. Additionally, microbial information about gene copy numbers of Bacteria (16S) and Fungi (18S) from Tresch et al. (2018b) has been included in the biological soil characteristics. We dropped Boron because of the high correlation with Potassium ($r=0.63$) and soil basal respiration because of the correlation with C mineralisation ($r=0.98$). The overall variation inflation factor (Borcard et al. 2011) is <2.5. SA: Soil stable aggregates, PR: Penetration resistance, BD: Bulk density, Corg: Organic carbon, Cmic: Microbial biomass carbon.

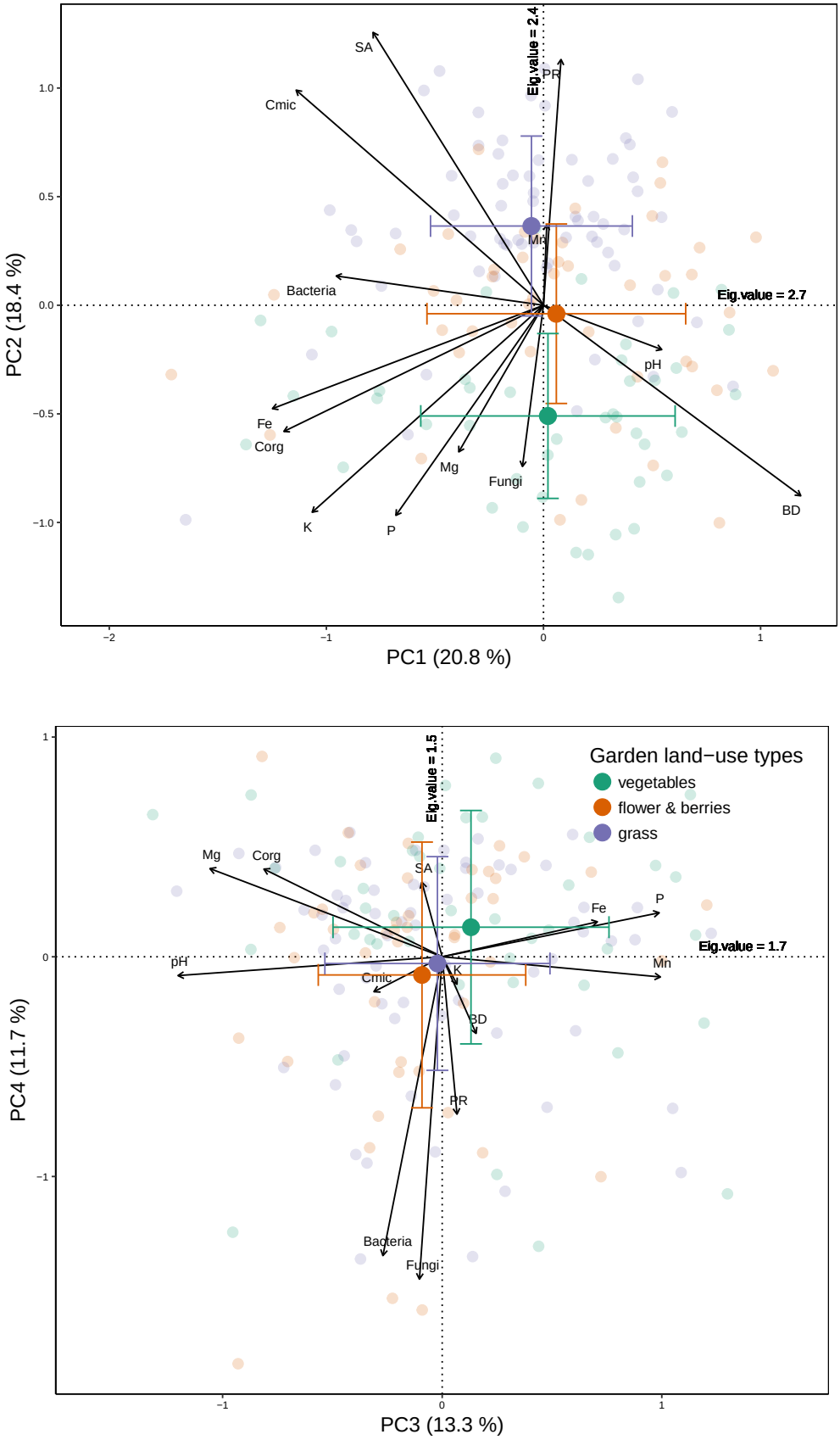

Figure S 5.4 – PCA of soil characteristics. First four axes are needed according to the Kaiser-Guttman criteria (Borcard et al. 2011) explaining 64.2 % of the total variation. SA: Soil stable aggregates, PR: Penetration resistance, BD: Bulk density, Corg: Organic carbon, Cmic: Microbial biomass carbon.

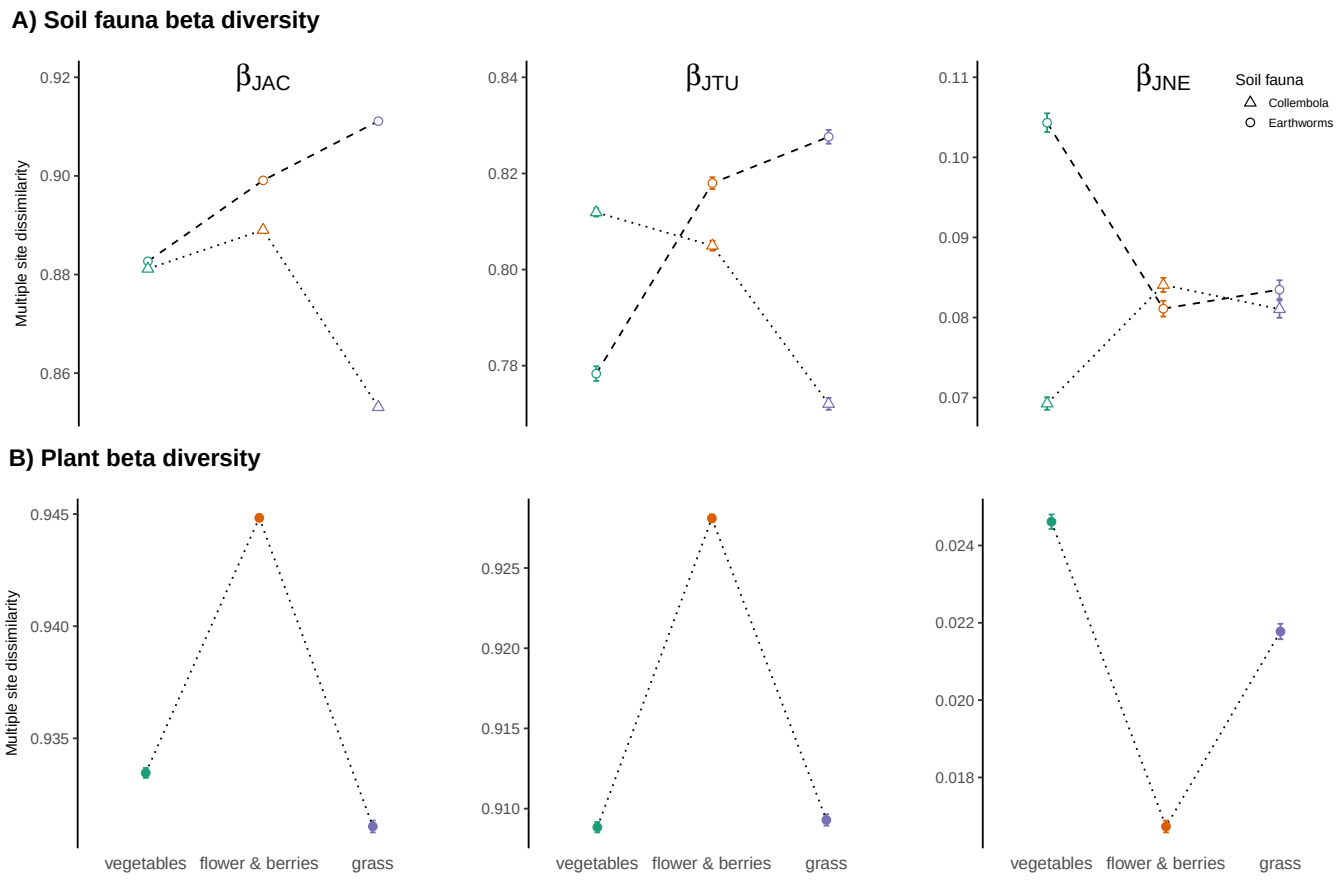

Figure S 5.5 – Soil fauna (A) and plant (B) beta diversity components based on species identity, calculated as mean values with 1000 repetitions of 10 plots following Baselga (2010), Baselga and Orme (2012). β_{JAC}= Total multiple site Jaccard dissimilarity, β_{JTU}= Turnover component, β_{JNE}= Nestedness component. Shapes indicate soil fauna groups and colour the different garden land-use types with error bars as standard errors.

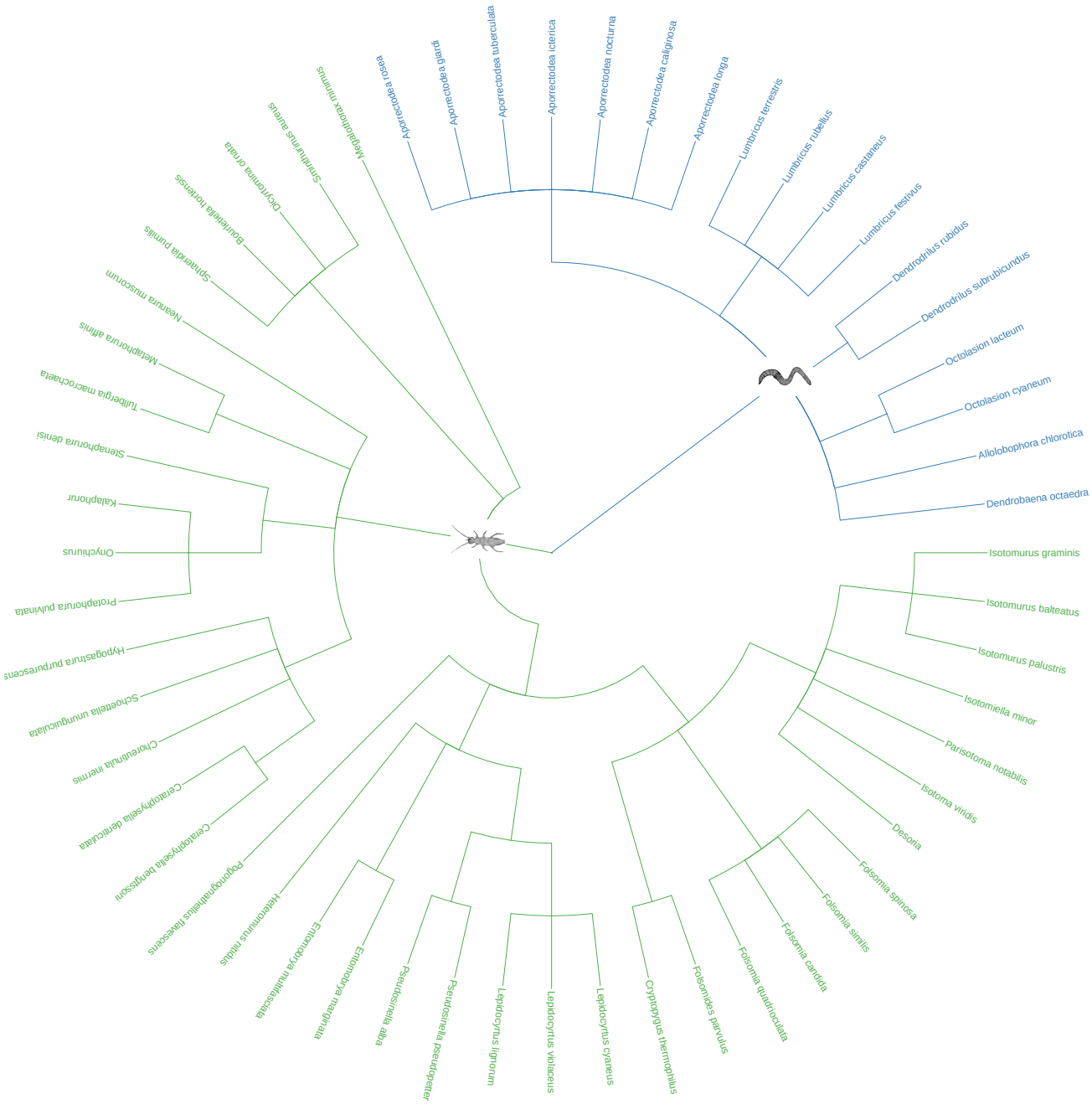

Figure S 5.6 – Phylogenetic tree of 18 earthworm and 39 springtail species with branch lengths calculated according to Paradis et al. (2004) and based on information from the open tree of life project Hinchliff et al. (2015).

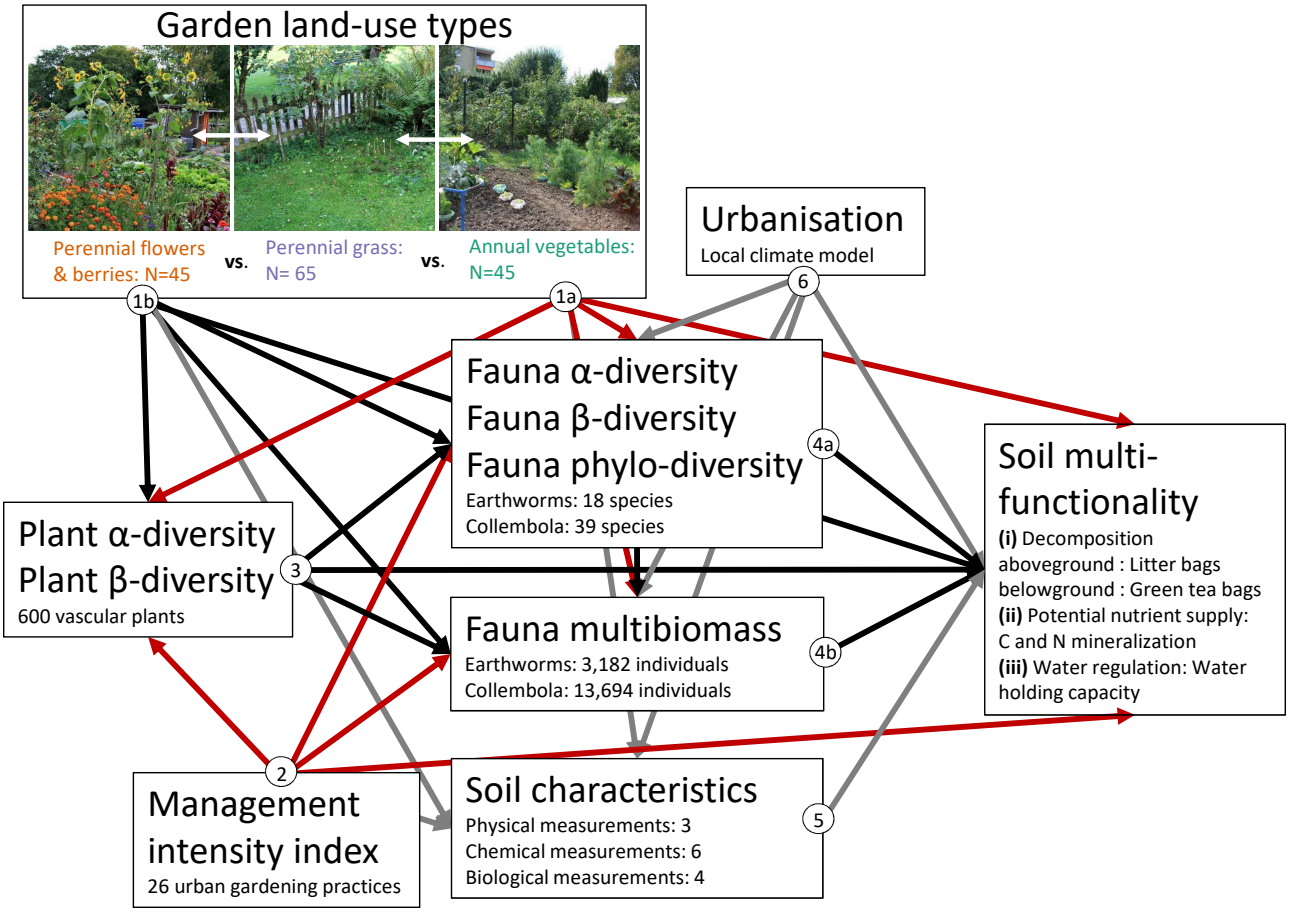

Figure S 5.7 – Alternative a priori SEM model investigating the causal relationships between urban gardening and soil multifunctionality. We expected that (1) different garden land-use types (*vegetables*, *flowers & berries*, *grass*) will have an effect on aboveground and belowground α and β-diversity. More specifically, we hypothesised that *vegetables* will have a negative effect on plant and soil fauna α and β-diversities and on soil multifunctionality compared to the other two garden land-use types. (2) Management intensity will negatively affect plant and soil fauna α and β-diversities and soil multifunctionality. (3) Higher plant α and β-diversity will increase soil fauna α and β-diversity and soil multifunctionality. (4) Soil fauna diversity aspects will positively influence soil multifunctionality. Soil characteristics, being affected by management and land-use types and urbanisation will have a direct effect on soil multifunctionality, depending on the measurements. (5) Urbanisation will have an effect on soil fauna and soil multifunctionality. Expected positive relationships are given in black and negative ones in red, grey arrows represent both positive and negative effects.

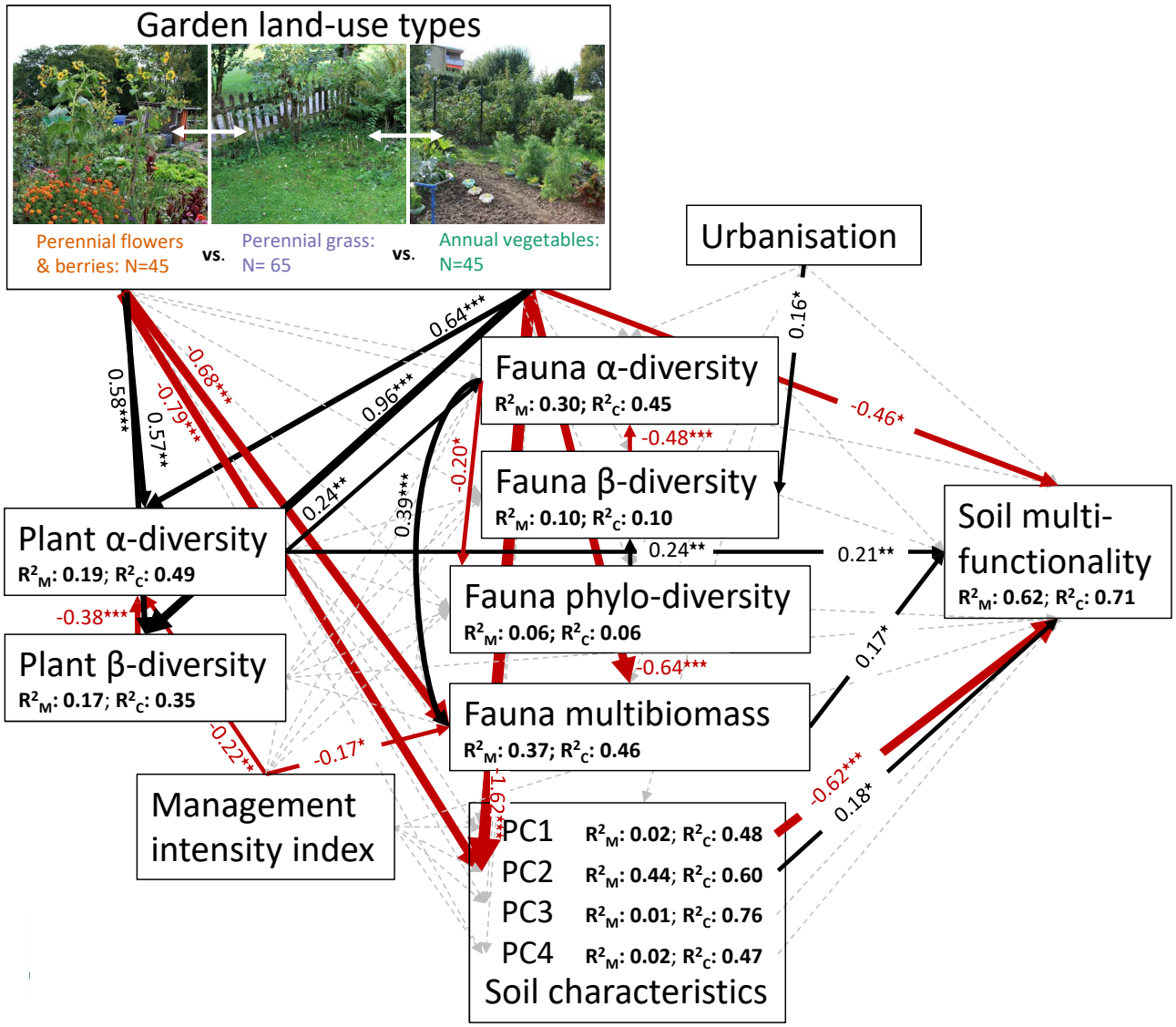

Figure S 5.8 – Alternative SEM including plant and soil fauna β-diversity and fauna phylogenetic diversity (AICc=876.8, Fisher's C=45.6, P=0.96). Arrows represent unidirectional relationships among variables. Black arrows denote significantly ($p<0.05$) positive and red arrows significantly negative relationships (Table S 5.7). Dashed grey arrows represent non significant relationships ($p>0.05$). The thickness of paths has been scaled based on the magnitude of the standardised regression coefficient. Conditional R^2's, based on the variance of both the fixed and random effects, as well as marginal R^2's for component models are given in the boxes of response variables. Soil multifunctionality consists of five measurements related to important soil functions.

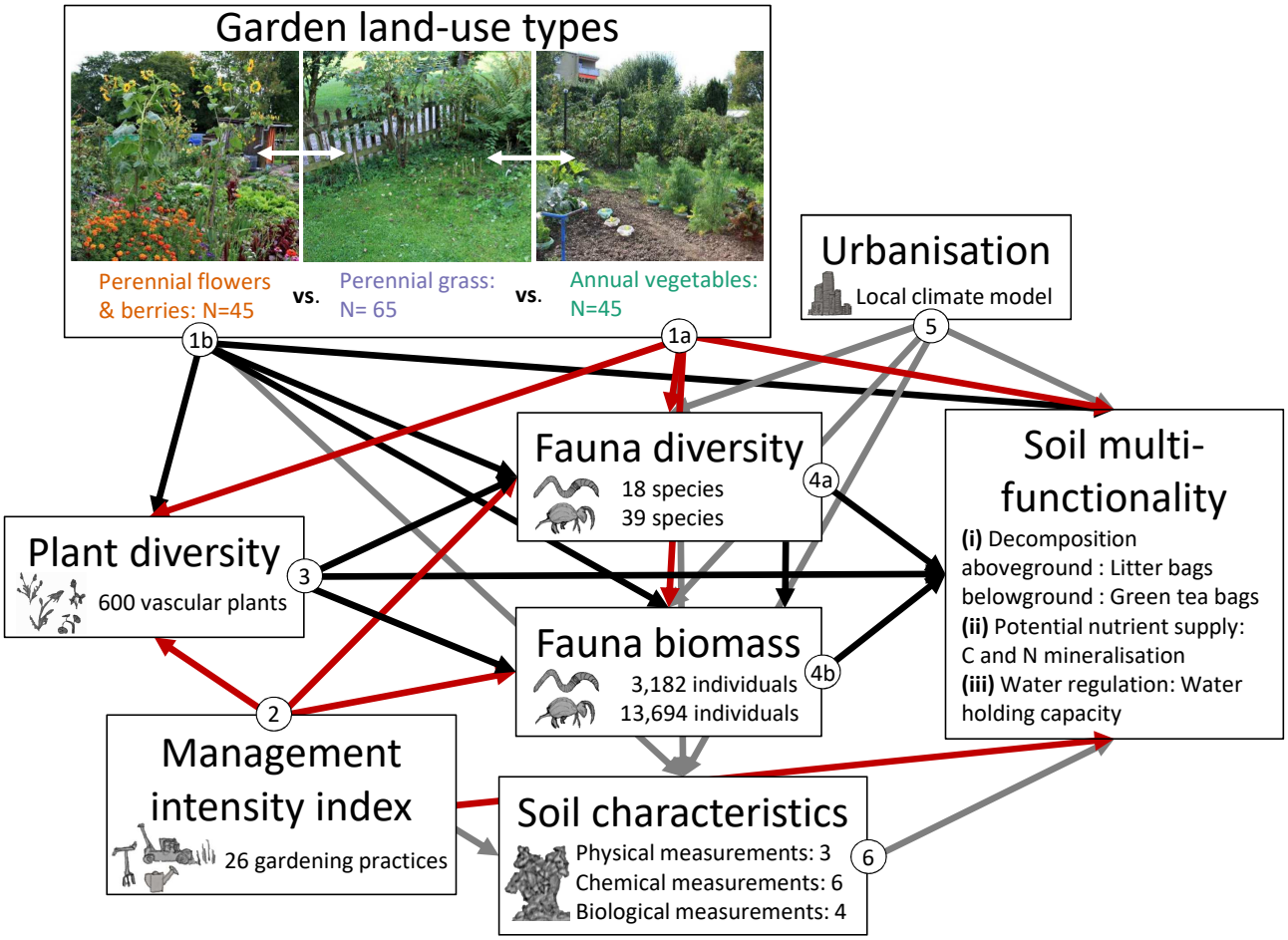

Figure S 5.9 – A priori SEM model with hypothesised direct and indirect effects of urban gardening on soil multifunctionality, including soil characteristics (cf. Figure 5.1). Expected positive relationships are given in black and negative ones in red, grey arrows represent both positive and negative effects. We expected that soil management will negatively affect plant and soil fauna diversity as well as soil multifunctionality (arrows 1 & 2). We hypothesised that higher plant diversity will have a positive effect on soil fauna and soil multifunctionality (arrows 3). We expected a positive effect of soil fauna diversity and biomass on soil multifunctionality (arrows 4). Urbanisation and soil characteristics (arrows 5 & 6) might have a positive or negative effect on soil fauna and soil multifunctionality.

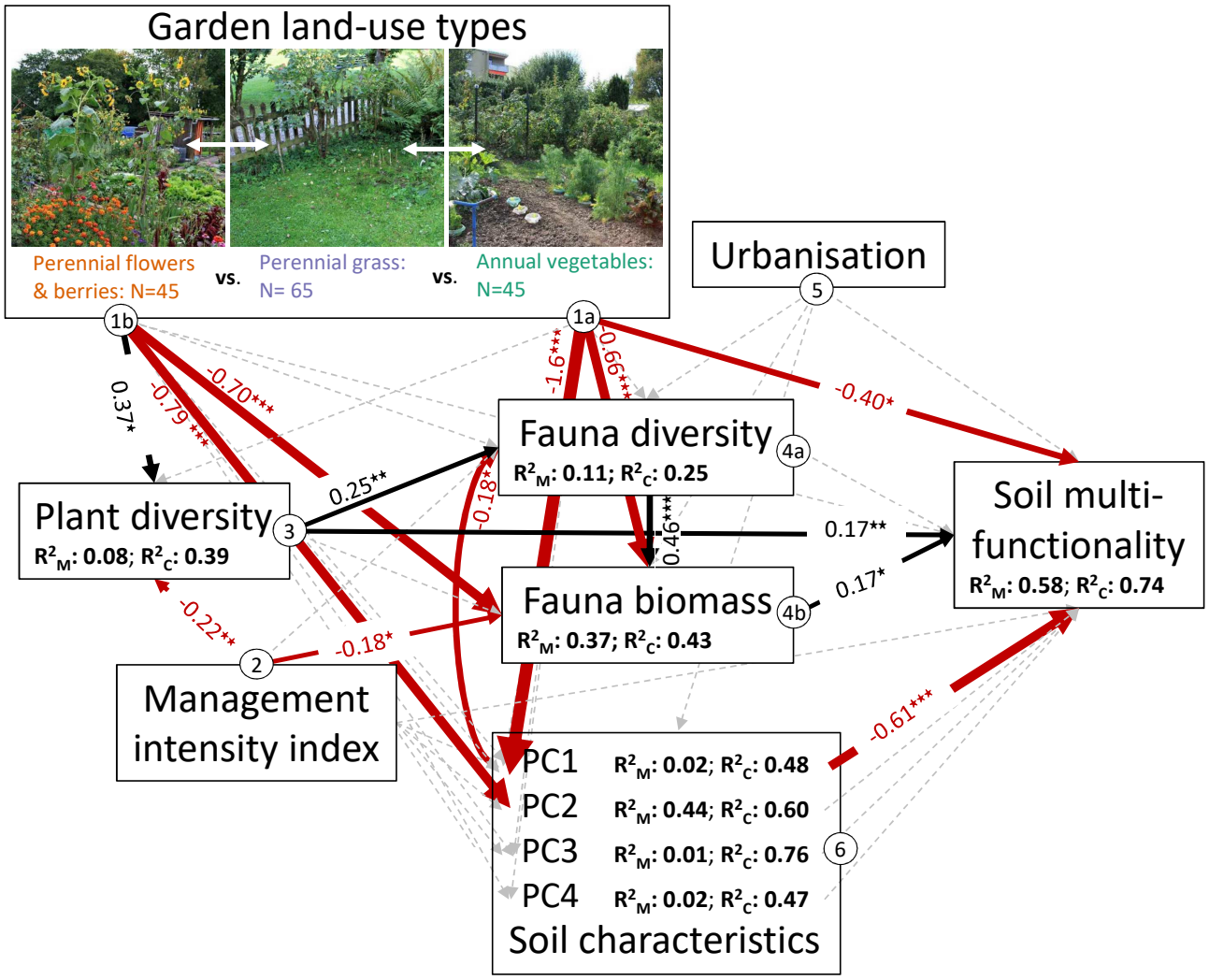

Figure S 5.10 – Final most parsimonious SEM (AICc=288.3, Fisher's C=24.3, P=0.93), including soil characteristics (cf. Figure 5.2). Arrows represent unidirectional relationships among variables. Black arrows denote significantly ($p<0.05$) positive and red arrows significantly negative relationships (Table 5.3). Dashed grey arrows represent non significant relationships ($p>0.05$). The thickness of paths has been scaled based on the magnitude of the standardised regression coefficient. Conditional R^2s, based on the variance of both the fixed and random effects, as well as marginal R^2s, based on the fixed effect parts for each component models are given in the boxes of the response variables. Soil multifunctionality consists of five measurements related to important soil functions.

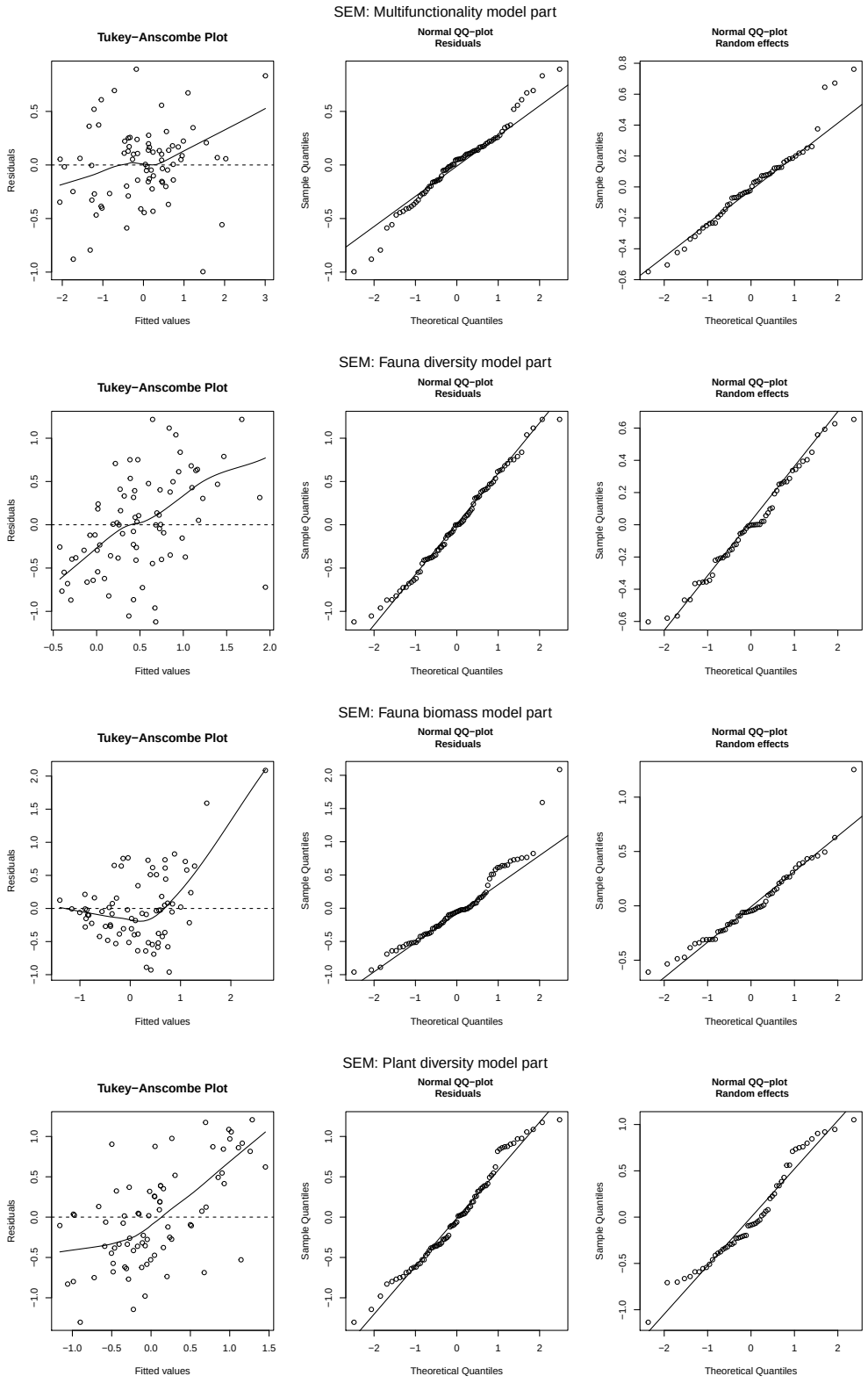

Figure S 5.11 – Residual plots for assessing model assumptions of the LMEM used in the SEM (Figure 5.2). See Table 5.3 for the complete SEM compositions. Residuals have to be independent and identically distributed, hence they should scatter around zero in the Tukey-Anscombe plots (Korner-Nievergelt et al. 2015). A few measurements do not fit well to the model as recognisable in the QQ-plots of the residuals, however the majority of the observations seem to fulfil the model assumptions well and since we did not assume a non-linear effect of the assessed variables with the response variables, we accepted the slight contradiction of model assumptions.

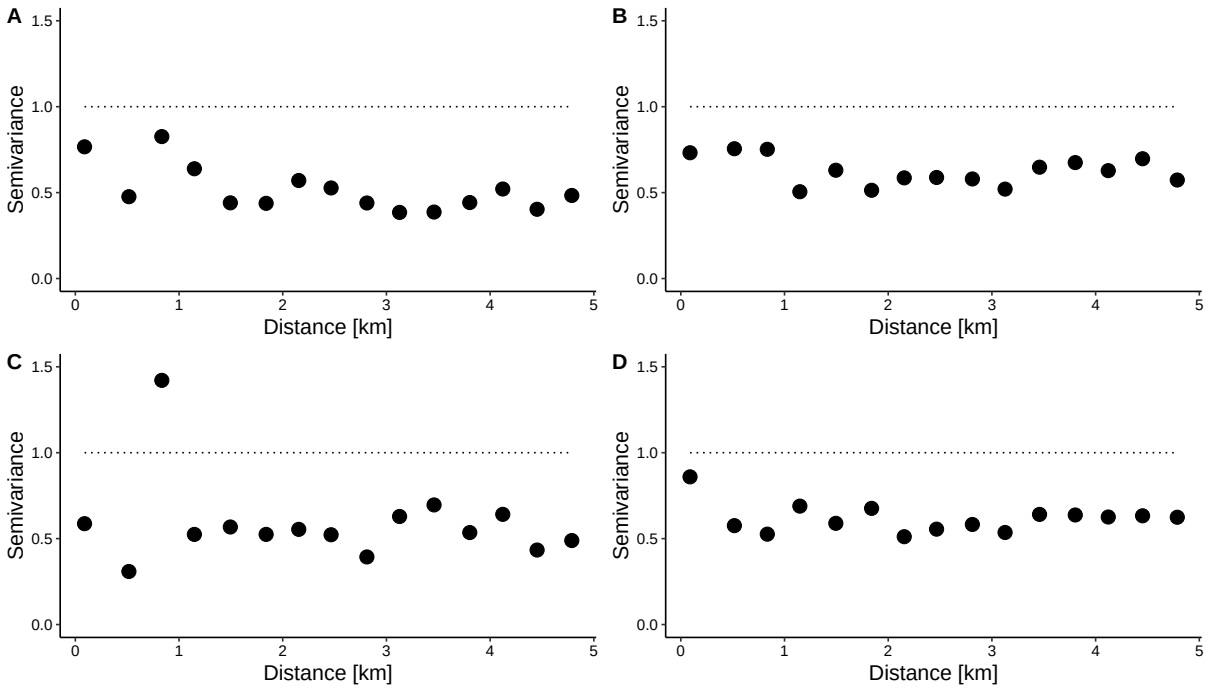

Figure S 5.12 – Semivariograms of LMEM residuals from submodels of the SEM (Figure 5.2): **A)** corresponds to the model part soil multifunctionality, **B)** to the model part fauna diversity, **C)** to the model part fauna biomass, and **D)** to the model part plant diversity. Semivariances (0.5 times the mean squared differences between sites) were computed with the R package 'gstat' (Pebesma 2004). In all plots values are close to 1 and show no clear patterns of spatial autocorrelation, indicating that the residuals are not more similar or dissimilar to each other than expected by chance (Korner-Nievergelt et al. 2015). Moreover, the calculated Moran's I autocorrelation index (Popescu et al. 2012, Paradis 2018) was not significant for all submodels of the SEM: **A)** p=0.56; observed=-0.01±0.008, expected=-0.013; **B)** p=0.46; observed=-0.02±0.008, expected=-0.013; **C)** p=0.86; observed=-0.01±0.007, expected=-0.013; **D)** p=0.61; observed=-0.01±0.008, expected=-0.013.

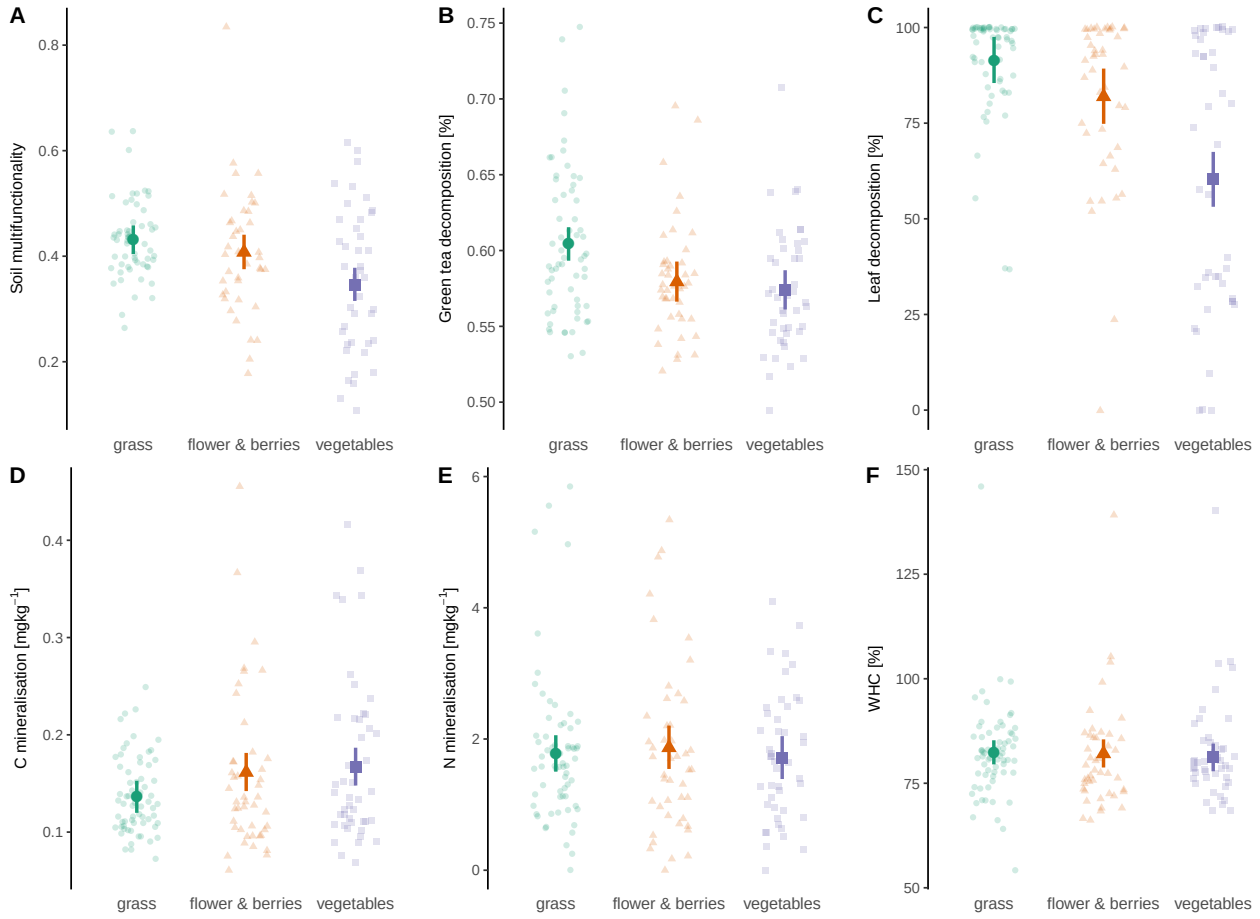

Figure S 5.13 – Soil multifunctionality **(A)** and its single components: belowground decomposition of green tea bags **(B)**, aboveground decomposition of leaf litter **(C)**, C mineralisation **(D)**, N mineralisation **(E)** and water holding capacity **(F)**, as a function of garden land-use types. Bold points represent mean values of the simulated Bayesian inference posterior distribution (Korner-Nievergelt et al. 2015) of the LMEM with garden ID as random factor and garden land-use types as fixed effects. Lines indicate 95 % credible intervals. Estimated LMEM coefficients of fixed effects can be found in Table S 5.5.

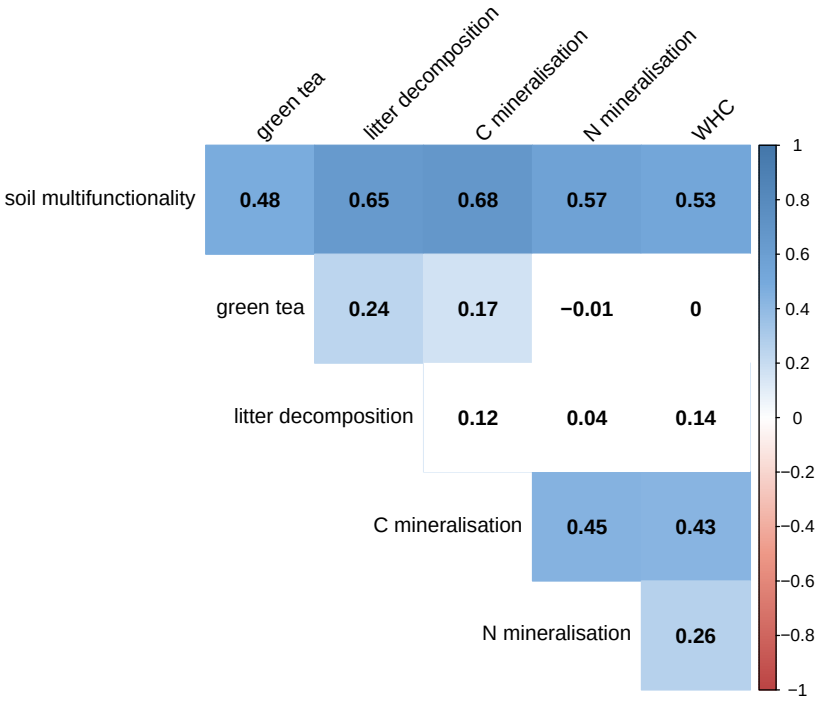

Figure S 5.14 – Pearson correlation matrix of soil multifunctionality and its components. WHC: Water holding capacity.

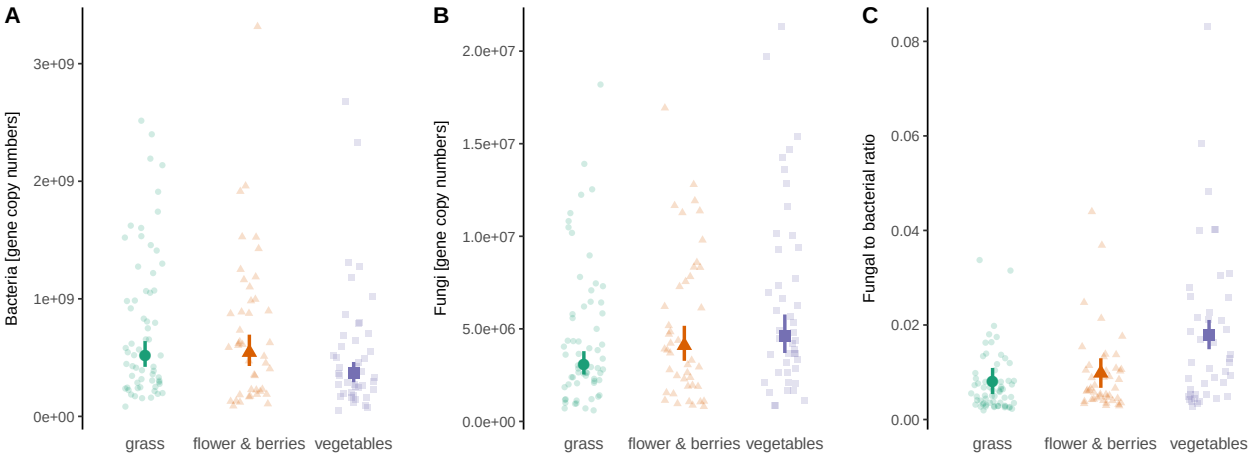

Figure S 5.15 – LMEM of bacterial gene copy numbers **(A)**, fungal gene copy numbers **(B)** and the fungal to bacterial ratio **(C)**, as a function of garden land-use types. Bold points represent mean values of the simulated Bayesian inference posterior distribution (Korner-Nievergelt et al. 2015) of the LMEM with garden ID as random factor and garden land-use types as fixed effects. Lines indicate 95 % credible intervals. Estimated LMEM coefficients of fixed effects can be found in Table S 5.6.

5.5.2 Supplementary Tables

Table S 5.1 – Soil fauna species identity and abundance of 39 springtail and 18 earthworm species characterising soil meso- and macrofauna species. Springtails were identified to species level (Gisin 1960, Zimdars and Dunger 1994, Fjellberg 1998; 2007, Bretfeld 1999), while earthworm species identification was done according to Bouché (1977) and Sims and Gerard (1999). All earthworm species sampled are already known in Switzerland. The springtail species marked with asterisks, were not listed in Fauna Europaea (Deharveng 2018). The two springtail species with two asterisks are new for Switzerland according to the literature (Handschin 1924, Gisin 1943; 1946; 1948; 1957; 1960, Dunger et al. 2004, Schulz 2007, Troxler 1990, Rusterholz et al. 2014) and expert opinions (Deharveng Luis, Cortet Jérôme, Heiniger Charlène, personal communications, 2018). Earthworm life forms as defined by Bouché (1977) and springtail life forms according to their ecological and functional traits as defined by Gisin (Gisin 1943).

Phylum	Class	Order	Family	Genus	Species	Author	Total	Vegetables	Flowers & Berries	Lawn	Life forms
Springtail species							13435	2078	2337	9020	
Arthropoda	Collembola	Entomobryomorpha	Isotomidae	*Parisotoma*	*notabilis*	Schaeffer, 1896	3075	292	158	2625	Hemiedaphic
		Poduromorpha	Onychiuridae	*Protaphorura*	*pulvinata*	Gisin, 1954	1810	317	285	1208	Euedaphic
		Symphypleona	Katiannidae	*Sminthurinus*	*aureus*	Lubbock, 1862	1465	130	147	1188	Hemiedaphic
		Entomobryomorpha	Isotomidae	*Folsomia*	*quadrioculata*	Tullberg, 1871	1117	121	351	645	Euedaphic
		Entomobryomorpha	Isotomidae	*Isotomiella*	*minor*	Schaeffer, 1896	817	75	193	549	Euedaphic
		Symphypleona	Bourletiellidae	*Bourletiella*	*hortensis*	Fitch, 1863	801	605	166	30	Epedaphic
		Entomobryomorpha	Isotomidae	*Isotoma*	*viridis*	Bourlet, 1839	685	49	47	589	Epedaphic
		Entomobryomorpha	Entomobryidae	*Pseudosinella*	*petterseni*	Börner, 1901	490	69	55	366	Euedaphic
		Poduromorpha	Hypogastruridae	*Schoettella*	*ununguiculata* *	Tullberg, 1869	423	2	0	421	Hemiedaphic
		Poduromorpha	Hypogastruridae	*Ceratophysella*	*denticulata*	Bagnall, 1941	408	11	303	94	Hemiedaphic
		Entomobryomorpha	Entomobryidae	*Lepidocyrtus*	*lignorum*	Fabricius, 1793	359	48	104	207	Epedaphic
		Entomobryomorpha	Isotomidae	*Folsomia*	*similis*	Bagnall, 1939	343	36	54	253	Euedaphic
		Entomobryomorpha	Entomobryidae	*Pseudosinella*	*alba*	Packard, 1873	303	28	71	204	Euedaphic
		Entomobryomorpha	Entomobryidae	*Cryptopygus*	*thermophilus*	Axelson, 1900	268	152	104	12	Hemiedaphic
		Entomobryomorpha	Isotomidae	*Isotomurus*	*balteatus* **	Reuter, 1876	129	4	24	101	Epedaphic
		Entomobryomorpha	Entomobryidae	*Lepidocyrtus*	*violaceus*	Geoffroy, 1762	121	2	4	115	Epedaphic
		Entomobryomorpha	Isotomidae	*Folsomides*	*parvulus*	Stach, 1920	108	25	34	49	Euedaphic
		Poduromorpha	Tullbergiidae	*Stenaphorura*	*denisi* *	Bagnall, 1935	105	1	33	71	Euedaphic
		Poduromorpha	Tullbergiidae	*Metaphorura*	*affinis*	Börner, 1903	103	10	12	81	Euedaphic
		Entomobryomorpha	Isotomidae	*Folsomia*	*spinosa*	Kseneman, 1936	101	24	55	22	Euedaphic
		Entomobryomorpha	Isotomidae	*Desoria*	*violacea*	Tullberg, 1876	96	4	38	54	Epedaphic
		Entomobryomorpha	Orchesellinae	*Heteromurus*	*nitidus*	Templeton, 1835	93	24	37	32	Hemiedaphic
		Entomobryomorpha	Tomoceridae	*Pogonognathellus*	*flavescens*	Tullberg, 1871	90	6	22	62	Epedaphic
		Poduromorpha	Tullbergiidae	*Mesaphorura*	*macrochaeta*	Rusek, 1976	87	38	25	24	Euedaphic
		Symphypleona	Sminthurididae	*Sphaeridia*	*pumilis*	Krausbauer, 1898	9	0	1	8	Hemiedaphic
		Entomobryomorpha	Entomobryidae	*Lepidocyrtus*	*cyaneus*	Tullberg, 1871	6	0	3	3	Epedaphic
		Entomobryomorpha	Entomobryidae	*Entomobrya*	*multifasciata*	Tullberg, 1871	4	1	3	0	Epedaphic
		Entomobryomorpha	Isotomidae	*Folsomia*	*candida*	Willem, 1902	3	3	0	0	Hemiedaphic
		Poduromorpha	Onychiuridae	*Kalaphorura*	*burmeisteri* *	Lubock, 1873	3	0	3	0	Euedaphic
		Poduromorpha	Hypogastruridae	*Choreutinula*	*inermis*	Tullberg, 1871	2	1	1	0	Hemiedaphic
		Entomobryomorpha	Isotomidae	*Isotomurus*	*palustris*	Muller, 1776	2	0	1	1	Epedaphic
		Neelipleona	Neelidae	*Megalothorax*	*minimus* *	Willem, 1900	2	0	1	1	Euedaphic
		Poduromorpha	Hypogastruridae	*Ceratophysella*	*bengtssoni*	Agren, 1904	1	0	0	1	Hemiedaphic
		Symphypleona	Dicyrtomidae	*Dicyrtomina*	*ornata*	Nicolet, 1842	1	0	0	1	Epedaphic
		Entomobryomorpha	Entomobryidae	*Entomobrya*	*marginata*	Tullberg, 1871	1	0	0	1	Epedaphic
		Poduromorpha	Hypogastruridae	*Hypogastrura*	*purpurescens* *	Lubock, 1967	1	0	0	1	Hemiedaphic
		Entomobryomorpha	Isotomidae	*Isotomurus*	*graminis* **	Fjellberg, 2007	1	0	1	0	Epedaphic
		Poduromorpha	Neanuridae	*Neanura*	*muscorum*	Templeton, 1835	1	0	0	1	Hemiedaphic
		Poduromorpha	Onychiuridae	*Onychiuroides*	*granulosus* *	Stach, 1930	1	0	1	0	Euedaphic
Earthworm species							3169	1253	769	1147	
Annelida	Oligochaeta	-	-	*Endogeic*	*juvenile*	-	1354	567	315	472	Endogeic
		-	-	*Anecic*	*juvenile*	-	609	155	164	290	Anecic
		Opisthopora	Lumbricidae	*Allolobophora*	*chlorotica*	Savigny, 1826	437	282	55	100	Endogeic
		Opisthopora	Lumbricidae	*Aporrectodea*	*caliginosa*	Savigny, 1826	173	78	51	44	Endogeic
		Opisthopora	Lumbricidae	*Lumbricus*	*terrestris*	Linnaeus, 1758	139	24	42	73	Anecic
		Opisthopora	Lumbricidae	*Aporrectodea*	*rosea*	Savigny, 1826	126	36	27	63	Endogeic
		Opisthopora	Lumbricidae	*Aporrectodea*	*longa*	Ude, 1885	90	31	37	22	Anecic
		Opisthopora	Lumbricidae	*Octolasion*	*lacteum*	Örley, 1885	59	33	13	13	Endogeic
		Opisthopora	Lumbricidae	*Aporrectodea*	*nocturna*	Evans, 1946	59	19	19	21	Anecic
		Opisthopora	Lumbricidae	*Allolobophora*	*icterica*	Savigny, 1826	44	5	15	24	Endogeic
		Opisthopora	Lumbricidae	*Aporrectodea*	*tuberculata*	Eisen, 1875	20	8	6	6	Endogeic
		Opisthopora	Lumbricidae	*Aporrectodea*	*ripicola*	Bouché, 1972	20	6	11	3	Anecic
		-	-	Epigeic	juvenile	-	14	6	2	6	Epigeic
		Opisthopora	Lumbricidae	*Dendrobaena*	*octaedra*	Savigny, 1826	6	1	1	4	Epigeic
		Opisthopora	Lumbricidae	*Dendrodrilus*	*subrubicundus*	Eisen, 1874	6	1	3	2	Epigeic
		Opisthopora	Lumbricidae	*Lumbricus*	*castaneus*	Savigny, 1826	4	1	3	0	Epigeic
		Opisthopora	Lumbricidae	*Dendrodrilus*	*rubidus*	Savigny, 1826	3	0	3	0	Epigeic
		Opisthopora	Lumbricidae	*Aporrectodea*	*giardi*	Ribaucourt, 1901	2	0	2	0	Anecic
		Opisthopora	Lumbricidae	*Lumbricus*	*festivus*	Savigny, 1826	2	0	0	2	Epigeic
		Opisthopora	Lumbricidae	*Lumbricus*	*rubellus*	Hoffmeister, 1843	1	0	0	1	Epigeic
		Opisthopora	Lumbricidae	*Octolasion*	*cyaneum*	Savigny, 1826	1	0	0	1	Endogeic

Table S 5.2 – Diversity indices of plants and soil fauna. Descriptive statistics (median value ± standard error (SE) and coefficient of variation (CV)) and LMEM fixed effects including means and 95% credible intervals (Mean value (2.5 %; 97.5 %)) of the simulated Bayesian inference posterior distribution (Korner-Nievergelt et al. 2015). Bold numbers indicate significant effects between two study garden land-use types, with credible intervals not crossing zero. Note that for the calculation of earthworm α-diversity indices only adult species were chosen, due to the dominance of anecic and endogeic juveniles (Table S 5.1).

	Median values ± SE								Fixed effect coefficients 50% (2.5%;97.5%)		
	All sites	CV	Vegetables	CV	Flowers & berries	CV	Grass	CV	Flowers & berries vs. vegetables	Grass vs. vegetables	Grass vs. flowers &berries
Earthworms α-diversity											
$D_{Simpson}$	2.0±0.1	52	1.85±0.2	49	2.57±0.2	48	2.0±0.2	55	**0.8 (0.12;1.9)**	0.05 (-0.31;0.61)	-0.41 (-0.62;-0.1)
$D_{Simpson}$ Anecic	1.0±0.05	42	1±0.1	45	1.6±0.09	40	1±0.07	41	0.2 (-0.08;0.57)	-0.06 (-0.27;0.21)	-0.22 (-0.39;-0.01)
$D_{Simpson}$ Endogeic	1.47±0.06	45	1.47±0.1	44	1.8±0.11	44	1.19±0.1	48	0.17 (-0.15;0.61)	0.04 (-0.22;0.39)	-0.11 (-0.35;0.21)
$D_{Simpson}$ Epigeic	1.0±0.02	24	1.5±0.11	47	1±0	0	1±0	0	-	-	-
$E_{Simpson}$	0.9±0.01	22	0.735±0.03	29	0.893±0.02	17	0.9±0.02	17	**0.12 (0.05;0.2)**	**0.17 (0.1;0.24)**	0.04 (-0.02;0.1)
$E_{Simpson}$ Anecic	1±0.01	8	1±0.01	9	1±0.01	8	1±0.01	8	-0.17 (-0.37;0.08)	**-0.22 (-0.39;-0.01)**	0.01 (-0.02;0.05)
$E_{Simpson}$ Endogeic	1±0.01	20	0.9±0.03	26	1±0.02	15	1±0.02	18	**0.09 (0.01;0.16)**	**0.09 (0.02;0.16)**	0.01 (-0.06;0.07)
$E_{Simpson}$ Epigeic	-	-	-	-	-	-	-	-	-	-	-
Springtail α-diversity											
$D_{Simpson}$	3.6±0.1	35	3.3±0.2	37	3.4±0.2	42	3.8±0.1	28	0.38 (-0.15;0.93)	0.43 (-0.08;0.92)	0.04 (-0.45;0.53)
$D_{Simpson}$ Epedaphic	1.1±0.05	42	1.0±0.04	25	1.0±0.08	42	1.43±0.08	42	0.14 (-0.11;0.41)	**0.48 (0.24;0.72)**	**0.33 (0.09;0.58)**
$D_{Simpson}$ Hemiedaphic	1.5±0.04	33	1.5±0.09	36	1.2±0.08	39	1.543±0.05	27	0.47 (-0.14;1.54)	0.54 (-0.07;1.52)	0.05 (-0.36;0.69)
$D_{Simpson}$ Euedaphic	2.3±0.09	44	2.0±0.18	52	2.7±0.16	40	2.244±0.12	40	0.47 (-0.07;1.32)	0.12 (-0.26;0.69)	-0.24 (-0.49;0.15)
$E_{Simpson}$	0.52±0.01	33	0.51±0.02	30	0.59±0.03	35	0.49±0.02	30	**0.07 (0.01;0.15)**	-0.04 (-0.11;0.03)	**-0.11 (-0.18;-0.05)**
$E_{Simpson}$ Epedaphic	0.9±0.02	24	1.0±0.03	23	1.0±0.03	22	0.83±0.03	26	-0.01 (-0.08;0.08)	-0.06 (-0.13;0.01)	-0.06 (-0.13;0.02)
$E_{Simpson}$ Hemiedaphic	0.83±0.02	25	0.86±0.03	22	0.96±0.03	23	0.762±0.02	27	0.03 (-0.06;0.12)	-0.07 (-0.14;0.01)	**-0.09 (-0.16;-0.02)**
$E_{Simpson}$ Euedaphic	0.69±0.02	29	0.79±0.03	25	0.76±0.03	30	0.601±0.02	27	-0.06 (-0.13;0.02)	**-0.14 (-0.2;-0.08)**	**-0.09 (-0.15;-0.02)**
Plants α-diversity											
$D_{Simpson}$ Plants	25±0.76	36	26±1.58	38	28±1.47	34	23±1.01	33	1.69 (-1.64;5.15)	-2.94 (-6;0.02)	**-4.66 (-7.76;-1.66)**
Soil fauna disturbance indices											
Collembolan ecomorphological index	5.8±0.1	30	4.4±0.3	39	6.2±0.2	26	5.7±0.2	24	**1.39 (0.76;2.05)**	**0.85 (0.26;1.43)**	-0.55 (-1.13;0.02)
Acari to collembola ratio	1.3±0.4	236	1.2±0.2	80	1.5±1.2	247	0.99±0.4	189	0.19 (-0.03;0.47)	-0.05 (-0.21;0.14)	**-0.73 (-0.92;-0.07)**
Fungal to bacterial ratio	0.007±0.0009	103	0.012±0.003	93	0.007±0.0001	87	0.006±0.0001	82	**-0.008 (-0.012;-0.004)**	**-0.01 (-0.013;-0.007)**	-0.002 (-0.005;0.001)
Earthworm anecic to endogeic ratio	0.58±0.1	145	0.26±0.05	88	0.67±0.2	115	0.69±0.2	143	**0.34 (0.13;0.59)**	**0.40 (0.19;0.64)**	0.05 (-0.10;0.22)
Soil fauna biomass											
Fauna biomass	0.16±0.01	57	0.13±0.01	56	0.12±0.02	70	0.20±0.01	46	0.01 (-0.03;0.05)	**0.07 (0.03;0.1)**	**0.06 (0.02;0.09)**
Earthworm biomass [gm^2]	99.4±6.8	73	97.8±13.6	73	112.2±14.9	80	95.0±8.5	64	0.14 (-0.17;0.59)	-0.02 (-0.27;0.31)	-0.14 (-0.36;0.14)
Springtail biomass [gm^2]	23.98±2.7	95	15.7±1.9	64	18.2±3.4	107	46.7±4.7	71	-0.08 (-0.35;0.3)	**1.42 (0.75;2.32)**	**1.63 (0.91;2.6)**
Microbial biomass (C_{mic}) [$mgkg^{-1}$]	780.85±21.3	33	639.76±36.2	35	772.57±37.0	32	821.63±32.2	29	**106.39 (13.5;202.56)**	**204.83 (119.1;287.83)**	**97.78 (11.35;181.47)**

Table S 5.3 – Management questions asked of all 85 participating urban gardeners of this study. Management intensity index was calculated as a scaled sum (divided by the number of questions) of all 26 garden management questions on a five level Likert scale. For *grass* sites we considered nine questions: MowGrass, FstCutGrass, FertGrass, WaterGrass, CareGrass, PestGrass, FlowerIslands, Weeds, Leaves. For *flower & berry* sites ten questions: FertForbs, WaterForbs, PestForbs, DiggingForbs, ForkForbs, CutTrees, PestTrees, Leaves, DrySticks, Weeds and for *vegetables* sites eleven questions: FertCrops, WaterCrops, PestCrops, CropRotate, MixCult, Mulch, GreenFert, DiggingCrops, ForkCrops, DrySticks, Weeds. Furthermore, the following five individual management practices were used: (i) Disturbance: combined management answers (yes/no) of major soil disturbances (”DiggingVeg”, ”DiggingFlower”, ”CareGrass”); (ii) fertiliser: ("FertGrass", "FertVeg", "FertFlower"); (iii) Pesticides: (”PestGrass”,”PestVeg”,”PestFlower”,”PestTrees”,”WeedingHerbicides”); (iv) Water: (”WaterGrass”, ”WaterVeg”, ”WaterFlower”); (v) frequency of weeding ("Weeds"). Higher factor levels indicate higher management intensity. Questions were originally asked in German.

PestGrass
How often do you use pesticides, fungicides or herbicides to protect your lawn?
Never (1)
Less than once per year (2)
1 to 3 times per year (3)
4 to 10 times per year (4)
More than 10 times per year (5)

PestFlower
How often do you use pesticides, fungicides or herbicides (without slug pellets) to protect your flowers?
Never (1)
Less than once per year (2)
1 to 3 times per year (3)
4 to 10 times per year (4)
More than 10 times per year (5)

PestVeg
How often do you use pesticides, fungicides or herbicides (without slug pellets) to protect your vegetables?
Never (1)
Less than once per year (2)
1 to 3 times per year (3)
4 to 10 times per year (4)
More than 10 times per year (5)

FertGrass
How often do you use fertilisers for your lawn?
Never (1)
Every 4 to 5 years (2)
Every 2 to 3 years (3)
Once a year (4)
More than once a year (5)

FertVeg
How often do you use fertilisers for your vegetables?
Never (1)
Every 2 to 3 years (2)
Once a year (3)
2 to 3 times per year (4)
More than three times per year (5)

FertFlower
How often do you use fertilisers for your flowers?
Never (1)
Every 2 to 3 years (2)
Once a year (3)
2 to 3 times per year (4)
More than three times per year (5)

Weeds
How often do you remove most of the weeds in your garden?
Never (1)
Rarely (2)
Sometimes (3)
Often (4)
Very often (5)

PestTrees
How often do you use insecticides, fungicides or herbicides to protect your trees and shrubs?
Never (1)
Less than once a year (2)
1 to 3 times per year (3)
4 to 10 times per year (4)
More than 10 times per year (5)

Leaves
How often do you remove most of the leaves in your garden?
Never (1)
Spring (2)
Autumn (3)
Every 2 to 3 weeks (4)
Weekly in autumn (5)

MowGrass
How often do you mow your lawn?
1 to 2 (1)
3 to 4 (2)
5 to 8 (3)
9 to 20 (4)
over 20 (5)

MixCult
Do you follow the principle of mixed cultivation (planting different varieties of vegetables and/or flowers in the same cultivation plot)?
Never (5)
Rarely (4)
Sometimes (3)
Mostly (2)
Always (1)

FlowerIslands
Do you leave islands of flowers when you mow your lawn?
Never (5)
Rarely (4)
Sometimes (3)
Mostly (2)
Always (1)

WaterGrass
How often do you water your lawn?

Never (1)
When dry (2)
once a week (3)
twice a week (4)
More than twice a week (5)

WaterVeg
How often do you water your vegetable beds?
Never (1)
When dry (2)
once a week (3)
twice a week (4)
More than twice a week (5)

WaterFlower
How often do you water your flower beds?
Never (1)
When dry (2)
once a week (3)
twice a week (4)
More than twice a week (5)

CareGrass
How often do you scarify your lawn (including reseeding)
Never (1)
Every 6 to 10 years (2)
Every 4 to 5 years (3)
Every 2 to 3 years (4)
Annually (5)

DiggingForbs
How often do you till your soil in the flower beds?
Never (1)
Every 3 years or less (2)
Every two years (3)
Once per year (4)
More than once per year (5)

DiggingCrops
How often do you till your soil in the vegetable beds?
Never (1)
Every 3 years or less (2)
Every two years (3)
Once per year (4)
More than once per year (5)

DrySticks
Do you leave withered flowers and sticks during the winter in your garden?
Never (5)
Rarely (4)
Sometimes (3)
Mostly (2)
Always (1)

FstCutGrass
When is the first time point of cutting your lawn?

April (5)
May (4)
Start of June (3)
End of June (2)
After June (1)

Mulch
Do you use organic material (mulch) to cover your vegetable beds?

Never (5)
Rarely (4)
Sometimes (3)
Mostly (2)
Always (1)

CropRotate
Do you consider changing flower beds (crop rotation) for the vegetables grown annually?
Never (5)
Rarely (4)
Sometimes (3)
Mostly (2)
Always (1)

GreenFert
Do you grow plants for green manure?

Never (5)
Rarely (4)
Sometimes (3)
Mostly (2)
Always (1)

WeedingHerbicide
Do you use commercial herbicides?

No (0)
Yes (1)

ForkForbs
How often do you loosen your soil with a fork without turning it around (or milling)?
More than once per year (5)
Once per year (4)
Every 2 years or less (3)
Every 3 years or less (2)
Never (1)

ForkCrops
How often do you loosen your soil with a fork without turning it around (or milling)?
More than once per year (5)
Once per year (4)
Every 2 years or less (3)
Every 3 years or less (2)
Never (1)

CutTrees
How often do you cut most of your forbs and trees?

More than once per year (5)
Once a year (4)
Every 2 years (3)
Every 3 to 5 years(2)
Less than every 5 years (1)

Table S 5.4 – NMDS ordination of earthworm and springtail community composition, using goodness of fit measures calculated with 10,000 permutations (envfit{vegan} Oksanen et al. (2017)). Values are ordered according to the pseudo squared correlation coefficient R^2 and printed in bold font if p <0.001.

Earthworm species	R^2	P-value
Allolobophora chlorotica	0.30	< **0.001**
Endogeic juvenile	0.20	< **0.001**
Anecic juvenile	0.19	< **0.001**
Lumbricus terrestris	0.17	< **0.001**
Aporrectodea caliginosa	0.13	< **0.001**
Octolasion lacteum	0.07	0.01
Aporrectodea longa	0.05	0.03
Dendrodrilus rubidus	0.04	0.12
Aporrectodea ripicola	0.03	0.15
Aporrectodea nocturna	0.02	0.17
Lumbricus castaneus	0.02	0.24
Lumbricus festivus	0.02	0.26
Epigeic juvenile	0.02	0.30
Allolobophora icterica	0.01	0.61
Dendrobaena octaedra	0.01	0.43
Dendrodrilus subrubicundus	0.01	0.67
Lumbricus rubellus	0.01	0.49
Octolasion cyaneum	0.01	0.50
Aporrectodea tuberculata	<0.01	0.80
Aporrectodea giardi	<0.01	0.67
Aporrectodea rosea	<0.01	0.83

Springtail species	R^2	P
Bourletiella hortensis	0.29	< **0.001**
Folsomia quadrioculata	0.18	< **0.001**
Parisotoma notabilis	0.14	< **0.001**
Cryptopygus thermophilus	0.13	< **0.001**
Sminthurinus aureus	0.11	< **0.001**
Isotoma viridis	0.10	< **0.001**
Ceratophysella denticulata	0.08	< **0.001**
Pseudosinella alba	0.08	< **0.001**
Metaphorura affinis	0.06	0.01
Pogonognathellus flavescens	0.05	0.02
Schoettella ununguiculata	0.05	0.01
Desoria violacea	0.04	0.05
Entomobrya marginata	0.03	0.09
Entomobrya multifasciata	0.03	0.1
Lepidocyrtus lignorum	0.03	0.09
Lepidocyrtus violaceus	0.03	0.13
Neanura muscorum	0.03	0.08
Heteromurus nitidus	0.02	0.21
Isotomiella minor	0.02	0.19
Pseudosinella petterseni	0.02	0.20
Stenaphorura denisi	0.02	0.22
Choreutinula inermis	0.01	0.66
Folsomia candida	0.01	0.52
Folsomia similis	0.01	0.66
Isotomurus balteatus	0.01	0.62
Kalaphorura burmeisteri	0.01	0.55
Lepidocyrtus cyaneus	0.01	0.48
Megalothorax minimus	0.01	0.34
Mesaphorura macrochaeta	0.01	0.35
Onychiuroides granulosus	0.01	0.55
Protaphorura pulvinata	0.01	0.69
Ceratophysella bengtssoni	<0.01	0.91
Dicyrtomina ornata	<0.01	0.89
Folsomia spinosa	<0.01	0.76
Folsomides parvulus	<0.01	0.81
Hypogastrura purpurescens	<0.01	0.99
Isotomurus graminis	<0.01	0.73
Isotomurus palustris	<0.01	0.95
Sphaeridia pumilis	<0.01	0.77

Table S 5.5 – Indices of β-diversity based on species identity of soil fauna (A) and plant (B) communities. All components of β-diversity were calculated by mean values of 1000 repetitions of 10 plots following Baselga (2010), Baselga and Orme (2012). β_{JAC}= Total multiple site Jaccard dissimilarity, β_{JTU} = Turnover component, β_{JNE}= Nestedness component. Descriptive statistics (median value ± SE) and effects (Mean value (2.5 %; 97.5 %) of the Bayesian posterior distribution (one-way ANOVA) of a linear model with land-use type as fixed effects (Korner-Nievergelt et al. 2015). Bold numbers indicate significant effects with credible intervals not crossing zero. Note that due to the permutation no random effect could be assigned to the linear model. EW= Earthworm, COL=Collembola, PLA = Plant.

	Median values ± SE				Fixed effect coefficients		
	All sites	Vegetables	Flowers & Berries	Grass	Flowers & berries vs. vegetables	Grass vs. vegetables	Grass vs. flowers & berries
A) Soil fauna β-diversity components							
β_{JAC} EW	0.9±0.001	0.883±0.001	0.901±0.001	0.912±0.001	**0.019 (0.018;0.02)**	**0.03 (0.029;0.031)**	**0.011 (0.01;0.012)**
β_{JNE} EW	0.085±0.001	0.099±0.001	0.077±0.001	0.078±0.001	**-0.024 (-0.027;-0.021)**	**-0.022 (-0.025;-0.019)**	**0.002 (-0.001;0.006)**
β_{JTU} EW	0.814±0.001	0.783±0.002	0.824±0.001	0.833±0.001	**0.043 (0.039;0.047)**	**0.052 (0.048;0.055)**	**0.009 (0.005;0.012)**
β_{JAC} COL	0.878±0.001	0.882±0.001	0.889±0.001	0.856±0.001	**0.007 (0.006;0.009)**	**-0.027 (-0.028;-0.026)**	**-0.034 (-0.035;-0.033)**
β_{JNE} COL	0.074±0.001	0.066±0.001	0.081±0.001	0.077±0.001	**0.017 (0.014;0.019)**	**0.014 (0.012;0.017)**	-0.003 (-0.005;0.000)
β_{JTU} COL	0.8±0.001	0.815±0.001	0.805±0.001	0.776±0.001	**-0.01 (-0.012;-0.007)**	**-0.041 (-0.044;-0.038)**	**-0.032 (-0.034;-0.029)**
B) Plant β-diversity components							
β_{JAC} PLA	0.937±0.001	0.934±0.001	0.945±0.001	0.932±0.001	**0.011 (0.011;0.012)**	**-0.002 (-0.003;-0.002)**	**-0.014 (-0.015;-0.013)**
β_{JNE} PLA	0.02±0.001	0.024±0.001	0.016±0.001	0.021±0.001	**-0.008 (-0.008;-0.008)**	**-0.003 (-0.003;-0.002)**	**0.005 (0.005;0.006)**
β_{JTU} PLA	0.916±0.001	0.909±0.001	0.929±0.001	0.910±0.001	**0.019 (0.019;0.02)**	0.001 (-0.001;0.001)	**-0.019 (-0.02;-0.018)**

Table S 5.6 – Strength of indirect and total pathway estimates of the final SEM (Figure 5.2, Table 5.3), calculated by multiplying the standardised coefficients along the path to the response variable and adding the direct pathways (Lefcheck 2016). Note that only significant direct and indirect pathways (P<0.05, SEM Table 5.3) were used.

	Direct pathway estimate	Indirect pathway estimate	**Total estimate**
Predictors of soil multifunctionality			
Plant diversity	0.17	$0.25 * 0.46 * 0.17 = 0.02$	**0.19**
Annual vegetables	-0.40	$-0.66 * 0.17 = -0.11$	**-0.51**
PC1	-0.61	$-0.18 * 0.46 * 0.17 = -0.01$	**-0.62**
Management intensity	n.s.	$(-0.22 * 0.17) + (-0.18 * 0.17) + (-0.22 * 0.25 * 0.46 * 0.17) = -0.07$	**-0.07**
Fauna diversity	n.s	$0.46 * 0.17 = 0.08$	**0.08**
Perennial flowers & berries	n.s	$(0.37 * 0.17) + (0.37 * 0.25 * 0.46 * 0.17) + (-0.7 * 0.17) = -0.05$	**-0.05**
Predictors of other response variables			
Management intensity - Fauna biomass	-0.18	$-0.22 * 0.25 * 0.46 = -0.03$	**-0.21**
Management intensity - Fauna diversity	n.s.	$-0.22 * 0.25 = -0.06$	**-0.06**
PC1 - Fauna biomass	n.s.	$(-0.18 * 0.46) = -0.08$	**-0.08**

Table S 5.7 – Alternative SEM (AICc=876.8, Fisher's C= 45.6, P=0.96) including soil fauna and plant β-diversity as well as soil fauna phylogenetic diversity (phyloDiv) indicating direct and indirect effects on soil multifunctionality from garden land-use types, garden management, plant α and β-diversity, soil fauna α and β-diversity, soil characteristics as well as urbanisation. R^2_M is based on fixed effects and R^2_C on fixed and random (garden ID) effects. Soil multifunctionality consisting of five measurements related to important soil functions.

	R^2_C	R^2_M	predictor	estimate	P	
Soil multifunctionality	0.71	0.62	Soil PC1	-0.62±0.06	<0.001	***
			Plant α-diversity	0.21±0.06	0.0023	**
			Fauna multibiomass	0.17±0.07	0.02	*
			Vegetables	-0.46±0.20	0.02	*
			Soil PC2	0.18±0.08	0.03	*
			Plant β-diversity	0.13±0.07	0.06	
			Soil PC3	0.11±0.06	0.07	
			Urbanisation	0.11±0.07	0.11	
			Fauna phylo-diversity	0.09±0.06	0.15	
			Management intensity	0.09±0.06	0.19	
			Flowers & berries	-0.11±0.20	0.47	
			Fauna β-diversity	0.05±0.07	0.48	
			Fauna α-diversity	0.03±0.07	0.68	
			Soil PC4	-0.01±0.06	0.91	
Fauna α-diversity	0.45	0.30	Fauna β-diversity	-0.48±0.07	<0.001	***
			Plant α-diversity	0.24±0.08	0.0035	**
			Management intensity	0.11±0.08	0.17	
			Flowers & berries	0.09±0.20	0.59	
			Urbanisation	-0.04±0.08	0.61	
			Vegetables	-0.049±0.2	0.79	
			Plant β-diversity	-0.02±0.08	0.81	
Fauna β-diversity	0.10	0.10	Fauna phylo-diversity	0.24±0.08	0.0038	**
			Urbanisation	0.16±0.08	0.046	*
			Vegetables	0.40±0.20	0.07	
			Plant α-diversity	-0.06±0.09	0.48	
			Flowers & berries	0.15±0.20	0.48	
			Plant β-diversity	-0.05±0.09	0.63	
			Management intensity	0.001±0.08	0.99	
Fauna phylo-diversity	0.06	0.06	Fauna α-diversity	-0.20±0.08	0.023	*
			Flowers & berries	0.35±0.20	0.10	
			Plant β-diversity	-0.08±0.10	0.42	
			Vegetables	0.18±0.20	0.42	
			Plant α-diversity	-0.05±0.09	0.62	
			Management intensity	-0.04±0.09	0.65	
			Urbanisation	-0.02±0.08	0.80	
Fauna multibiomass	0.46	0.37	Fauna α-diversity	0.39±0.08	<0.001	***
			Flowers & berries	-0.68±0.20	<0.001	***
			Vegetables	-0.64±0.20	<0.001	***
			Management intensity	-0.17±0.07	0.018	*
			Fauna β-diversity	-0.14±0.08	0.07	
			Plant α-diversity	0.11±0.08	0.17	
			Urbanisation	0.05±0.07	0.50	
			Fauna phylo-diversity	0.03±0.07	0.69	
			Plant β-diversity	0.02±0.08	0.81	
Plant α-diversity	0.49	0.19	Plant β-diversity	-0.38±0.08	<0.001	***
			Vegetables	0.64±0.20	<0.001	***
			Flowers & berries	0.58±0.20	<0.001	***
			Management intensity	-0.22±0.08	0.0061	**
Plant β-diversity	0.35	0.17	Vegetables	0.96±0.20	<0.001	***
			Flowers & berries	0.57±0.20	0.0013	**
			Management intensity	-0.04±0.08	0.64	
Soil PC1	0.48	0.02	Urbanisation	-0.12±0.10	0.24	
			Vegetables	0.17±0.20	0.30	
			Management intensity	-0.05±0.08	0.55	
			Flowers & berries	0.04±0.20	0.81	
Soil PC2	0.60	0.44	Vegetables	-1.62±0.10	<0.001	***
			Flowers & berries	-0.79±0.10	<0.001	***
			Management intensity	-0.05±0.06	0.47	
			Urbanisation	0.04±0.07	0.61	
Soil PC3	0.76	0.01	Vegetables	0.16±0.10	0.16	
			Management intensity	0.07±0.07	0.32	
			Flowers & berries	-0.02±0.10	0.86	
			Urbanisation	-0.01±0.10	0.90	
Soil PC4	0.47	0.02	Vegetables	0.22±0.20	0.18	
			Urbanisation	-0.09±0.10	0.40	
			Flowers & berries	-0.04±0.20	0.83	
			Management intensity	-0.01±0.08	0.93	

Table S 5.8 – Estimated LMEM coefficients of soil multifunctionality **(A)** and its single components: belowground decomposition of green tea bags **(B)**, aboveground decomposition of leaf litter **(C)**, C mineralisation **(D)**, N mineralisation **(E)** and water holding capacity **(F)**, as a function of garden land-use types (cf. effect plots Figure S 5.13). Garden ID was set as random effect in all models. Given are the mean, the 2.5% and the 97.5% quantiles of the Bayesian posterior distribution. Bold numbers indicate significant fixed effects, with credible intervals not crossing zero (Korner-Nievergelt et al. 2015).

	Fixed effect coefficients		
	Flowers & berries vs. Grass	Vegetables vs. Grass	Flowers & berries vs. Vegetables
(A) Soil multifunctionality	-0.02 (-0.06;0.01)	**-0.08 (-0.12;-0.05)**	**0.06 (0.02;0.10)**
(B) Green tea decomposition [%]	**-2.5 (-4.1;-0.8)**	**-3.0 (-4.7;-1.4)**	0.1 (-1.0;2.0)
(C) Litter decomposition [%]	**-9.5 (-18.3;-0.7)**	**-31.1 (-39.7;-22.5)**	**21.7 (11.9;31.4)**
(D) C mineralisation [$mgkg^{-1}$]	**0.03 (0.01;0.05)**	**0.03 (0.01;0.05)**	-0.01 (-0.03;0.02)
(E) N mineralisation [$mgkg^{-1}$]	0.09 (-0.30;0.48)	-0.06 (-0.43;0.32)	0.15 (-0.28;0.58)
(F) Water holding capacity [%]	-0.26 (-3.87;3.32)	-1.16 (-4.58;2.27)	0.91 (-3.08;4.88)

Table S 5.9 – Estimated LMEM coefficients of Bacteria **(A)**, Fungi **(B)**, and the fungal to bacterial ratio **(C)**, as a function of garden land-use types (cf. effect plots Figure S 5.13). Garden ID was set as random effect in all models. Given are the mean, the 2.5% and the 97.5% quantiles of the Bayesian posterior distribution. Bold numbers indicate significant fixed effects, with credible intervals not crossing zero (Korner-Nievergelt et al. 2015).

	Fixed effect coefficients		
	Flowers & berries vs. Grass	Vegetables vs. Grass	Flowers & berries vs. Vegetables
(A) Bacteria [gene copy numbers]	0.05 (-0.21;0.31)	**-0.35 (-0.60;-0.10)**	**0.40 (0.11;0.69)**
(B) Fungi [gene copy numbers]	**0.28 (0.05;0.52)**	**0.40 (0.17;0.63)**	-0.12 (-0.38;0.14)
(C) Fungal to bacterial ratio	0.002 (-0.002;0.005)	**0.010 (0.007;0.013)**	**-0.008 (-0.010;-0.004)**

Table S 5.10 – Soil fauna phylogenetic diversity assessed as phylogenetic species variability (PSV).

Soil fauna phylogenetic diversity, which is usually well correlated with functional diversity in most biodiversity ecosystem functioning studies (Gessner et al. 2010), was assessed as phylogenetic species variability (PSV), representing the mean of the phylogenetic correlations among species, in this case springtail and earthworms, in a community (Paradis 2011). Phylogenetic trees ('rotl' package Michonneau et al. (2016)) were constructed based on the open tree of life project (Hinchliff et al. 2015) with branch lengths ('ape' package Paradis et al. (2004)) to calculate PSV (Paradis 2011).

Table S 5.11 – Measurement details of soil functions and properties used to calculate soil multifunctionality.

We used five measurements to calculate soil multifunctionality with the averaging approach (Byrnes et al. 2014). **1.)** The soil function litter decomposition aboveground was measured with litter bags (18 cm x 18 cm; see Finerty et al. (2016)) with a mesh size of 4 mm on the top and on the bottom a mesh size of 1 mm, in order to prevent smaller pre-decomposed fragments from being lost during the recollection phase. We placed one litter bag on top of the soil layer in each urban garden plot (N=170) for six months (December 2015-May2016), during which most of the leaf litter accumulating in gardens will be decomposed by soil organisms. We only used litter bags with 4 mm mesh size for this calculation and not the ones with 1 mm mesh size on the top of the litter bags, in order to include also macrofauna decomposers in the proxy for aboveground decomposition. Additional leaf litter traits (e.g. C to N ratio or leaf tensile strength) can be found in Tresch et al. (2019) Table A.1. The litter material (*Zea mays* L.) has been oven dried at 40°C and separated manually into leaf and stem parts before weighing. The starting weight in each litter bag was 2±0.01 g leaf and 2±0.01 g stem material (central leaf vein). Furthermore, only leaf litter has been used, since the mean mass loss (79.6±2.2 %) has been significantly higher compared to the more recalcitrant stems (37.9±20.8 % Tresch et al. (2019)).

2.) Litter decomposition belowground of mainly soil microfauna (Keuskamp et al. 2013) was measured by the mass loss of green tea bags in accordance to the tea bag index method by Keuskamp et al. (2013). Per garden plot, four replicated tea bags for each tea type (green and rooibos tea) were buried at a depth of 8 cm for 90 days (mid-October until mid-January 2016). The mass loss, expressed as percentage change before and after decomposition was calculated after drying at 60°C and subsequent incineration of the tea bags without the nylon net (Tresch et al. 2018b), in order to subtract small soil particles (< 0.25 mm, the size of the tea bag mesh) which possibly entered the tea bags during the phase of decomposition (Tresch et al. 2018a). Only green tea decomposition has been used for the calculation of soil multifunctionality, because of higher mean decomposition rates (Keuskamp et al. 2013) (59.0±3.7 %) compared to rooibos tea (29.6±2.8 %) found in Tresch et al. (2018a).

3.) Soil nutrient supply has been assessed by the measurements of N_{min} and C_{min}. N_{min} was measured in an extract with 0.01 M $CaCl_2$ (1:4 w/v) following Krauss et al. (2017).

4.) C_{min} rates were calculated as cumulative values after 4 weeks by incubating 30 g moist soil (40-50 % water holding capacity) at 20°C. CO_2 flux calculations were based on the increase of CO_2 concentration in the head-space over 6 hours, measured once per week for the 4 week time period with a gas chromatograph (7890A, Agilent Technologies, USA) as described in Tresch et al. (2018a) Table S 5.12. The linearity of the enrichment was tested according to Krause et al. (2017).

5.) The capacity of the soil to store water was measured by the soil water holding capacity (WHC). We measured WHC with a cylinder method, where field moist soil is saturated with water on a sand bath following Schinner et al. (1996).

Table S 5.12 – Urban garden land-use types by garden types. Table A) displays all sampled urban garden plots. Table B) illustrates total number of observations without NA's used for the SEM and other statistical analyses. The discrepancy in observations is due to many reasons, such as missing litter bags on some sites or missing values in laboratory analyses for the measurement of the soil quality indices.

(A)	Urban garden land-use types			
	Perennial flowers & berries	Perennial grass	Annual vegetables	**Total**
Allotment garden sites	19	29	36	**84**
Domestic garden sites	33	42	11	**86**
Total	**52**	**71**	**47**	**170**

(B)	Urban garden land-use types			
	Perennial flowers & berries	Perennial grass	Annual vegetables	**Total**
Allotment garden sites	18	27	34	**79**
Domestic garden sites	27	38	11	**76**
Total	**45**	**65**	**45**	**155**

6

Discussion and Conclusion

6.1 General discussion

Urban gardens are important parts of urban greenspaces. In many cities, including the city of Zurich, they are the dominant land-cover type of urban greenspaces. Therefore, the way gardeners manage their small parcels of land can have an influence on the local as well as landscape scale biodiversity. Worldwide, there is an increasing interest to foster urban biodiversity and ES from greenspaces, such as water infiltration, carbon storage, cooling air temperatures or creating hot-spots for biodiversity and for human recreation (cf. Table 1.1). In an urban context, even small urban gardens can provide ES with a high impact for urban biodiversity and for the well-being of citizens (Dearborn and Kark 2010). However, few studies have measured ecosystem functions in urban gardens, probably due to difficulties in obtaining permission to sample and measure in privately owned urban gardens. With ongoing urbanisation and densification of cities as well as climate change, human pressure on urban greenspaces is increasing and thus the understanding and quantification of services provided by these ecosystems become highly important.

In chapters 2 and 3, a city-wide assessment of soil quality has been conducted. One of the main findings was that decades of urban gardening activities increased the overall soil quality of urban garden sites. This was indicated by biological soil quality indicators such as C_{mic} and the abundance and diversity of earthworms. The design and management of the garden land-use type was the driving factor influencing the multivariate grouping of soil quality indices. This was also highlighted by eco-physiological indicators of soil quality. The microbial quotient (C_{mic}/SOC), describing the amount of microorganisms related to the organic carbon content in the soil, has been shown to be sensitive to agricultural soil disturbances such as tillage or crop rotation (Anderson and Domsch 1989). Over all 170 investigated sites, we found, as expected, a higher microbial quotient in perennial grass sites compared to flower and berry sites with lowest values in annual vegetable sites. The high management intensity in those sites led to detectable differences in soil microbial communities, irrespective of the total amount of organic carbon. As we hypothesised, the metabolic quotient (qCO_2), the ratio of basal respiration rate to C_{mic} describing the substrate mineralised per unit of microbial biomass, was highest for the most intensively disturbed urban garden land-use type annual vegetables and lowest for perennial grass sites. This has proven that biological soil quality indices, which have been shown to respond sensitive to management practices in agricultural soils (Mäder et al. 2002), could also be used in urban garden soils in order to detect intensive garden management practices. Soil protective management practices such as the use of compost or mulch could be associated with increased soil quality indices, while intensive practices like digging, tillage and the use of plant protection agents decreased soil biological activity. In conclusion, soil quality of urban gardens, irrespective of the garden type, was increased compared to other urban soils and agricultural soils (cf. Table 3.4). A current ecological theory of urban soils states that the high impact of anthropogenic management on pedogenic processes leads to a convergence of soil characteristics especially in biological soil properties such as SOC or TON (Pouyat et al. 2015). Urbanisation certainly has a strong impact on biological soil characteristics, through the replacement of natural vegetation by nonnative species or by impervious surfaces (Lal 2018b). However, managed urban soils, such urban garden soils, might be more affected by the decades of soil management (Joimel et al. 2016) and local geogenic factors (Beyer et al. 2001) such as the amount of clay. For example, organic amendments such as compost are known to affect biological soil quality indicators such as C_{mic} or enzymatic activities (Albiach et al. 2000) in garden soils, but also physical parameters such as porosity, aggregate stability or water holding capacity (Giusquiani et al. 1995, Cogger 2005). In general, we found rather diverging soil characteristics with increasing urbanisation. For example, SOC increased with urbanisation. Nonetheless, the hypothesis that the variety of different management practices will lead to diverging soil characteristics should be investigated in other cities as well, taking into account more than five soil quality measurements as Pouyat et al. (2015) did. In addition, soil functions should also be included in future studies, due to their importance for key ES in cities. Moreover, the analysis of soil heavy metals in chapter 3, could not be linked to intensive soil management practices, but to spatial patters of traffic and industry. We found large variation in heavy metal concentration, for instance Pb varied from 18.5 to 1076 mg kg^{-1}. The upper guideline limit for Pb in Swiss soils of 1000 mg kg^{-1} (Swiss Federal Council 1998) has been exceeded only once, but 16 % of all garden sites are requiring further inspections (> 300 mg kg^{-1}). These facts were communicated to the garden owners in the case of private gardens and to the official authorities in Zurich (Amt für Landschaft und Natur, Fachstelle Bodenschutz). Interestingly, we also found that soil functions, such as decomposition,

assessed as belowground decomposition with the tea bag index (Keuskamp et al. 2013), were affected by garden management decisions, with highest decomposition values for green tea bags in perennial grass sites. This relationship of management intensity and soil function decomposition was investigated in more detail in the following chapter.

In chapter 4, we analysed the relationship between soil fauna diversity and the soil function litter decomposition. Litter bags with different mesh sizes were used in order to account for the contribution of macro- and mesofauna to litter decomposition. We were interested in whether there is a trend of higher soil functioning with increased soil fauna diversity. Similar to observations found with productivity as ecosystem service and plant diversity as a measure of biodiversity (Duffy 2009, Cardinale et al. 2011, Caruso et al. 2018). It has been shown that there is a positive relationship between diversity and litter decomposition (Allan et al. 2013, Handa et al. 2014, Weisser et al. 2017), but the effect of altered decomposer communities on litter decomposition remain difficult to predict, especially in real world experiments (Hättenschwiler et al. 2005, García-Palacios et al. 2016a). The majority of studies have analysed the effect of litter species diversity on decomposition, with a mean positive effect on litter mass loss across different biomes (Handa et al. 2014). However, the relationship between ecosystem functioning and belowground diversity in soils is less well known, especially in the case of the spatially very heterogeneous urban soils. Furthermore, we have chosen litter decomposition as a model ecosystem function, because of its importance in maintaining soil quality (Schram-Bijkerk et al. 2018) and because there are only few studies about the effect of urbanisation on litter decomposition in urban soils (Dorendorf et al. 2015). In summary, we found the highest litter decomposition rates in litter bags including soil macrofauna species. This indicates the importance of an intact soil fauna for decomposition. In an a priori conceptual model we hypothesised direct effects from decomposer species, including earthworms, isopoda, gastropoda and collembola, on litter decomposition and indirect effects such as soil characteristics, garden management and urbanisation affecting both soil fauna organisms and litter decomposition. This path model unravelled that litter decomposition was directly affected by soil fauna species richness and microbial activity, assessed by the community level physiological profile measurements of the MicroResp technique. In accordance to my hypothesis, that belowground diversity of decomposer organisms is positively connected with soil function litter decomposition, results of both the LMEM and the SEM indicated a positive effect of species richness on litter decomposition. However, functional diversity assessed by the trait even distribution (TED; Fontana et al. (2016)), as well as phylogenetic diversity assessed as phylogenetic species variability (PSV; Paradis (2011)) were not explaining much variance of decomposition in this data set. Besides the high correlation between taxonomic and functional diversity in many biodiversity ecosystem functioning investigations (Gessner et al. 2010), this might also be explained by the relatively small variation among soil fauna communities found in the urban gardens, resulting in similar functional and phylogenetic indices. Interestingly, the SEM also showed, that a higher plant diversity positively affected soil meso- and macrofauna species richness, as well as microbial activity and thus indirectly also affected litter decomposition. This gives rise to the assumption that above- and belowground diversity is as well connected in heavily disturbed garden soils. Although we assumed that rather local effects, such as different soil characteristics, than landscape effects, such as urbanisation, would have the biggest effect on litter decomposition, the data indicated a positive effect on decomposition with increasing urbanisation. This might be related to the fact that in our data set the abundance of decomposer species increased with urbanisation. This pattern might be driven by an elevated abundance of isopods in more urbanised gardens, probably due to the availability of habitats and calcium needed for crustacea species. The garden land-use type with the highest soil disturbance, annual vegetables, showed a decreased potential for decomposition and intensive soil management practices, such as digging or loosening the soil, were negatively correlated with litter decomposition. Complementary to litter mass loss, an analysis of organic compounds found in litter residues after the decomposition period, confirmed the importance of soil fauna species richness on the litter decomposition processes. The shortage of organic material, for instance because of the frequent removal of grass clippings and leaf litter, has been suggested to negatively affect soil fauna communities and disrupting the soil function decomposition (Szlavecz et al. 2018). In this chapter, we showed that a decreased soil fauna diversity and more labile organic compounds (midDRIFTS analysis) were found on the highly disturbed sites. Hence, a changed decomposer community composition not only affected litter mass loss, but also which organic compounds will be decomposed faster.

On the basis of the results of this chapter that above- and belowground diversity could also be connected in urban garden soils, we investigated this relationship using aboveground plant data generated from SPD and belowground diversity of soil fauna in the next and last last chapter of this PhD thesis.

In chapter 5, we intended to highlight effects of urban food production on soil fauna community structure and soil multifunctionality. The differences in soil quality and belowground diversity between the garden land-use types, especially between perennial grass and annual vegetables sites, was the rational background of analysing the relationship between aboveground diversity using the plant data set of 600 plant species (Frey and Moretti 2019), and belowground diversity and related soil functions. We focused on five independent measurements, important for urban food production in order to describe soil multifunctionality. They include aboveground (litter bags) and belowground (tea bags) decomposition, nutrient supply for plant growth (N and C mineralisation) and water regulation (water holding capacity). The concept of ecosystem multifunctionality has recently gained recognition in order to analyse changes of a range of ecosystem drivers (Manning et al. 2018), such as management intensity (Allan et al. 2015), or climate change (Delgado-Baquerizo et al. 2016), but has not been applied to urban soils. We assumed that garden management will directly affect plant diversity, since many of them are constantly managed by gardeners. We hypothesised that plant diversity will have a positive effect on soil fauna diversity, in this case earthworm and collembola species, and soil fauna biomass, and thus also on soil multifunctionality. Based on the previous results showing higher decomposition rates and increased abundance of certain soil organisms in more urbanised gardens, we included a direct pathway in the SEM between urbanisation, soil fauna diversity, soil fauna biomass, and soil multifunctionality. The results of the SEM showed that the strongest effects on soil multifunctionality were caused by soil characteristics. Similarly to the effects of biological soil characteristics on soil quality (chapter 3), indices of high soil biological quality such as organic and microbial carbon, and the abundance of bacteria, had a positive effect on soil multifunctionality, whereas management intensity decreased soil multifunctionality by reducing soil fauna biomass. Higher plant diversity in gardens did not only have a direct effect on soil multifunctionality but also an indirect effect by increasing biomass and species richness of soil fauna. These results particularly highlight the potential of increasing aboveground diversity and its positive effects on potential soil functions. The influence of gardeners by creating different garden land-use types was again a main influencing factor, affecting soil functionality, fauna biomass and plant diversity, as well as diverse soil characteristics. The data shows that food production on annual vegetables, decreased the biomass of earthworms and collembola species and on average a less diverse plant community has been found on those sites. Nevertheless, I see a great potential for optimising soil quality and functioning by increasing plant diversity with soil protective management practices such as inter-cropping, applying green manure (i.e. *Phacelia tanacetifolia*; Studer (2010)), catch crops (i.e. *Trifolium pratense*; Lindner et al. (2012)) or mulch. Additionally, the use of plant indicator values, assessed as community weighted means of the plant data set, proved useful to show differences in soil fauna community structure even in anthropogenic arranged plant assemblages. For instance the effect of high nutrient loads, predominantly found in annual vegetable beds, was correlated with a specific plant indicator value, affecting the community structure of both earthworms and collembola. However, urbanisation had no significant effect on soil multifunctionality in this analysis, but it increased the overall model performance and because of this it was integrated in the final SEM.

6.2 General conclusion and perspectives

Urban gardens are an essential but often overlooked and neglected part of urban greenspaces. In the face of increasing human population, especially in urban areas, and increasing soil sealing, those urban greenspaces are being challenged (Tappert et al. 2018). It is known that particularly urban gardens can harbour immense biodiversity, but their value for citizens goes far beyond this or simply the production of food (Goddard et al. 2010, Dearborn and Kark 2010, Lin and Egerer 2017).

This thesis represents a step forward in highlighting the importance of urban gardens for urban biodiversity and urban ecosystem functioning. In summary, effects of garden management practices on soil quality, soil fauna communities and certain soil functions has been assessed (see chapter 6.1). Nevertheless, the picture is far from being complete, and several aspects could not be considered, but would be of great interest for future research. One of them being effects of soil contamination on soil functioning and food quality. Many gardeners are

not aware of the many negative consequences due to the sometimes excessive use of pesticides in gardens (Zaller 2018, Gubser and Butterweck 2018). Pesticide applications are often not measured precisely by the gardeners and labelled rates can be much higher than for agricultural use (Sponsler et al. 2019), leading to even higher pesticide values in surface water of urban compared to agricultural areas (Gilliom et al. 2006). Since 2001, the use of herbicides by private users is restricted in Switzerland (SR 814.81 2001). Private users may no longer use herbicides on their driveways, gravel paths, roofs and terraces due to acute risks of leaching into water bodies (Gubser and Butterweck 2018). Although the use of pesticides is restricted in many Swiss urban allotment gardens, including those in Zurich (Lorenz and Bossardt 2018), many gardeners still use a wide range of pesticides, often in combination with synthetic fertilisers (Christl et al. 2004). However, the total amount of pesticides used in gardens and households is difficult to assess (Zaller 2018). For example, it is estimated that six percent of the annual pesticides usage in the US are used in residential areas (Adwood and Paisley-Jones 2017). In Switzerland, around 400 tons of pesticides are sprayed every year in private gardens, including total herbicides, which are mainly used to keep paths and places free of weeds (Fasolini 2015). In Germany, about 6000 tons are used annually by non-professional users (BVL 2018). A recent study has revealed that every second gardener in Switzerland has never heard of the prohibition from 2001 (Gubser and Butterweck 2018). This is in line with the pesticide usage of urban gardeners in Zurich. An additional analysis of the management survey of SPC and SPD (cf. Table S 5.3), revealed that almost every second (45.2 %) urban garden study site (n=168) was still treated with pesticides. The highest percentages of pesticide application were found in perennial flowers and berries (48.1 %) followed by perennial grass (47.1 %) and vegetable beds (39.1 %). The reasons for this can be numerous, such as old habits, or the use up of old stocks. Nevertheless, another survey in allotment gardens of Zurich revealed that a majority (72 %) of the gardeners agree that the family garden association should decide on the permitted fertilisers and plant protection products (Christl et al. 2004). To conclude, there should be a rethinking of the use of pesticides and a serious reduction in the usage of chemical plant protection agents in urban greenspaces in order to preserve biodiversity (Sánchez-Bayo and Wyckhuys 2019).

However, contamination can also originate from other sources such the application of contaminated sewage sludge, ashes or compost (Christl et al. 2004) but also because of the proximity to pollution sources such as roads and railways (Antisari et al. 2015). Based on the literature review about major soil contamination in garden soils and the close contact to the gardeners during the field work, I suggest that besides the need to provide more information on best management practices (Christl et al. 2004, Bretzel et al. 2018, Raymond et al. 2019, Tresch 2019), maintaining soil quality and reducing health risks, in the form of generally understandable brochures (e.g. Jokinen et al. (2016), Lorenz and Bossardt (2018), Lerman et al. (2018), Studer (2010)), there should be a cost efficient service for the analysis of main soil contaminants. This analysis should include Pb, Cd, and PCB for soil samples from vegetable beds and Hg as well as PAH in the presence of children, due to the risk of direct soil uptake (Christl et al. 2004). This service should be designed in such a way that gardeners can send in their soil samples, or pay for an exact sampling, for instance with GPS coordinates, enabling a future re-sampling on the exact same sites. This would allow to analyse effects of specific management practices or also if there had been any inherited waste deposits on those sites. Ideally, this would be coordinated by the cantonal soil protection offices. The same holds true for the need of a simple nutrient balance analysis (i.e. Kinsey and Walters (1999)), although soil contaminants are a more urgent problem with regards to serious human health issues, than nutrient deficiency for urban produced crops.

Another interesting topic, not covered in this thesis, is the effect of urban trees in urban gardens. Since urban trees in parks or other public areas have been shown to provide beneficial ES in cities such as removing air pollution, storing C or mitigating urban heat island effects (Nowak et al. 2006, Edmondson et al. 2012, Strohbach and Haase 2012, Bodnaruk et al. 2017). The effect of trees on the local plant and fauna diversity in urban gardens has not yet been investigated. Such a study could be combined with the effect of trees on soil properties and soil functions. Generally, a study about the effects of plant assemblages on ES and resource input, such as fertilisers or water, could deliver new insights into the functioning of urban landscapes and lead to recommendations for gardeners and city nurseries.

In addition, although the biodiversity assessment of soil organisms was an important part of this thesis, we could not take into account the overall diversity of soil organisms, which is in fact almost impossible, given the large number of taxonomic groups. How-

ever, the analysis of other soil invertebrates such as nematodes or enchytraeidae could eventually have revealed more connections to soil management practices, as they have been used as test species in ecotoxicological studies and biological soil assessments (Crotty et al. 2015, Maraldo et al. 2015). Also recently assessed soil quality indicators such as labile organic carbon fractions (especially particulate organic matter carbon, hot water extractable carbon, and permanganate oxidisable carbon), that have been shown to be sensitive to soil management and soil functions (Bongiorno et al. 2019a;b), could have given more insights about the effect of management practices on soil quality and functions.

Future studies in urban gardens can build on the large biodiversity assessment of SPC and SPD, which is according to our knowledge the biggest inventory of urban biodiversity worldwide, covering over 1100 invertebrate species (> 155'000 individuals) and over 1100 plant species. Moreover, this interdisciplinary project has also highlighted the importance of social ES provided by urban gardens, such as the increase of human health by recreation or stress reduction through physical gardening work (Hofmann et al. 2018). Especially in SPB, it has been pointed out that urban gardens are a place for social interaction among gardeners with different ethnic and educational backgrounds. In addition, first results of a large survey in the cities of Zurich, Bern, and Lausanne revealed that gardeners were in favour of improving biodiversity in their gardens and believe that biodiverse gardens are better for their recreation. This tendency towards biodiversity could be used to investigate effects of increasing biodiversity together with gardeners, for instance in a citizen science approach. It has been shown that urban gardening provides a great opportunity to explain ecological processes and to broader the knowledge of urban nutrient cycles and the value of biodiversity for ES (Clayton 2007, Bretzel et al. 2018, Egerer et al. 2018c). While this current project investigated long-term effects of different garden management practices, a follow up study could for example include a catalogue of practices to enhance local biodiversity (above- and belowground) and thus also the recreation value of more biodiverse gardens. Such a project could be established together with gardeners and city administrations, based on the high support of biodiversity issues. This integration of the persuasion of gardeners could finally lead to new guidelines for garden organisations and city green administrations.

In all investigations the management of garden land-use types had a major impact on soil quality, biodiversity and several soil functions. Taken together with the results from the analysis of management intensity and several individual garden management practices, I can give some general recommendations for improving soil quality and biodiversity in garden soils. Soil protective organic management practices (e.g. Sanders and Heß 2019) such as intercropping or green manures together with a lower soil disturbance, particularly in annual vegetable beds, should be applied more often. For example, a rotary tiller is not needed in gardens, because of its detrimental impacts on soil organisms (e.g. Pfiffner and Luka 2007) and because a loosening of the soil with digging forks is usually sufficient enough for planting and for mixing organic matter into the soil surface. These measures will not only increase the abundance of soil organisms in the long term, but will also be beneficial for soil quality and litter decomposition. Another rather simple management advice is to plant more flowers or even promote plant diversity, as it has been shown to increase soil multifunctionality (chapter 5) and the diversity of soil organisms (chapter 5 and 4). This is in line with results from grasslands, shrublands and forests, indicating more SOC storage and productivity with ecosystem managements maintaining higher plant diversity (Chen et al. 2018). However, there are still a lot of additional questions that could be addressed within future studies, for example soil functions related to carbon storage, water filtering or binding soil pollutants. Furthermore, it is necessary to compare different management regimes and their impact on biodiversity. For example a recent study has shown that also on perennially vegetated lawns, a lower management intensity is beneficial for biodiversity (Lerman et al. 2018). They found that mowing only every three weeks instead of every week increased the number of flowers by 2.5 times and thus also the abundance and diversity of bee populations. These initial results encourage to investigate effects of ecological management practices on biodiversity and soil quality in future studies. Moreover, a program to certify ecological gardening practices, similar to the Backyard Wildlife Habitat program of the US National Wildlife Federation, the Wild About Gardens campaign from the Royal Horticultural Society in the UK or the "Natur im Garten Plakette" in Austria, would foster biodiversity actions in urban gardens of Switzerland.

Overall, this thesis also revealed that urban garden soils were often neglected in urban greenspace planning strategies, although they are a key resource for urban biodiversity and impact many ES in cities, including certain SDG's. Hence they should be integrated in future urban planning strategies especially on the city level.

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Acknowledgement

First and foremost, I would like to thank my supervisors Dr. Andreas Fliessbach, Dr. Marco Moretti and Dr. Claire Le Bayon for their continuous support during all phases of my PhD. It was always a pleasure to discuss issues about the PhD project together with them as a team. Their assistance and also constructive criticism had a major influence on the outcome of my work.

My sincere thanks also goes to Dr. Paul Mäder, who helped me with his many years of experience and financed the last two months my PhD. In this regard, I want to thank the whole Department of Soil Science at FiBL for the pleasant working atmosphere and the possibility to make many measurements myself in the soil laboratory. Special thanks goes to Bernhard Stehle, Martina Lori, Toni Kuhn, Adolphe Munyangabe, Dr. Michael Scheifele, Dr. Maike Krauss, Dr. Hans-Martin Krause, Dr. Sarah Symanczik, Fernando Sousa and Dominika Kundel for supporting me in various measurements, data analysis questions or for having inspiring coffee breaks at FiBL.

Moreover, I'm grateful for the BetterGardens team, under the lead of Dr. Robert Home, who always helped me in improving my manuscripts and especially my PhD colleagues at WSL: David Frey and Andrea Zanetta, for their great help during field work and the phase of data analysis and writing. I hope that we will be able to protect more insect lives in future projects.

In addition, this thesis would not have been possible without the help of many students during field work, especially I would like to thank the great "team S" 2015/2016: Lena Fischer, Stefan Grubelnig and Reto Henzmann. I thank to Dr. Frank Rasche, who invited me to measure in his laboratory for three months at the University of Hohenheim and Prof. Dr. Rainer Schulin for giving me access to the X-Ray device at ETH.

Last but not least, I would like to thank all people who have accompanied me through my PhD: My family and dearest friends in Basel (Katja and Daniel Haefelfinger, Pascal and Johanna Oehler, Leonie Haefelfinger, Luca Tarelli, Michael Weber, Ben and Steph Frauchiger, Michi and Rahel Gysel, Nadine and Joel Drozd, Andreas Schomburg et al.) and in Nidwalden (Ruth Tresch, Stefan and Karin Tresch, Martina Tresch, Oliver Zoller, Pascal Bircher, Danny Gasser, Manuel Schaub, Rafael Von Wyl et al.).

Thank you Rebekka for having chosen me! You make sure that I did not forget that there are also other important things in life. You are the love of my life!

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About me

Graduated soil scientist in environmental geosciences with an interdisciplinary background in geomorphology and environmental geochemistry. Currently working as a PhD student at the department of soil science at the Research Institute of Organic Farming (FiBL) Switzerland and at the research group of Biodiversity and Conservation Biology at the Swiss Federal Research Institute for Wood Snow and Landscape (WSL).

The doctoral thesis has been part of the interdisciplinary research project "Strategies for Better Gardens: Integrated Analysis of Soil Quality, Biodiversity and Social Value of Urban Gardens", which has been founded by the SNF Sinergia program.

Education

Jan 2015 - May 2019, PhD Biology

PhD candidate at the interuniversitary program Organismal Biology, University of Neuchâtel, Institute of Biology, Functional Ecology Laboratory under supervision of Dr. Claire Le Bayon (Uni Neuchâtel), Dr. Andreas Fliessbach (FiBL) and Dr. Marco Moretti (WSL).

2012 - 2014, M.Sc. Environmental Geosciences

M.Sc. in Geoscience, Research Group Prof. Alewell, Environmental Geoscience, Department of Environmental Sciences, University of Basel, CH. Master Thesis: "Influence of the slope steepness on soil erosion modelling measured with rainfall simulations in the Urseren Valley", Supervision Dr. Katrin Meusburger.

2009 - 2012, B.Sc. Geosciences

B.Sc. in Geoscience, Research Group Prof. Kuhn, Physical Geography and Environmental Change, Department of Environmental Sciences, University of Basel, CH. Bachelor Thesis: "Analyse von empirischen Daten des portablen Wind- und Regensimulators (PWRS) zur Vereinfachung der Probennahme", Supervision Dr. Wolfgang Fister.

2008 - 2009, Paramedic, Swiss Military Service

Education on paramedics as a full time military servant (Durchdiener).

2001 - 2008, Matura canton Nidwalden

Swiss Academic Baccalaureate, Kollegium St. Fidelis Stans, Switzerland. Major in physics and applied mathematics.

Work Experience

February 2019, Guest lecturer, ZHAW
Zurich University of Applied Sciences; Course: Plant Utilisation; Module: Plants and Substrates; Topic: Soil Biodiversity 6 lessons.

July 2018, February 2019, Substitute teacher
Araau Cantonal School, in Biology, Mathematics and Physics 10^{th} school year.

February 2019, Assistance R course, WSL
Introduction into R, Dr. Jan Wunder.

May 2018 & 2019, Assistance R course, WSL
Advanced data management & manipulation using R, Dr. Jan Wunder.

April 2018 & 2019, Assistance R course, WSL
Scientific Visualisations using R, Dr. Jan Wunder.

2017 - 2018, Supervision Master Thesis
Master Thesis Bernhard Stehle: "Assessing microbial community-level physiological profiles (CLPPs) of urban gardens using a whole soil approach", Department of Biology, University of Konstanz.

April 2014 - Dec 2014, Student Assistant (HIWI) Uni Basel
Working in the research Group of Prof. Dr. Christine Alewell. Key tasks were the resampling project for the soil erosion assessment in Swiss mountainous areas (PhD project of Laura Arata), including fieldwork and preparation of the soil cores for radionuclide measurements (^{137}Cs, $^{239-240}$Pu). Additionally, the effects of snow movements (shallow landslides) on soil erosion were investigated with snow glide shoe measurements and field observations.

October 2011 - August 2013, Student Assistant (HIWI) Uni Basel
Working for the research Group of Prof. Dr. Nikolaus Kuhn. The main duties included experimental soil erosion research under laboratory conditions and field measurements (see Fister et al., 2013).

October 2011 - August 2013, Internship rain forest Ecuador
Internship reforestation of primary rain forest in Misahualli, Ecuador. Sustainable forestry and biodiversity protection.

January 2012 - February 2012, Internship AUE Basel Stadt
Internship AUE (Amt für Umwelt und Energie), Basel Stadt. Practical experiences in ground water protection and ground water level monitoring (GIS based) including hydrological data processing.

2007 - 2011, Summer jobs
Summer jobs as a carpenter (Waser Holzbau AG, Oberrickenbach (NW) and Flury Innen und Aussen AG, Stans (NW)) and gardener (private gardens, Buochs, NW).

2010, Driving licence (B)

Peer-reviewed publications

PhD project BetterGardens

- Frey, D., **Tresch, S.**, Young, C., Zanetta, A., Bauer, N., Fliessbach, A., Ghazoul, J., Home, R., Moretti, M., 2019. Social-ecological interactions determine the richness and structure of urban invertebrate communities. People and Nature in prep.
- **Tresch, S.**, Frey, D., Le Bayon, R.-C., Mder, P., Stehle, B., Fliessbach, A., Moretti, M., 2019. Direct and indirect effects of urban gardening on aboveground and belowground diversity influencing soil multifunctionality. Scientific Reports, 9, 9769.
- **Tresch, S.**, Frey, D., Le Bayon, R.-C., Zanetta, A., Rasche, F., Fliessbach, A., Moretti, M., 2019. Litter decomposition driven by soil fauna, plant diversity and soil management in urban gardens. Science of the Total Environment, 658, 1614-1629.
- **Tresch, S.**, Moretti, M., Le Bayon, R.-C., Mäder, P., Zanetta, A., Frey, D., Fliessbach, A., 2018. A Gardeners Influence on Urban Soil Quality. Frontiers in Environmental Science. Frontiers in Environmental Science, 6.
- **Tresch, S.**, Moretti, M., Le Bayon, R.-C., Mäder, P., Zanetta, A., Frey, D., Stehle B., Kuhn A., Munyangabe A., Fliessbach, A., 2018. Urban Soil Quality Assessment A Comprehensive Case Study Dataset of Urban Garden Soils. Frontiers in Environmental Science, 6, Data Report Article.
- Home, R., Lewis O., Bauer N., Fliessbach A., Frey D., Lichtsteiner S., Moretti M., **Tresch S**., Young C., Zanetta A., Stolze M. 2018. Effects of garden management practices, by different types of gardeners, on human wellbeing and ecological and soil sustainability in Swiss cities. Urban Ecosystems, 22, 189-199.

Other projects:

- Schmidt, S., **Tresch, S.**, Meusburger, K., 2019. Modification of the RUSLE slope length and steepness factor (LS-factor) based on rainfall experiments at steep alpine grasslands. MethodsX, 6, 219-229.
- Lori, M., Symanczik, S., Mäder, P., Efosa, N., Jaenicke, S., Buegger, F., **Tresch, S.**, Goesmann, A., Gattinger, A., 2018. Distinct Nitrogen Provisioning From Organic Amendments in Soil as Influenced by Farming System and Water Regime. Frontiers in Environmental Science, 6.

Other publications

- **Tresch, S.**, Frey, D., Moretti, M., 2019. Gärten mit höherer Pflanzenvielfalt fördern Bodentiere und Bodenfunktionen. Natur + Landschaft: Inside. in prep.
- **Tresch, S.**, Fliessbach, A., Le Bayon, R.-C., Frey, D., Moretti, M., 2018. Artenvielfalt von Pflanzen und Bodentieren fördert den Abbau von organischem Material in Stadtgärten. Natur + Landschaft: Inside. 3/2018.
- Home, R., Stolze, M., **Tresch, S.**, Fliessbach, A., Lewis, O., Bauer, N., Moretti, M., Young, C., Zanetta, A., Frey, D., 2017. BetterGardens: qualité du sol, biodiversité et valeur sociale des jardins de la ville. VBB-Bulletin. 17.
- Frey, D., Young, C., Zanetta, A., **Tresch, S.**, Fliessbach, A., Bauer, N., Moretti, M., 2017. Bettergardens: Biodiversität, Bodenqualität und sozialer Wert von Stadtgärten. Natur + Landschaft: Inside. 2/2017.
- **Tresch, S.**, Pfiffner, L., 2017. Regenwürmer - Baumeister der Bodenfruchtbarkeit. Bioaktuell. 4/2017.
- **Tresch, S**., Fliessbach, A., 2016. Decomposition study using tea bags. FertilCrop Technical Note. Research Institute of Organic Agriculture (FiBL), Frick.
- **Tresch, S.**, Fliessbach, A., 2016. Bodenorganismen haben lieber Grüntee. Bioaktuell. 9/2016.
- Lichtsteiner, S., Home, R., Moretti, M., Frey, D., Fliessbach, A., **Tresch, S.**, Young, C., Bauer, N., 2016. Der ökologische und soziale Wert von Stadtgärten. HOTSPOT. 33.
- Fister, W., Rüegg, H.-R., **Tresch, S.**, Greenwood, P., 2013. Präzisionsanlage zur experimentellen Untersuchung von Bodenerosionsprozessen. Reg. Basil. 54.

Interviews

- Brune, R., 29.04.2019 & 27.05.2019. Mission B: Vielfalt der Regenwürmer in Gärten. Radio SRF 3.
- Weiss, H., 2018. Regenwürmer Schwerstarbeiter im Boden. Tierwelt 7/2018.
- Weiss, H., 2017. Einflüsse der Regenwürmer im Boden, wie kann man diese im Garten fördern?. Welt der Tiere. 5/2017.
- Schulte, R., 2015. Feldmeister - Am Nabel der Bioforschung, Coop Zeitung. 36.

Conference contributions

- International Conference URBIO: Urban Biodiversity and Food Security, Cape Town, 12.09.18. Biodiverse gardens increase litter decomposition, poster.
- 48th Annual Meeting of the Ecological Society DE, AT, CH (GFÖ), Vienna, 02.09.18. Garden management impacts soil quality belowground biodiversity and ecosystem functioning in urban ecosystems, poster & talk.
- 1. Schweizer Landschaftskongress, Luzern, 23.08.18. Importance of soil quality management to support ecosystem services in urban gardens - A case study from Zurich, poster.
- Biology 18 Annual Swiss Biology Conference, Neuchâtel, 14.02.18. What drives the decomposition of organic material in urban gardens?, poster.
- Biology 16 Annual Swiss Biology Conference, Lausanne, 11.02.16. Soil quality & biodiversity indicators in urban gardens of Zurich, poster.
- Ecosummit conference Montpellier, France, 15.01.16. Eco- physiological soil quality indices affected by urban garden management in the city of Zurich (CH), poster.
- Swiss Geoscience Meeting 2015, Basel, 20.11.15. Soil quality indicators in urban gardens of Zurich, poster.

Other presentations

- Projekt Symposium BetterGardens, WSL, Birmensdorf, 04.03.19. Ergebniss Projekt C: Einfluss von Gartenaktivitten auf die Bodenqualität und Biodiversität.
- Stadtgärtnerei Basel Netzwerktreffen Biodiversität, Basel, 21.02.19. Key note: Bodenqualität und Biodiversität fördern in urban Gärten.
- EPFL Laboratory of Ecological Systems seminar, Lausanne, 15.01.19. Urban gardening affects soil quality and multifunctionality.
- Fachstelle Bodenschutz (FaBo) Kanton ZH, Zürich, 01.10.18 & 02.03. 15. Bodenuntersuchungen im Projekt BetterGardens.
- Stand am Tag der offenen Tür am FiBL, Frick, 19.08.18. Einblicke in die Vielfalt der Bodenorganismen: Der Boden lebt.
- Blockkurs Ökologie Uni Basel, FiBL, Frick, 17.05.18. Bodenfunktionen und Biodiversität in urban Böden Zürichs.
- Fachgruppe Vollzug Bodenbiologie (VBBio), FHNW Olten, 22.11.16. Erste Ergebnisse der Bodenqualität von Gartenböden in Zürich.
- BioTerra Jahrestagung, Einsiedeln, 27.01.18. Key note: Bodenbiodiversität in Gartenböden.
- BioTerra Naturgartentag, FHNW Wädenswil, 24.11.17. Key note: Biodiversität im Boden.

Skills

Laboratory expertise

- Soil physical measurements (soil texture, WHC, PV, BD, penetration resistance, SA)
- Soil chemical measurements (pH, EC, nutrients, heavy metals (X-Ray Fluorescence))
- Soil biological measurements (basal respiration, Cmic, Nmic, Nmin, Earthworms, Collembola, Mites)
- Soil organic matter characterisation (SOC, TON, DOC, DON, Mid-DFIRTS (diffuse reflectance Fourier transform mid-infrared spectroscopy)
- Soil microbial activity and community level physiological profiles (MicroResp system and GC), 16S & 18S qPCR
- Stable isotopes ($^{12}C/^{13}C$)
- Rainfall simulations indoor and outdoor

Computer software

- R - statistical analysis (data management, univariate & multivariate models, SEM, Bayesian approaches), scientific visualisations
- LaTex
- Arc GIS
- JMP, SPSS
- MS Office, Photoshop

Languages

- German (native)
- English (fluent)
- French (intermediate)
- Spanish (basics)

Personal interests

- Hiking
- Ski touring
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