Journal of Urban Ecology

*Journal of Urban Ecology*, 2017, 1–10

doi: 10.1093/jue/jux002
Short note

SHORT NOTE

# Introducing GLUSEEN: a new open access and experimental network in urban soil ecology

Richard V. Pouyat,[1,*] Heikki Setälä,[2] Katalin Szlavecz,[3] Ian D. Yesilonis,[4] Sarel Cilliers,[5] Erzsébet Hornung,[6] Stephanie Yarwood,[7] D. Johan Kotze,[2] Miklós Dombos,[8] Michael P. McGuire,[9] and Thomas H. Whitlow[10]

[1]USDA Forest Service, Research & Development, 1400 Independence Ave. NW, Washington, DC 20250 USA, [2]Department of Environmental Sciences, University of Helsinki, Niemenkatu 73, Lahti, FIN 15140, Finland, [3]Department of Earth and Planetary Sciences, Johns Hopkins University, 3400 N. Charles St., Baltimore, MD 21218 USA, [4]USDA Forest Service, c/o Baltimore Ecosystem Study, 5523 Research Park, Suite 350, Baltimore, MD 21228 USA, [5]Unit of Environmental Sciences and Management, North-West University, Potchefstroom, South Africa, [6]University of Veterinary Medicine Budapest, Budapest, H-1077, Hungary, [7]Environmental Science and Technology, University of Maryland, 1204 HJ Patterson Hall, College Park, MD 20742 USA, [8]Institute for Soil Sciences and Agricultural Chemistry, Hungarian Academy of Sciences, Herman Ottó út 15, Budapest, 1022, Hungary, [9]Department of Computer and Information Sciences, Towson University, 8000 York Road, Towson, MD 21252 USA and [10]Section of Horticulture, 23 Plant Science, Cornell University, Ithaca, NY 14853 USA

*Corresponding author. E-mail: rpouyat@fs.fed.us

**Submitted:** 20 May 2016; **Received (in revised form):** 30 November 2016; **Accepted:** 6 December 2016

## Abstract

In an effort to study urban soil ecological systems, we have recently piloted the Global Urban Soil Ecology and Education Network (GLUSEEN). When fully implemented, GLUSEEN will be a distributed network that builds upon a worldwide set of decomposition experiments using nylon-mesh teabags sited in various urban soil habitat types. As an open and distributed experimental network focused on urban and exurban areas, GLUSEEN will have both scientific and public participatory advantages. Additionally, a matrix of urban soil habitat types based on anthropogenic disturbances and management regimes is presented. The matrix provides an experimental framework to address the Network's goal of comparing soil decomposition, biota, and characteristics across and within urban regions at multiple scales. Questions addressed include: (1) What is the relative importance of native vs. anthropogenic factors on soil characteristics? (2) How do assembly rules of soil communities differ in urban habitats, and how does this translate to ecological functions? (3) Do urban soil ecosystem attributes converge and soil communities homogenize at global and regional scales? (4) How can observations of ecological structure and function of urban soils by citizen scientists advance our understanding of soil ecology? As a proof of concept, we tested and demonstrated the practicality of nylon mesh teabags to measure decomposition between two soil habitat types exhibiting differences in soil abiotic and biotic factors over a 6-month period. Additionally, we illustrate the usefulness of the soil habitat matrix using published data that compared soil characteristics across five cities in four different habitat types.

**Key words:** citizen science, decomposition, experimental network, multi-city comparison, soil biodiversity, urban soils

## Introduction

To both study urban soil ecological systems and to ultimately engage citizen scientists, we have recently piloted a Global Urban Soil Ecology and Education Network (GLUSEEN), which when fully implemented will be a worldwide, multi-city comparison investigating the effects of human activity and urban environments on decomposition and community structure of soil biota. GLUSEEN is designed as an open and distributed experimental network (Craine et al. 2007; Hanson 2007) that will build upon a worldwide set of observations and experiments following the same protocols. The universal observation of GLUSEEN is the measurement of decomposition, in which a substrate is held constant and incubated in soil of several urban habitat types, similar in concept to a global soil decomposition experiment described in Wall et al. (2008). The set of observations will consist of a multi-tiered approach that vary in complexity, cost and applicability to citizen science from relatively simple measurements of decomposition to more complex measurements that include observations and assessments of soil biota and other soil factors.

As an open and distributed experimental network, GLUSEEN will have both scientific and social participatory advantages, especially since the network will be juxtaposed with densely populated urban landscapes. From a scientific perspective, distributed experimental networks provide high statistical power over multiple geographic locations because of the high number of observations as long as protocols, site selection and other experimental factors are held constant (Fraser et al. 2013). In turn, high statistical power provides the opportunity to address broad, but central questions in soil ecology (e.g. Wall et al. 2008). Moreover, if the sites are organized across environmental gradients or state factor sequences, investigators have the potential to relate response variables to environmental or human factors (Craine et al. 2007; Pouyat et al. 2010). From a social perspective, open distributed networks enable participants as a 'community of interest' through their participation in the governance, data collection, data synthesis and refinement or development of hypotheses (Hanson 2007).

The fact that worldwide more than 50% of the human population live in urban areas (United Nations 2014) and that these populations are largely disconnected from nature (Miller 2005), the participation of citizen scientists in an urban soil network represents an exciting opportunity to reconnect people with the ecosystems they live in and to learn about the services ecosystems provide (Henderson 2012; Soga and Gaston 2016). Indeed, an often neglected component of urban ecosystems is the soil and the organisms that inhabit it (Setälä et al. 2014). Thus, urban soils have great potential to serve as *in situ* laboratories for students and residents to learn about ecological systems (Johnson and Catley 2009).

Here, we present the scientific questions of GLUSEEN and provide a proof of concept for the study design and our most simple and universal protocol to measure decomposition using commercially available nylon mesh teabags as standardized litter bags. The methodology, introduced by Keuskamp et al. (2013), is to quantify the mass loss of tea leaves (a surrogate for plant detritus) over time to measure decomposition rate—a protocol that is relatively simple to use—and thus we feel is within the skill set of citizen scientists, grade-school teachers and students. The teabag protocol will be the simplest measure of decomposition in a multi-tiered approach that will also include more sophisticated protocols for questions related to varying quality of detritus as well as chemical changes of detritus as it decays through time and under varying environmental conditions. In addition, we present and assess a matrix of urban soil habitat types based on human disturbance and management regimes. The habitat types provide an experimental framework for GLUSEEN to address the Network's basic goal of comparing decomposition, soil biota and soil characteristics across and within urban regions at local, regional and global scales.

## Questions addressed by GLUSEEN

Urban soils provide many of the same ecosystem processes and functions as 'natural' or agricultural soils, e.g. decomposition and nutrient cycling, carbon sequestration, water purification and regulation, medium for plant growth and habitat for an enormous diversity of organisms (Pavao-Zuckerman 2013; Setälä et al. 2014). A particularly important service provided by soil biota is the decomposition of organic matter. Previous global comparisons of the decay of a reference substrate have yielded important insights into the control of decomposition, particularly with respect to climate and the importance of local soil fauna (Wall et al. 2008). Similar comparisons of decomposition rate across urban ecosystems and their native counterparts should provide insights into the control of decay processes with respect to land-use change (Yesilonis and Pouyat 2012), but also how changes in land use may interact with a changing global climate (Carreiro and Tripler 2005). Therefore, by exploiting impacts and environmental changes associated with urban land-uses, GLUSEEN will address the following questions related to anthropogenic effects on decomposition and soils:

i. What is the relative importance of native (e.g. climate and parent material) vs. anthropogenic (e.g. management and disturbance) factors on soil characteristics?

Soils in urban landscapes vary widely in their character as a result of anthropogenic factors that are introduced by human settlement. The resultant patchwork of parcels, or 'mosaic' of soil conditions, is representative of both direct and indirect effects that occur during and after urban development (Pouyat et al. 2010). Examples of direct effects include physical disturbances such as cut-and-fill practices used in urban development (McGuire 2004; Trammell et al. 2011), management supplements such as irrigation and fertilization (Law et al. 2004; Tenenbaum et al. 2006; Zhu et al. 2006), land cover alteration (Byrne et al. 2008) and compaction through trampling (Godefroid and Koedam 2004). Indirect effects include environmental changes such as the urban heat island effect (Brazel et al. 2000; Savva et al. 2010; Hall et al. 2015), atmospheric deposition (Lovett et al. 2000; Rao et al. 2014; Huang et al. 2015) and changes in plant and animal species composition (Ehrenfeld 2003; McKinney 2006; Aronson et al. 2014). Initial investigations of urban soils focused mainly on highly disturbed soils with no visible structure, low organic matter and having been contaminated with trace metals or other toxic compounds (Craul and Klein 1980; Patterson et al. 1980; Short et al. 1986; Jim 1993). More recently, investigations have shown a wide range of soil conditions, which in some cases are actually more favourable for plant growth than the pre-existing native soil (Hope et al. 2005; Pouyat et al. 2007a; Davies and Hall 2010; Edmondson et al. 2012).

The wide range of conditions in the urban soil mosaic can be utilized as a suite of 'natural experiments' from which to study urban soils (Pouyat et al. 2009). This approach is possible because in densely populated areas humans parcel land based on ownership boundaries, which often overlap with anthropogenic effects on urban landscapes. Thus, urban soils can be delineated based on their management regime (Hope et al. 2005; Zhu et al. 2006) or maintained cover (Byrne et al. 2008; Yesilonis et al. 2016), level of use and disturbance (Pouyat et al. 2007b) and time since disturbance or site history (Raciti et al. 2011). These delineated patches can then act as field manipulations or natural experiments where their comparisons can address the relative importance of anthropogenic and native factors on soil characteristics (Pouyat et al. 2010). Additionally, comparisons between remnant patches embedded within an urban context can occur with a reference patch of similar cover and soil type outside the urban area to assess urban environmental effects on decomposition and soil characteristics (Pouyat et al. 2009).

ii. How do assembly rules of soil communities differ in novel urban habitats, and how does this translate to ecological functions?

Soil community structure and function reflect both natural and human disturbance and stress. For example, forest harvesting practices, cultivation and urbanization can dramatically alter the species composition of soil biota (Setälä et al. 2000; Birkhofer et al. 2008). Previous assessments of human impacts on soil function and community structure have mainly focused on agricultural, and to a lesser extent on managed forested areas. Much less is known on the structure and function of soil communities in urban and peri-urban areas and whether effects of urbanization are similar across regional, continental and global scales (Pouyat et al. 2010).

In urban landscapes, habitats for soil organisms are extremely patchy with a wide range of conditions (Pouyat et al. 2010). In addition to the direct and indirect effects mentioned above, soil is often transported to augment construction sites or planting beds, which unintentionally introduces soil organisms. Additionally, management activities such as pesticide use, composting, mulching and irrigation may enhance or eliminate populations of soil organisms (Byrne et al. 2008). This ‘facilitated assembly,’ in combination with environmental filtering and dispersal abilities, or ‘self-assembly,’ is thought to determine local community composition in plants and other taxa (Swan et al. 2011). In the case of soil organisms where dispersal abilities vary considerably, but for most organisms are limited, it is less likely that urban soil communities become saturated with species. Such communities are loosely packed and thus the interactions between taxa less common than in systems where high species packing is the norm. In these loosely packed systems, the loss of a species would be functionally more pronounced in comparison to a species packed system where the loss of species would be compensated by other species, which is typically the case with non-anthropogenic soils. Consequently, disturbed and managed urban soils should represent soil communities in which species diversity plays a larger role in ecosystem function than in natural soils (Setälä et al. 2005).

Linking structural changes to functional differences is one of the greatest challenges of soil ecology. Ecosystems in general and soil systems in particular are considered to be functionally redundant by many authors (e.g. Lawton and Brown 1994; Andrén et al. 1995; Setälä et al. 2005). This suggests that it is biomass rather than species number or community composition that controls ecosystem process rates. Some species of soil biota, however, are known to be functionally more influential than others (Lavelle et al. 2006). Such key species may be found in various trophic groups of soil organisms although more research is needed to test this hypothesis. Ultimately, comparisons among major functional groups can be related to decomposition rate and soil nutrient cycling processes among soil habitat types found in urban landscapes, and thus advance our understanding of the relationship of ecological structure and function.

iii. Do soil ecosystem attributes ‘converge’ and soil communities ‘homogenize’ at global and regional scales in the urban setting (Pouyat et al. 2003; McKinney 2006; Pouyat et al. 2010; Groffman et al. 2014)?

The Urban Ecosystem Convergence Hypothesis (UECH) states that urbanization drives the structure and function of native ecosystems (e.g. soil carbon stocks, leaf area index) over time toward a range of similar endpoints regardless of the prevailing climate and other local factors (Pouyat et al. 2003). The hypothesis suggests that at regional and global scales, soil characteristics and soil community structure will be more similar in disturbed and managed soils than in the native soils that these urban soils replaced. This convergence in characteristics of urban impacted soils relative to native soils is due in part to differences in climate and native soil factors that typically occur at regional and global scales, but less so for urban environments, e.g. air temperature (Hall et al. 2015). More fundamentally, it is due to (1) efforts by people to overcome the presence of any site limitations to biogenic processes, such as irrigation to support plant primary production; (2) the relatively consistent effect of human-caused physical disturbances to soil structure, porosity and plant cover; and (3) the overwhelming effects resulting from the built environment such as the sealing of soils with impervious surfaces (Pouyat et al. 2007b). Thus in a global comparison of urban soils, the difference between native and anthropogenic soil should be greatest for cities located in biomes with the greatest limitations on NPP or decomposition, such as semi-arid grasslands or boreal forests, and for those cities with native factors (e.g. calcareous parent material) that disproportionately affect soil development (Pouyat et al. 2010, 2015). Therefore, those soil characteristics that are strongly related to biological processes, e.g. soil organic carbon (Hope et al. 2005), should respond differently than properties that are strongly influenced by parent material, e.g. texture (Pouyat et al. 2007a).

Similar to the convergence of soil physical and chemical characteristics and soil process rates, Biotic Homogenization is hypothesized to be occurring as a result of urban land-use change (McKinney 2006; Groffman et al. 2014). Homogenization, however, has been mostly tested for relatively large organisms with straightforward taxonomy, such as plants and birds (Olden et al. 2006; Schwartz et al. 2006 Smart et al. 2006; Aronson et al. 2014). Therefore, GLUSEEN represents an opportunity to test the importance of urban land-use change in the homogenization of smaller soil organisms across global scales. For those soil invertebrate species that have been unintentionally or deliberately carried across continents, the expectation is that they will be cosmopolitan in their distribution, which appears to be the case with some earthworm species (Hendrix et al. 2008; Blakemore 2009). Moreover, their local and regional distributions can be tested with comparisons between exurban and urban site patches. Species residing only in the latter may lead to further inquiries on their adaptedness to urban environments (McDonnell and Hahs 2015). For soil microbial communities, the impacts of urban land-use are less clear but early results suggest that there may be as much phylotype diversity in a large urban park as has been globally observed (Ramirez et al. 2014), which is partly the result of the diversity of habitats found in urban landscapes (Szlavecz et al. 2011).

iv. How can observations of ecological structure and function of urban soils by students and citizen scientists advance our understanding of soil ecology?

Due to the close proximity to more than half of the global human population, urban soils have tremendous potential to inform students and residents about the importance of biodiversity and ecosystem function (Johnson and Catley 2009; Makarushka 2012). For instance, most people are not aware of the enormous biotic diversity that exists in soils found in their backyards, school grounds, park green spaces and vacant lots. Many soil animals are easy to culture in the laboratory and manipulate to conduct experiments in the classroom. This ease of experimentation allows for addressing basic questions on soil community and food web dynamics (Moore et al. 2000). As an example, earthworms, which are often invasive and found in urban landscapes (Steinberg et al. 1997), provide an opportunity to present ecological concepts on keystone species or the importance of ecological engineers (Szlavecz et al. 2011). Additionally, the role of soil biota in decomposition and nutrient cycling is closely tied to soil health, an increasingly important issue in urban agriculture and food security (Zezza and Tasciotti 2010; Grewal et al. 2011). Urban agriculture in turn raises public awareness about the health of a soil (Brown et al. 2016) and can provide, through hands-on-experience, horticultural and marketable skills for inner city youth (Brown and Jameton 2000).

Beyond having close proximity to people, urban ecological systems embody various human disturbances and environmental changes, which represent unique opportunities for exploring 'whole ecosystem' manipulations occurring within the urban soil mosaic (Pouyat et al. 2009). These manipulations consist of various anthropogenic factors such as those analogous to climate change (Ziska et al. 2004; Carreiro and Tripler 2005), pollution effects (Huang et al. 2015), habitat fragmentation (McKinney 2006) or unique assemblages of species (Hobbs et al. 2006; Swan et al. 2011), among others. As a result, the study of urban soil ecological systems should enhance our understanding of overall ecological structural and functional responses to disturbance, environmental stress and species dispersal. By utilizing a network of study sites across many metropolitan areas that vary in development pattern, cultural and economic factors and regional climatic conditions, we can assess with relatively high statistical power the generality of the response across a highly diverse set of socio-ecological conditions. Additionally, the utilization of a multi-tiered approach to measure soil abiotic and biotic characteristics will enable the engagement of students and citizen scientists with a variety of backgrounds, expertise and interest. Therefore, the inherent juxtaposition of dense human populations with soil in urban landscapes provides both an opportunity for student and citizen engagement with soil science, and the acquisition of data across many types of urban soils and at multiple scales.

## Network study design and measurement of decomposition

GLUSEEN is based on a suite of similar observations, measurements and experiments carried out in several habitat types (described below) associated with urban and urbanizing landscapes on a global scale. In addition to the network of urban sites, at each location a native habitat type peripheral to the urban area serves as a 'reference'" for the urban sites, allowing for both a local and global comparison across urban and native habitats. The universal measurement of the network, decomposition, will vary in complexity and cost from very simple comparisons of a standard substrate and its mass loss over time, to more complex comparisons, such as chemical changes of the substrate as it decays and soil organism composition and abundance measurements.

As a minimum requirement for inclusion in the network, and the least expensive and simplest measurement for decomposition, we use nylon-meshed teabags (Keuskamp et al. 2013) as a substitute for the more commonly used litterbags (Wieder and Lang 1982). The litterbag approach is widely used to study decomposition at the soil surface. Leaf litter is enclosed in hand-made permeable bags and collected at periodic intervals for measurement of the mass remaining, which is a function of the decomposition rate. The use of pre-prepared teabags is more cost-effective and greatly reduces the number of steps and measurements that are typically required in the preparation of litterbags (Keuskamp et al. 2013), and thus more conducive for use in teaching and by citizen scientists (e.g. Teatime 4 Science, http://www.teatime4science.org/). More specific information on the teabag protocol can be found in Supplement 2.

We utilize an experimental matrix based on two disturbance and three management intensities that provides a typology of six urban soil habitat types that can be identified in most urban landscapes (Table 1). The soil habitat types approximately correspond to a continuum of anthropogenic effects from relatively low influences (native) to those somewhat impacted by urban environmental effects such as remnant forest patches (Pouyat et al. 2009), to types that are highly altered by physical disturbances and management (Fig. 1). The latter include massive, or highly disturbed soils without structure (e.g. Short et al. 1986), engineered soils such as green roof media and street tree pit soils (e.g. Grabosky et al. 2002; Bartens et al. 2010), and soils that were once drastically disturbed, but are now managed, such as public or residential lawns (e.g. Raciti et al. 2011). Intermediate in anthropogenic effects are habitat types that are managed but experience relatively low disturbance such as perennial gardens (e.g. Edmondson et al. 2014). Thus, comparisons among the soil habitat types should reflect the impact of physical site disturbances (e.g. site grading), subsequent management activities (e.g. fertilization, irrigation), intensity of use (e.g. trampling), and plant cover and urban environmental changes (e.g. air pollution, habitat press) on soil response variables. Moreover, making these comparisons at a global scale across many biomes should reveal generalities of soil responses to urban land-uses.

### Proof of concept study design and protocols

Meetings were held in 2013 and 2014 at the University of Helsinki in Lahti and at the Johns Hopkins University in Baltimore to discuss and develop an experimental design and protocols for the network and criteria to be used for site selection. For the preliminary proof of concept study, 20 sites per city (five replicates for each of the four soil habitat types) were identified and cross verified using photographs by sampling teams in each biome. The sites were located to represent the soil habitat type based on surface inspection. One rectangular plot with dimensions of $1.5 \times 2$ m was established per replicate site and sampled using a 50 cm grid (see Supplements 1 and 2 for specific sampling protocols). The soil characteristic results of the proof of concept study have been published (Pouyat et al. 2015), data on soil microbial community composition are in revision (Schmidt et al. in press), and data on decomposition are currently being analysed. As additional protocols are developed

**Table 1.** Matrix of soil habitats found in urban landscapes that are based on management intensity and disturbance regime[a]

| Disturbance | Management | | |
|---|---|---|---|
| | Low (LM) | Medium (MM) | High (HM) |
| High (HD) | **Fill soil, vacant lot (ruderal)** | **Lawn, public greenspace (turfgrass)** | Golf greens, urban agriculture, green roofs (engineered) |
| Low (LD) | **Woodland, Grassland (remnant)** | Managed woodland (managed remnant) | Perennial Gardens (horticulture) |

[a]Soil habitat types used in the pilot study are in bold face (Reference type is not included in the matrix).

**Figure 1.** Field photographs of reference and urban habitats used in the GLUSEEN pilot study. A: reference (coniferous forest), Helsinki; B: reference (grassland), Potchefstroom; C: reference (deciduous forest), Budapest; D: remnant (deciduous forest), Baltimore; E: turf, Lahti; F: ruderal, Potchefstroom. See table 1 for description of habitat types. (Photograph credits: E. Hornung, D.J. Kotze, E. Powell, H. Setälä).

and tested, they will be made available on the GLUSEEN website (see below).

As a proof of concept, we initially compared five soil characteristics across five different cities that range in climate from boreal-hemiboreal (Helsinki and Lahti, Finland) to humid-subtropical (Baltimore, USA), to continental (Budapest, Hungary) and to semiarid (Potchefstroom, South Africa) biomes. Therefore the climate and geographical location of these cities represents a broad range of precipitation and temperature, development pattern, population and social factors (Table 2). In each city, three of the six habitat types were sampled: remnant (deciduous or coniferous forest, grassland), ruderal (massive or fill soil) and turfgrass (public greenspace), in addition to a reference site that corresponds to the remnant type. Thus, based on the matrix described in Table 1, these habitat types roughly conformed to a continuum of anthropogenic influence from highest to lowest, or ruderal > turfgrass > remnant > reference types where differences between ruderal and turfgrass represent a management vs. disturbance effect and differences between remnant and reference represent an urban environmental effect (Fig. 1).

## Assessment of the matrix of soil habitat types

As described earlier, three of the four core questions of GLUSEEN relate to how urban soils vary and compare with the native soils they have replaced, how soil biota respond to these changes, and how urban soil characteristics converge and

**Table 2.** Biomes, soil moisture and temperature regimes, soil order and primary parent material of the initial five cities in the Global Urban Soil Ecology and Education Network (GLUSEEN)[a]

| Cities (Population) | Biome | Moisture Regime | Temperature Regime | Soil order | Parent material |
|---|---|---|---|---|---|
| Baltimore, USA (621 000) | Humid-subtropical | Udic | Mesic | Ultisol | Mafic rock |
| Budapest, Hungary (1 728 000) | Continental | Ustic | Mesic | Alfisol (Leptosol) | Dolomite |
| Helsinki, Finland (1 400 000) | Boreal-hemiboreal | Udic | Mesic | Spodosol | Granite |
| Lahti, Finland (102 000) | Boreal-hemiboreal | Udic | Mesic-Cryic | Spodosol | Granite/till |
| Potchefstroom, South Africa (250 000) | Semiarid | Aridic | Thermic | Aridisol | Shale/diabase |

[a]Population of each city in parentheses.

species assemblages homogenize across continental and global scales relative to native soils. For question 1, the comparison of a reference and remnant type in the same metropolitan area assesses soil responses of a forest or grassland growing under urban environmental conditions (air pollution, heat island effect, etc.). In a similar fashion, comparisons between a public greenspace in turfgrass cover and a ruderal site assesses the effect of lawn management or prior major disturbance. Whereas for question 3, comparisons of habitat types across cities, and therefore biomes, would be a means to test the UECH. For example, a comparison of native habitat types across different biomes should yield a large variation in soil organic carbon—a soil property that is highly affected by biogenic activity and therefore climate factors. Similarly, the generality of management or disturbance effects on soil organic carbon can be assessed with comparisons of native with turfgrass and ruderal sites.

To address questions 1 and 3 and to show the operational utility and scientific robustness of the matrix of six soil habitat types, we summarize results on soil measurements reported in a previous paper by Pouyat et al. (2015). In this pilot study, we compared measurements for total carbon (TC) and nitrogen (TN), available phosphorus (P) and potassium (K), and pH among reference, remnant, turfgrass and ruderal soil habitat types in each of five cities (Table 2). Specifically, we compared the coefficient of variation (CV) of each soil property among the cities for each habitat type. A higher CV suggests less similarity, while a lower CV suggests greater similarity. Therefore, if a soil property were to have a higher CV for native vs. urban habitat type, we would interpret this as a 'convergence,' or that urban soil types are more similar across biomes than native soils (Pouyat et al. 2015).

Results of the five city comparison showed that CVs for soil pH, OC and TN indeed exhibited a convergence across a continuum represented by the four soil habitat types, i.e. CVs ranked in the order of reference > remnant > turfgrass ≥ ruderal types. For soil characteristics highly impacted by management activities or disturbance, such as OC and TN, we expected these results. Whereas for soil pH we suggest the convergence was the result of the interaction of both anthropogenic (e.g. calcium carbonate from building materials) and native (e.g. calcareous parent material) soil factors. By contrast, the CV for soil K and P was higher for the ruderal and turfgrass than the reference soil habitat types and thus exhibited a 'divergence,' which was an unexpected result (see Pouyat et al. 2015 for more discussion of these results).

Thus, the typology of soil habitat types categorized from a matrix of management and disturbance worked well for conducting multiple-scale comparisons of urban soil properties and in addressing two of the Network's cores questions. In particular, the ability to compare soil characteristics among habitat types within a city and corresponding native soil, and among different cities or biomes, made it possible to assess the relative importance of urban environmental changes (temperature, pollution) and human activities (management vs. disturbance) of soils located in urban landscapes. Moreover, the matrix of six soil habitat types provided the flexibility to create a sampling design across a diversity of urbanized landscapes (for the pilot study we chose three). This flexibility is particularly important for an open distributed network like GLUSEEN that encourages use by teachers, students and citizen scientists who may not have access to all of the soil habitat types in the matrix.

## Assessment of teabag protocol

Keuskamp et al. (2013) were the first to use commercially available nylon mesh, pyramid-shaped teabags to measure decomposition in various habitat types across the globe. Their study showed that the method can be applied to differentiate rates of mass loss in tea leaves among a wide range of ecosystems in the relatively brief incubation period of 90 days. A large fraction of tea leaves is water soluble, however, and thus it is likely that a high proportion of the initial mass loss reported by Keuskamp et al. (2013) was due to an abiotic process (leaching) rather than the mass lost from biologically mediated decomposition. Therefore, we modified the authors' protocol for using teabags so that the comparisons being made will more reflect biological differences across sites (Supplement 2).

Additionally, Keuskamp et al. (2013) derived a 'teabag index' by mathematically combining the decay curves of both types of tea, with the green tea used to determine the decay constant and the red tea (rooibos) the stabilization factor, thus making it possible to interpret a decomposition curve using both types of teabags. The authors successfully applied the index to a wide set of habitat types. In our analysis, we did not consider the index, since it introduces a complexity to the protocol that is not conducive for citizen scientists and requires the use of two different types of teabags. We do, however, recognize its potential for comparing decomposition and stabilization rates and associated controlling factors in urban soils across many different habitat types and biomes and thus may later employ the method as a higher tiered protocol.

An initial trial was done in the laboratory to test the effectiveness of the teabags comparing two reference soil habitat types that represented different soil communities—a deciduous vs. a coniferous forest soil (Supplement 3). Both teabag types lost mass during the initial soaking periods in the laboratory, with the greatest loss occurring in the green teabags, 0.85 compared to 0.60 g, or 40 and 27% loss in mass, respectively (Fig. 2). This difference is consistent with Keuskamp et al. (2013) who measured a water soluble fraction of 49 and 22% for the green and red teabags, respectively. The red teabags consistently lost mass over

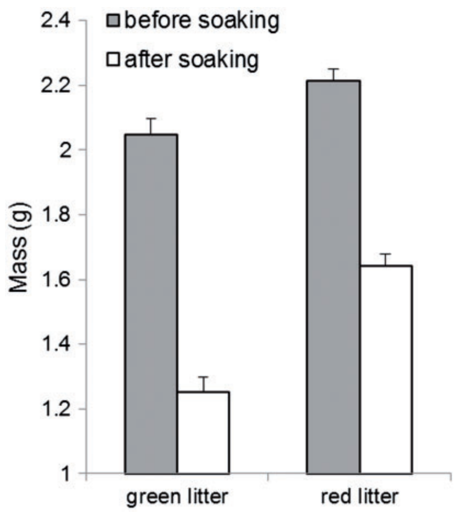


**Figure 2.** Mass (mean ± SD) of green and red tea before and after soaking in hot tap-water.

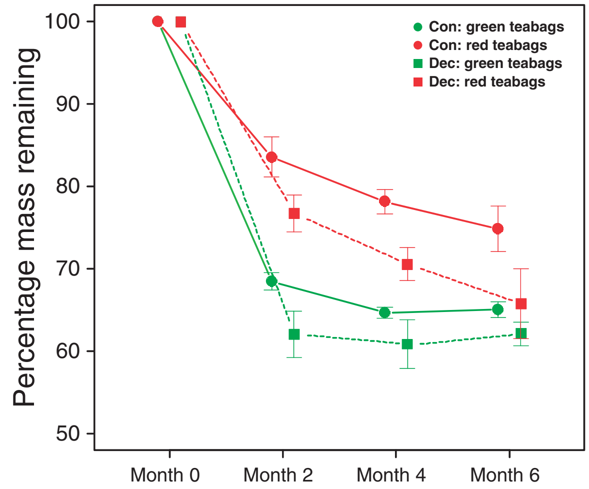


**Figure 3.** Percentage mass loss (mean ± SD) of soaked green and red teabags at 2-month intervals incubated at 15 °C. Con = coniferous forest soil, Dec = deciduous forest soil.

the 6-month incubation period while the green teabags lost most of their mass by the end of 2 months. The greatest loss in mass for both teabag types occurred in soil from deciduous forests (Fig. 3). In addition, the green teabags had a significantly greater loss in mass than the red teabags over the entire 6-month incubation period (two-way ANOVA; tea type, $F = 22.84$, $P < 0.001$; forest type, $F = 24.78$, $P < 0.001$; interaction, $F = 5.05$, $P = 0.04$). By the end of the 6-month incubation, green teabags in the deciduous soil lost approximately 38% compared to 24% mass by the red teabags in the coniferous soil treatment (Fig. 3).

The overall loss in mass by the red and less so the green teabags is comparable to laboratory incubations (15°C) of the same brand of teabags used by Keuskamp et al. (2013), who found a more rapid loss of mass in the green teabags (bottoming out after 20–40 days) than in our study (>60 days) and a slightly less rapid loss in mass of the red teabags after 60 and 120 days. By the end of their incubation period (120 days) the authors found an almost two-fold higher loss in mass for the green than in the red teabags, 40 vs. 75%, compared to 62 vs. 72%, for green and red teabags incubating in soil from deciduous forests in our study. After 240 days (not reported in Keuskamp et al. 2013) the overall mass loss for both green and red teabags in soil from deciduous forests bottomed out at 62 and 65%, respectively (Fig. 3). The difference in green teabag mass loss in Keuskamp et al. (2013) and our study can best be explained by our pre-leaching of the teabags prior to incubation. The green teabags lost significantly more mass after leaching than the red bags (Fig. 2). This result suggests the green teabags had more mass to lose without prior leaching, particularly in the first weeks of the incubation period. However, after 240 days, the overall mass lost in the deciduous soil was only slightly greater for green than for red teas.

The teabag incubations in our and Keuskamp et al.'s study suggest that litter (tealeaves) in red teabags are more recalcitrant to soil microbial decay than the green teabags. In fact, Keuskamp et al. (2013) found a large difference in carbon to nitrogen ratios between the green and red tea types of 12.2 ± 0.1 and 42.9 ± 1.8, respectively. Additionally, in our study both teabag types exhibited sensitivity to the differences in biota of the deciduous and coniferous soil types (Fig. 3). The coniferous soil was more acidic and likely dominated by a fungal microbial community, while the deciduous soil had a relatively high pH that was likely to be dominated by a bacterial microbial

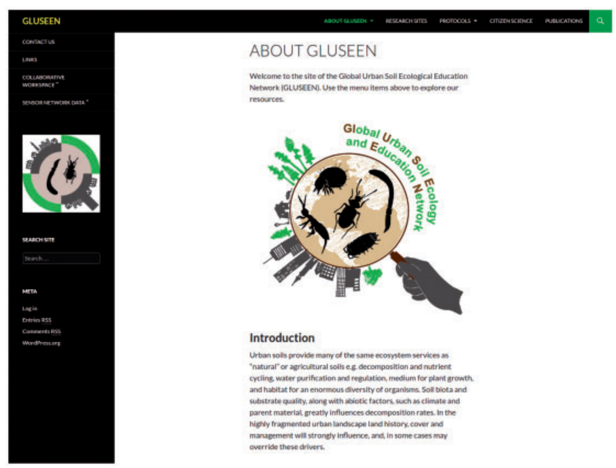


**Figure 4.** Home page for the GLUSEEN project (www.gluseen.org). Information about network members, research sites, protocols (site selection, soil sampling, teabags and earthworm protocols currently available), and a list of publications will be made available. (Logo credit: Marié Du Toit).

community (Ingham et al. 1989). Furthermore, even after leaching the bags, and thus an initial loss in water soluble constituents (up to 40% in the green teabags), the differences in decomposition were measurable within a 6-month period, which represents a tangible short-term result that coincides with most school semesters as well as within the timespan of citizen science projects.

## Data sharing and collaboration

The distributed nature of the GLUSEEN project will require innovative collaborative tools to allow participants (scientists, teachers, students and citizen scientists) to share ideas, data and research results (http://www.gluseen.org/) (Fig. 4). The Collaborative Network for Urban Soil Ecology (http://elgg.gluseen.org) is in development to give participants of the GLUSEEN network the opportunity to share ideas and collaborate using data collected across a number of sites using a web-based interface.

The collaborative website is using the Elgg framework (www.elgg.org), which provides a number of facilities for users that are common to social networking sites such as profile management, group creation, uploading and sharing of photos, blogging, posting ideas and commenting on shared posts and media. When fully operational, GLUSEEN participants will be able to use the social network features to support investigations at a single site, across all sites, or a subset of sites with a common element such as a soil habitat type or field manipulation. Therefore the collaborative website will enable a wide variety of ways for participants to fit their study sites into the network.

The collaborative network will also include novel features that allow participants to interactively integrate, analyze and visualize data across a number of dimensions. The data infrastructure, based on the SciServer architecture (http://www.sciserver.org/), will provide a centralized database whereby users can, with relative ease, upload their site and experimental data (Fig. 5). Additionally, all data in the database are exposed through a web services interface where participants can query data using standard SQL or by using a query interface being developed for the collaborative network. Participants will also be able to visualize the data (e.g. Fig. 6) using an interface developed using d3.js (http://www.d3js.org).

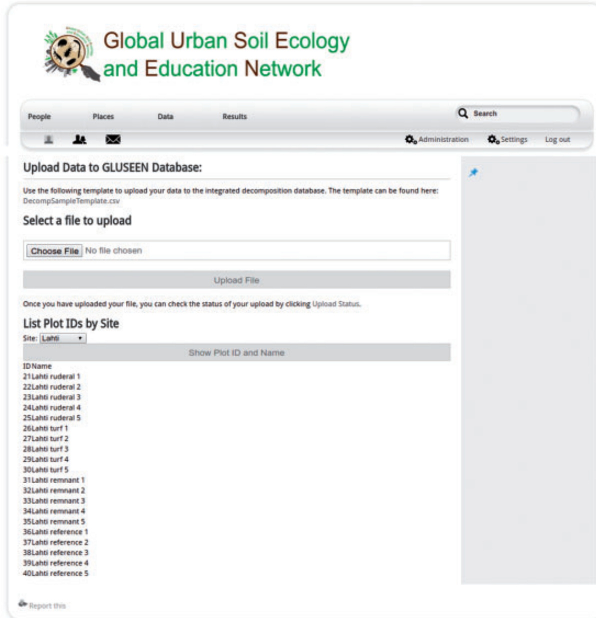


**Figure 5.** Data upload interface for the collaborative website. Users upload a csv template of their data and it is incorporated in the GLUSEEN database.

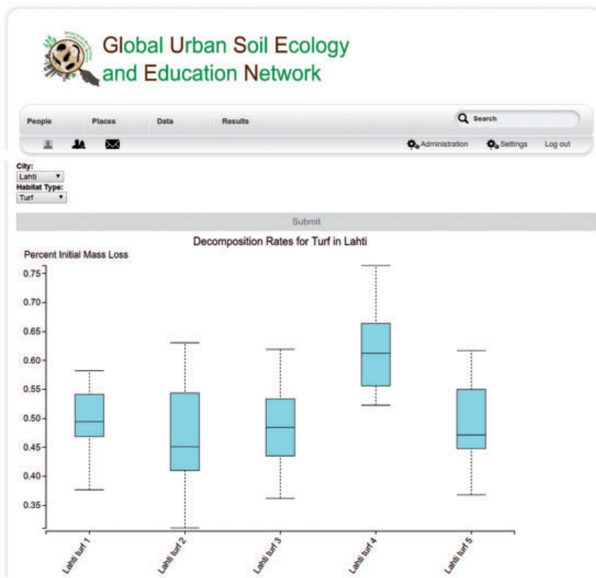


**Figure 6.** Example of data visualization on the GLUSEEN collaborative website. From the database Lahti, Finland was selected as city, and turf was selected as habitat. Boxplots show teabag mass loss after 6 months.

## Conclusions

We introduced GLUSEEN, an emerging open access and experimental network in urban soil ecology, and report on a proof of concept for measuring (1) differences in soil biological and chemical characteristics among various urban soil habitat types and (2) decomposition rates between two types of forest soil using commercially available teabags as a substitute for litterbags. We feel teabags are more accessible, suitable and cost-effective than litterbags for use in grade schools, urban citizen science programs, and open dispersed networks such as GLUSEEN. The typology used for urban soil habitat types thus far appear to represent anthropogenic impacts on soils across cities embedded within very different biomes. Previous reported research showed that soil habitat types representing the most intense anthropogenic factors (urban park greenspaces and ruderal or highly disturbed soils) were more similar in soil pH, TN and TC across four biomes than the native soils sampled in each biome, largely supporting the UECH. In addition, over a 4-–6-month period the pre-leached teabags exhibited a sensitivity to differences in soil biota between two reference forest soil types (deciduous and coniferous) found in the boreal-hemiboreal biome, suggesting that teabags are suitable for testing differences in decomposition across at least reference soil habitat types in the network. The type of teabags, however, must be kept constant in making these comparisons.

## Acknowledgements

The idea for GLUSEEN was made possible by a Fulbright Specialist grant to R. Pouyat at the University of Helsinki. The GLUSEEN network is supported by a supplemental grant to NSF-ACI 1244820. Soil analyses were supported by the FEKUT grant (SZIE-AOTK-KK-UK 12007) of the Hungarian Ministry of Human Resources (E.H.). The National Research Foundation (NRF) of South Africa provided financial assistance for field and laboratory research in Potchefstroom. The database development is funded by the SciServer project (NSF-ACI 1261715). We thank M. Bernard, A. Dec, E. Draskovits, S. Mishra, S. Molnar, E. Powell, J. Raddick, B. Souter and Z. Toth for their efforts in support of this study. The use of trade, firm, or corporation names in this publication is for the information and convenience of the reader. Such use does not constitute an official endorsement or approval by the U.S. Department of Agriculture or the Forest Service of any product or service to the exclusion of others that may be suitable.

## Supplementary data

Supplementary data is available at Journal of Urban Ecology online.

*Conflict of interest statement*. None declared.

## References


Andrén, O., Bengstsson, J., and Clarholm, M. (1995) 'Biodiversity and species redundancy among litter decomposers', in *The Significance and Regulation of Soil Biodiversity Volume 63 of the series Developments in Plant and Soil Sciences*, pp. 141–151. *Proceedings of the International Symposium on Soil Biodiversity*, held at Michigan State University, East Lansing, May 3–6, 1993.

Aronson, M. F. J. et al. (2014) 'A global analysis of the impacts of urbanization on bird and plant diversity reveals key anthropogenic drivers', *Proceedings of the Royal Society of London, Series B*, **281(**1780**)**: 1471–2954.

Bartens, J., Wiseman, P. E., and Smiley, E. T. (2010) 'Stability of Landscape Trees in Engineered and Conventional Urban Soil Mixes', *Urban Forestry & Urban Greening*, **9**: 333–8.

Birkhofer, K. et al. (2008) 'Long-Term Organic Farming Fosters Below and Aboveground Biota: Implications for Soil Quality, Biological Control and Productivity', *Soil Biology & Biochemistry*, **40**: 2297–308.

Blakemore, R. J. (2009) 'Cosmopolitan Earthworms: A Global and Historical Perspective', in Shain D.H. (ed) *Annelids in Modern Biology*, pp. 257–83, Hoboken, NJ: Wiley-Blackwell.

Brazel, A. et al. (2000) 'The Tale of Two Climates—Baltimore and Phoenix Urban LTER Sites', *Climate Research*, **15**: 123–35.

Brown, K. H., and Jameton, A. L. (2000) 'Public Health Implications of Urban Agriculture', *Journal of Public Health Policy*, **21**: 20–39.

Brown, S. L., Chaney, R. L., and Hettiarachchi, G. M. (2016) 'Lead in Urban Soils: A Real or Perceived Concern for Urban Agriculture?', *Journal of Environment Quality*, **45(1)**: 26–36. DOI: 10.2134/jeq2015.07.0376.

Byrne, L. B., Bruns, M. A., and Kim, K. C. (2008) 'Ecosystem Properties of Urban Land Covers at the Aboveground–Belowground Interface', *Ecosystems*, **11**: 1065–77.

Carreiro, M. M., and Tripler, C. E. (2005) 'Forest Remnants Along Urban–Rural Gradients: Examining their Potential for Global Change Research', *Ecosystems*, **8**: 568–82.

Craine, J. M. et al. (2007) 'Building EDENs: The Rise of Environmentally Distributed Ecological Networks', *BioScience*, **57**: 45–54.

Craul, P. J., and Klein, C. J. (1980) *Characterization of Streetside Soils of Syracuse*, New York, METRIA, **3**, pp. 88–101.

Edmondson, J. L. et al. (2012) 'Organic Carbon Hidden in Urban Ecosystems', *Scientific Reports*, **2**: 963. DOI: 10.1038.

——— et al. (2014) 'Urban Cultivation in Allotments Maintains Soil Qualities Adversely Affected By Conventional Agriculture', *Journal of Ecology*, **51**: 880–9.

Davies, R., and Hall, S. J. (2010) 'Direct and Indirect Effects of Urbanization on Soil and Plant Nutrients in Desert Ecosystems of the Phoenix Metropolitan Area, Arizona (USA)', *Urban Ecosystems*, **13**: 295–317.

Ehrenfeld, J. G. (2003) 'Effects of Exotic Plant Invasions on Soil Nutrient Cycling Processes', *Ecosystems*, **6**: 503–23.

Fraser, L. H. et al. (2013) 'Coordinated Distributed Experiments: An Emerging Tool for Testing Global Hypotheses in Ecology and Environmental Science', *Frontiers in Ecology and the Environment*, **11**: 147–55.

Godefroid, S., and Koedam, N. (2004) 'The Impact of Forest Paths Upon Adjacent Vegetation: Effects of the Path Surfacing Material on the Species Composition and Soil Compaction', *Biological Conservation*, **119**: 405–19.

Grabosky, J., Bassuk, N., and Marranca, B. Z. (2002) 'Preliminary Findings from Measuring Street Tree Shoot Growth in Two Skeletal Soil Installations Compared to Tree Lawn Plantings', *Journal of Arboriculture*, **28**: 106–8.

Grewal, S. S. et al. (2011) 'An Assessment of Soil Nematode Food Webs and Nutrient Pools in Community Gardens and Vacant Lots in Two Post-Industrial American Cities', *Urban Ecosystems*, **14**: 181–94.

Groffman, P. M. et al. (2014) 'Ecological Homogenization of Urban USA', *Frontiers in Ecology and the Environment*, **12**: 74–81.

Hall, S. J. et al. (2015) 'Convergence of Microclimate in Residential Landscapes Across Diverse Cities in the United States', *Landscape Ecology*, **31**: 101–17.

Hanson, P. C. (2007) 'A Grassroots Approach to Sensor and Science Networks', *Frontiers in Ecology and the Environment*, **5**: 343.

Henderson, S. (2012) 'Citizen Science Comes of Age', *Frontiers in Ecology and the Environment*, **10**: 283.

Hendrix, P. F. et al. (2008) 'Pandora's Box Contained Bait: The Global Problem of Introduced Earthworms', *The Annual Review of Ecology, Evolution, and Systematics*, **39**: 593–613.

Hobbs, R. J. et al. (2006) 'Novel Ecosystems: Theoretical and Management Aspects of the New Ecological World Order', *Global Ecology and Biogeography*, **15**: 1–7.

Hope, D. et al. (2005) 'Spatial Variation in Soil Inorganic Nitrogen Across an Arid Urban Ecosystem', *Urban Ecosystems*, **8**: 251–73.

Huang, J., et al. (2015) 'Urbanization in China Drives Soil Acidification of *Pinus massoniana* Forests', *Scientific Reports*, **5**: 13512.

Ingham, E., Coleman, D., and Moore, J. (1989) 'An Analysis of Food-Web Structure and Function in a Sortgrass Prairie, a Mountain Meadow, and a Lodgepole Pine Forest', *Biology and Fertility of Soils*, **8**: 29–37.

Jim, C. Y. (1993) 'Soil Compaction as a Constraint to Tree Growth in Tropical and Subtropical Urban Habitats', *Environmental Conservation*, **20**: 35–49.

Johnson, E. A., and Catley, K. M. (2009) 'Urban Soil Ecology as a Focal Point for Environmental Education', *Urban Ecosystems*, **12**: 79–3.

Keuskamp, J. A. et al. (2013) 'Tea Bag Index: A Novel Approach to Collect Uniform Decomposition Data Across Ecosystems', *Methods in Ecology and Evolution*, **4**: 1070–5.

Lavelle, P. et al. (2006) 'Soil Invertebrates and Ecosystem Services', *European Journal of Soil Biology*, **42**: S3–S15.

Law, N. L., Band, L. E., and Grove, J. M. (2004) 'Nitrogen Input from Residential Lawn Care Practices in Suburban Watersheds in Baltimore County, MD', *Journal of Environmental Planning and Management*, **47**: 737–55.

Lawton, J. H., and Brown, V. K. (1994) 'Redundancy in Ecosystems', in Schulze E. D., and Mooney H. A. (eds) *Biodiversity and Ecosystem Function*, pp. 255–70, Berlin: Springer.

Lovett, G. M. et al. (2000) 'Atmospheric Deposition to Oak Forests Along an Urban-Rural Gradient', *Environmental Science & Technology*, **34**: 4294–300.

Makarushka, M. (2012) 'Detroit's Vacant Lots Provide 'Natural Laboratory' for Studying Soil Processes', *Soil Horizons*, doi:10.2136/sh2012-53-2-lf.

McDonnell, M. J., and Hahs, A. K. (2015) 'Adaptation and Adaptedness of Organisms to Urban Environments', *Annual Review of Ecology, Evolution, and Systematics*, **46**: 261–80.

McGuire, M. (2004) 'Using DTM and LiDAR data to analyze human induced topographic change', in Proceedings of ASPRS 2004 fall conference, September 12–16, 2004, Kansas City, MO. PDF available at: http://eserv.asprs.org/eseries/source/Orders/

McKinney, M. L. (2006) 'Urbanization as a Major Cause of Biotic Homogenization', *Biological Conservation*, **127**: 247–60.

Miller, J. R. (2005) 'Biodiversity Conservation and the Extinction of Experience', *Trends in Ecology and Evolution*, **20**: 430–4.

Moore, J. C. et al. (2000) '*A Springtail in the Classroom: Folsomia Candida as a Model for Inquiry-Based Laboratories', American Biology Teacher*, **62**: 512–9.

Olden, J. D., Poff, N. L., and McKinney, M. L. (2006) 'Forecasting Faunal and Floral Homogenization Associated with Human Population Geography in North America ', *Biological Conservation*, **127**: 261–71.

Patterson, J. C., Murray, J. J., and Short, J. R. (1980) The impact of urban soils on vegetation. METRIA: 3, Proceedings of the Third Conference of the Metropolitan Tree Improvement Alliance. Rutgers, the State University, New Brunswick, NJ.

Pavao-Zuckerman, M. A. (2013) 'Urbanization, Soils, and Ecosystem Services', in Wall D. H., Bardgett R. D., Behan-Pelletier V., Herrick J. E., Jones T. H., Ritz K., Six J., Strong D. R., and van der Putten W. H. (eds) *Soil Ecology and Ecosystem Services*, pp. 271–81. UK: Oxford University Press.

Pouyat, R. V. et al. (2003) 'Soil Carbon in Urban Forest Ecosystems', in Kimble J. M., Heath L. S., Birdsey R. A., and Lal R. (eds) *The Potential of U.S. Forest Soils to Sequester Carbon and Mitigate the Greenhouse Effect*, Boca Raton, FL: CRC Press.

—— et al. (2007a) 'Soil Chemical and Physical Properties that Differentiate Urban Land-Use and Cover', *Soil Science Society America Journal*, **71**: 1010–9

—— et al. (2007b) 'Urban Land-Use Change Effects on Biogeochemical Cycles', in Canadell J. G., Pataki D. E., and Pitelka L. F. (eds) *Terrestrial Ecosystems in a Changing World. Global Change, The IGBP Series*, pp. 45–58, Berlin-Heidelberg-New York: Springer.

—— et al. (2009) 'Investigative Approaches to Urban Biogeochemical Cycles: New York Metropolitan Area and Baltimore as Case Studies', in McDonnell M. J., Hahs A., and Breuste J. (eds) *Ecology of Cities and Towns: A Comparative Approach*, pp. 329–54, Cambridge, Oxford: Cambridge University Press.

—— et al. (2010) 'Chemical, Physical, and Biological Characteristics of Urban Soils', in Aitkenhead-Peterson J. and Volder A. (eds) *Urban Ecosystem Ecology (Agronomy Monograph 55)*, pp. 119–52, Madison, WI: ASA-CSSA-SSSA.

—— et al. (2015) 'A Global Comparison of Surface Soil Characteristics Across Five Cities: A Test of the Ecosystem Convergence Hypothesis', *Soil Science*, **80**: 136–45.

Raciti, S. M. et al. (2011) 'Accumulation of Carbon and Nitrogen in Residential Soils with Different Land Use Histories', *Ecosystems*, **14**: 287–97.

Ramirez, K. S. et al. (2014) 'Biogeographic Patterns in Below-Ground Diversity in New York City's Central Park are Similar to Those Observed Globally', *Proceedings of the Royal Society, Series B*, **281**: pii: 20141988.

Rao, P. et al. (2014) 'Atmospheric Nitrogen Inputs and Losses Along an Urbanization Gradient from Boston to Harvard Forest, MA', *Biogeochemistry*, **121**: 229–45.

Savva, Y. et al. (2010) 'Land Use and Vegetation Cover Effects on Soil Temperature in an Urban Ecosystem', *Soil Science Society Journal*, **74**: 469–80.

Schmidt, D. J. E. et al. (in press) 'Urbanization Leads to the Loss of Ectomycorrhizal Fungal Diversity and the Convergence of Archaeal and Fungal Soil Communities', *Nature Ecology and Evolution*.

Schwartz, M. W., Thome, J. H., and Viers, J. H. (2006) 'Biotic Homogenization of the California Flora in Urban and Urbanizing Regions', *Biological Conservation*, **127**: 282–91.

Setälä, H., Haimi, J., and Siira-Pietikäinen, A. (2000) 'Sensitivity of Soil Processes in Northern Forest Soils: Are Stand Management Practices A Threat?', *Forest Ecology and Management*, **133**: 5–11.

——, Berg, M. B., and Jones, T. H. (2005) 'Trophic Structure and Functional Redundancy in Soil Communities', in *Biological Diversity and Function in Soils*, 236–249, Cambridge: Cambridge University Press.

—— et al. (2014) 'Urban and Agricultural Soils: Conflicts and Trade-Offs in the Optimization of Ecosystem Services', *Urban Ecosystems*, **17**: 239–53.

Short, J. R. et al. (1986) 'Soils of the Mall in Washington, DC: I. Statistical Summary of Properties', *Soil Science Society of America Journal*, **50**: 699–705.

Smart, S. M. et al. (2006) 'Biotic Homogenization and Changes in Species Diversity Across Human-Modified Ecosystems', *Proceedings of the Royal Society of London, Series B*, **273**: 2659–65.

Soga, M., and Gaston, K. J. (2016) 'Extinction of Experience: The Loss of Human–Nature Interactions', *Frontiers in Ecology and the Environment*, **14**: 94–101.

Steinberg, D. A. et al. (1997) 'Earthworm Abundance and Nitrogen Mineralization Rates Along an Urban–Rural Land Use Gradient', *Soil Biology and Biochemistry*, **29**: 427–30.

Swan, C. M. et al. (2011) 'Biodiversity and Community Composition in Urban Ecosystems: Coupled Human, Spatial and Metacommunity Processes, pp. 179–86, in Niemelä J., Breuste J. H., Guntenspergen G., McIntyre N. E., Elmqvist T., and James P. (eds) *Urban Ecology: Patterns, Processes, and Applications*, Oxford: Oxford University Press.

Szlavecz, K., Warren, P. S., and Pickett, S. T. A. (2011) 'Biodiversity in the Urban Landscape', pp. 75–101, in Cincotta R. P., and Gorenflo L. J. (eds) *Human Population: Its Influences on Biological Diversity*. *Ecological Studies 214*, Berlin Heidelberg: Springer-Verlag.

Tenenbaum, D. E. et al. (2006) 'Analysis of Soil Moisture Patterns in Forested and Suburban Catchments in Baltimore, Maryland, Using High-Resolution Photogrammetric and LIDAR Digital Elevation Datasets', *Hydrological Processes*, **20**: 219–40.

Trammell, T. L. E., Schneid, B. P., and Carreiro, M. M. (2011) 'Forest Soils Adjacent to Urban Interstates: Soil Physical and Chemical Properties, Heavy Metals, Disturbance Legacies, and Relationships with Woody Vegetation', *Urban Ecosystems*, **14**: 525–52.

United Nations (2014) 'The World Population Situation in 2014', A Concise Report. Department of Economic and Social Affairs, Population Division ST/ESA/SER.A/354. United Nations, New York, 2014.

Wall, D. H. et al. (2008) 'Global Decomposition Experiment Shows Soil Animal Impacts on Decomposition are Climate Dependent', *Global Change Biology*, **14**: 2661–77.

Wieder, R. K., and Lang, G. E. (1982) 'A Critique of the Analytical Methods Used in Examining Decomposition Data Obtained from Litter Bags', *Ecology*, **63**: 636–1642.

Yesilonis, I. D., and Pouyat, R. V. (2012) 'Carbon Stocks in Urban Forest Remnants: Atlanta and Baltimore as Case Studies', in Lal R. and Augustin B. (eds) *Carbon Sequestration in Urban Ecosystems*, pp. 103–20, New York: Springer.

—— et al. (2016) 'Historical Land Use and Stand Age Effects on Forest Soil Properties', *Forest Ecology and Management*, **370**: 83–92.

Zezza, A., and Tasciotti, L. (2010) 'Urban Agriculture, Poverty, and Food Security: Empirical Evidence from a Sample of Developing Countries', *Food Policy*, **35**: 265–73.

Zhu, W. X. et al. (2006) 'Soil Characteristics and the Accumulation of Inorganic Nitrogen in an Arid Urban Ecosystem', *Ecosystems*, **9**: 711–24.

Ziska, L. H., Bunce, J. A., and Goins, E. W. (2004) 'Characterization of an Urban–Rural $CO_2$/Temperature Gradient and Associated Changes in Initial Plant Productivity During Secondary Succession', *Oecologia*, **139**: 454–8.