Ecohydrology of Urban Ecosystems

Chapter xx

ECOHYDROLOGY OF URBAN ECOSYSTEMS

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1. Introduction

More than half of the world population lives in urban areas with projections showing a population increase up to 66% by 2050 (UN, 2015). To accommodate the growing number of city dwellers, urban areas are expanding twice as fast as their population (e.g., Seto et al., 2012), raising concerns about sustainability and livability of cities. Given the increasing importance of urban areas in both driving and being impacted by global environmental changes, there is a rapidly growing interest in understanding their dynamics (Seto and Shepherd, 2009).

Urbanization drives significant environmental changes at multiple spatial and temporal scales. The most direct impact of urbanization is the removal of natural spaces and biodiversity to build structures and impervious surfaces, thus creating a complex mosaic of land uses and covers. In urban areas, artificial surfaces are often mixed with pervious green areas, such as parks, gardens, remnant forests, and green corridors. This heterogeneous patchwork results in complex interactions between local microclimate, and hydrological and biogeochemical cycles, which affect ecosystem properties and functions (Grimm et al., 2008; Seto et al., 2011).

Pervious soil and vegetation are important components of urban ecosystems and provide essential services, such as pollution mitigation, biogeochemical cycling, and community health and well-being (Livesley et al., 2016a). At the same time, vegetation and soils that persist in the urban landscape are increasingly under threat due to human disturbances and stress factors related to changes in temperatures, water availability, nutrient, and pollution levels. Within this context, an urban ecohydrology perspective allows us to place more emphasis on interactions and feedbacks among soil, plant, and atmosphere specifically related to cities (Jenerette and Alstad, 2010; Wagner and Breil, 2013).

In urban areas, as in natural environments, ecosystem dynamics and hydrological processes are strongly coupled. The conversion of natural and vegetated land to urban uses alters land surface properties, and hence the energy balance and water fluxes. Evident changes in local microclimate have been observed in terms of higher temperature in urban areas than in their rural surroundings, a phenomenon known as urban heat island (UHI) effect (Oke, 1982). The introduction of impervious surfaces and drainage infrastructure affects ecosystem hydrologic regimes. These changes reduce the volume of water infiltrating soils, and hence groundwater recharge, as well as the volume of soil water lost through evapotranspiration, thus increasing the volume of runoff (Xiao et al., 2007). Urban development may also lead to the coexistence of native plant with non-native species, creating unique biotic communities that may have high water demand, different patterns of evapotranspiration, interception, and infiltration (Pataki et al., 2011a). Finally, urban areas strongly impact biogeochemical cycles, thus affecting urban ecosystem dynamics (Grimm et al., 2008). Cities are usually net sources of CO_2 and other greenhouse gases (GHGs) such as nitrous oxide (N_2O) and methane (CH_4), with anthropogenic activities controlling the diurnal and seasonal patterns of these fluxes (Kaye et al., 2005).

In arid and semiarid urban areas, where almost 22% of the world population lives (McDonald et al., 2011), limited natural resources magnify the importance of understanding the coupled relationship between ecosystems and hydrological processes. Urbanization drastically changes the structure and functions of arid and semiarid ecosystems; landscape plantings in an otherwise desert setting along with water and energy subsides have a great impact on both hydrologic regime and carbon stocks and fluxes.

This chapter reviews recent studies and advances in understanding and modeling links and feedback among soil, vegetation, and atmosphere in urban environments, with a particular focus on arid and semi-arid regions.

2. Urban ecosystem services

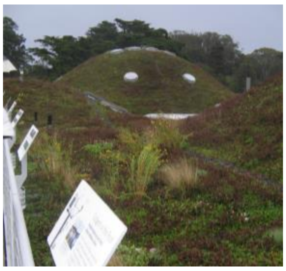
Urbanization is one of the most irreversible human impacts on the global biosphere, fragmentizing natural environments, introducing non-native species, increasing surface runoff and erosion, and degrading or altering ecosystem processes. The rapid expansion of urban areas raises the attention on preserving and restoring ecological processes and functions to reduce the ecological footprints of cities (Menz et al., 2013).

Vegetation and natural soils may play a crucial role in reducing the negative impacts of urbanization on the physical environment. In urban areas, most of the natural and semi-natural elements serve as green infrastructure. Green spaces – street trees, lawns and parks, urban forests, cultivated land, and green corridors, as well as green roofs and green walls technologies (Table 1) – provide a collection of environmental, socio-cultural, and economic benefits essential to the quality of life of urban dwellers (Dover, 2015; Palmer et al., 2015). Green areas are widespread in the urban environment. For example, in the United States, urban lawns cover a land area three times larger than any other irrigated crop in the country (Milesi et al., 2005), while trees cover about 35% of the urban land (Nowak and Greenfield, 2012).

Numerous studies over recent years have documented the role of urban green spaces in promoting ecosystem health and resilience, contributing to biodiversity conservation, and enhancing urban ecosystem functionality through the provision of ecosystem services (Chiesura, 2004; Dobbs et al., 2014; Sander, 2016; Tyrväinen et al., 2005). Ecosystem services are defined as the indirect and direct benefits that urban populations derive from ecosystem functions on local and global scales, which enhance urban sustainability and climate change adaptation (Costanza et al., 1997; Costanza et al., 2014). Following this definition, the Millennium Ecosystem Assessment (MA, 2003) and the Economics of Ecosystem Services and Biodiversity (TEEB, 2009) classified ecosystem services in four major categories (Figure 1) each relying on fundamental ecological processes (Pataki et al., 2011b): three of them (provisioning, regulating, cultural) with a direct and short-term impact on people, while supporting and habitat services show an indirect and long-term impact. As a consequence, the temporal scales of the provided ecosystem services assume a fundamental role in every decision-making process.

Urban green spaces provide direct and locally generated services, which are essential for both ecosystem and human health in urban areas. These include the potential to mitigate the UHI effect, limit GHG emissions, regulate the hydrological cycle, affect carbon (C) and nitrogen (N) biogeochemistry, and improve air and water quality by removing pollutants (Bolund and Hunhammar, 1999; Coutts et al., 2013b; Escobedo et al., 2011; Livesley et al., 2016b). Interactions with the natural environment also positively influence the physical and mental health of urban dwellers (Chiesura, 2004).

Table 1: *Types of Urban Green Spaces (Photo Credits: Edoardo Daly)*

Types of Urban Green Spaces		**Description**
Street trees		Stand-alone trees, often surrounded by paved ground.
Lawns Parks		Turf grass systems, with or without trees and other plants, created and maintained by humans for aesthetic and recreational purposes. Sports grounds and playgrounds may also be included in this group.
Urban forests		Less managed areas with higher density of trees and shrubs compared to the urban parks. They can be remnant, regrowth, or newly created forests.
Cultivated land		Patches of land used for the local production of food products (e.g., community gardens, small scale urban farms).
Green corridors		Linear green spaces which connect natural habitats and wildlife populations. When located along waterways they are called riparian corridors.
Green roofs Green walls		Green roofs are building roofs partially or completely covered with vegetation. Green walls are vertical gardens on the side of a building.

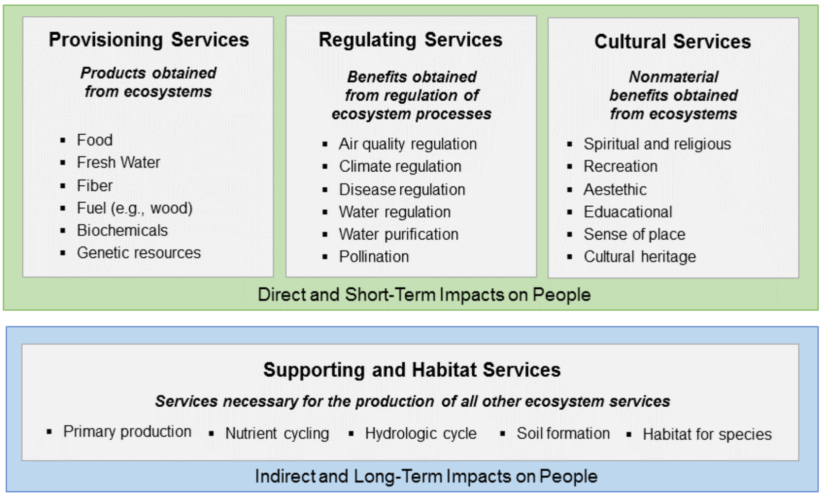


Figure 1 *Ecosystem services classification based on the Millennium Ecosystem Assessment (MA, 2003) and The Economics of Ecosystems and Biodiversity (TEBB, 2010)*

In addition to their ecological and socio-cultural values, ecosystem services have demonstrated economic value (Elmqvist et al., 2015; Pandeya et al., 2016). For example, the loss of urban vegetation can lead to increased energy costs for heating and cooling buildings, health care expenses related to respiratory diseases, and maintenance of expensive infrastructures to abate noise and pollution (Gómez-Baggethun and Barton, 2013). McPherson et al. (2005) examined the economic benefits associated with many services of urban trees (e.g., energy savings, air quality improvement, and stormwater runoff reduction among others) in five cities in the USA over a period of three years. They found annual benefits returns between $1.37 and $3.09 per tree. Specifically, in relation to tree shading and evaporative cooling, energy savings reached up to ~$550,000 a year in locations where trees were located in close proximity to buildings. Much lower savings (~$115,000) were reported in cities where trees were mainly planted along wide roads.

The urban ecosystem services framework, integrating biophysical and socio-economic concepts, is an essential component of the green infrastructure planning and preserving strategies, which aim to maximize well-being outcomes for city dwellers (Parnell et al., 2013; Sirakaya et al., 2017). On the contrary, there is a growing literature on ecosystem disservices (e.g., respiratory allergies to wind-pollinated plants, pathogens and pests, and invasive or native species causing damage to property), which are also important to be taken into account in the planning of green infrastructure establishment and maintenance (e.g., Lyytimäki, 2014; von Döhren and Haase, 2015).

As green infrastructure is increasingly adopted to mitigate the impact of urbanization on the natural environment, there is a need to obtain empirical evidence on the magnitude of such impacts, both beneficial and adverse (e.g., Matthews et al., 2015). For example, green roofs are commonly reported as a key measure to mitigate heat in urban areas. However, their performance is highly dependent upon their design (e.g., soil depth), vegetation species, and climatic conditions, which might require irrigation to support evapotranspiration (Coutts et al., 2013a). Several factors thus need to be considered for their effective use and installation.

3. Green areas and climatic, hydrological, and biogeochemical processes

Urbanization strongly alters an ecosystems structure and functions, which has a profound impact on biophysical and ecological processes at multiple scales. Urban ecosystems play a crucial role in global climate change, being significant sources of GHGs, including 78% of total carbon emissions (Grimm et al., 2008). In addition, by altering energy, water, and momentum exchanges between the land surface and atmosphere, urban development can significantly alter climate (Bonan, 1997). Land-use and land-cover transformations due to urbanization have significant impacts on hydrological processes and water quality dynamics (DeFries and Eshleman, 2004; Van Loon et al., 2016a; Van Loon et al., 2016b), as well as biogeochemical cycles (Pataki et al., 2011b).

This section separately reviews the links between urbanization and microclimate, water fluxes, and greenhouse gas emissions from urban soils, considering also the role of urban green spaces in providing benefits at different scales. The focus is on the role of urban vegetation in affecting urban microclimate and restoring more natural hydrological and biogeochemical processes, as well as the role that the built landscape plays in the physiological response of urban vegetation. Urban vegetation is generally subject to different biophysical and ecological conditions than rural and natural environments, in particular with respect to soil and climatic conditions. Therefore, in order to provide a positive effect, urban vegetation needs to remain healthy in spite of severe growing conditions and abiotic stress factors, e.g., high temperatures, low moisture conditions, light intensity, chemical stress, and air pollution (Roberts, 1977; Sieghardt et al., 2005).

3.1. MICROCLIMATE

Emissions of greenhouse gases and anthropogenic land-cover and land-use changes associated with urbanization have significant impacts on the local microclimate (Georgescu et al., 2014; Kalnay and Cai, 2003). Evident changes have been observed in terms of properties of urban surfaces (e.g., thermal, radiative, and aerodynamic properties), which in turn modify atmospheric conditions such as temperature and precipitation patterns (e.g., Oke, 1982; Oke, 1987; Yow, 2007)

Urban areas have been known to be warmer than the surrounding rural areas due to the UHI effect (Chapman et al., 2017). The magnitude of such urban-rural temperature difference varies as a function of the geographic location, the corresponding climatic region, local topography as well as specific characteristics of the built environment (e.g., urban landscape geometry and intensity of human activities) (Grawe et al., 2013).

Reviewing two decades of urban climate research (i.e., 1980-2000), Arnfield (2003) pointed out some generalizations about the UHI effect. It has long been known that UHI strongly depends on weather conditions (i.e., cloud cover and wind speed) and shows a distinct diurnal and seasonal cycle. In particular, UHI is more intense during summer, especially at night, and under stable weather conditions with light winds and clear skies.

Land cover changes and the structure of urban canyons (defined as the space above the street and between the buildings) are relevant factors that alter the energy balance, thus controlling the UHI development. The reduction of vegetation cover and the increase of artificial surfaces cause the reduction of latent heat exchange in urban canopies, which favors the UHI, except in arid areas where irrigation may increase the latent heat flux (Yow, 2007). Urban geometry and the thermal and radiative properties of building materials result in more energy absorbed and stored within urban canyons (Harman et al., 2004; Kusaka and Kimura, 2004). As a consequence, the release of stored heat is the dominant contributor to the urban warming at night, regardless of the climatic zone.

UHI intensity during daytime varies geographically because of the influence of local background climate. Zhao et al. (2014) found that UHI intensity increases in humid regions due to vegetation loss and the reduction in the convective heat transfer efficiency. Conversely, UHI intensity should decrease in arid zones where urban landscapes may have a better convection efficiency than the adjacent rural land dominated by low vegetation.

Simulations and observations in semi-arid and arid regions, however, show an urban-induced warming effect (e.g., Georgescu et al., 2013; Hedquist and Brazel, 2006) with a magnitude that can be more significant at midday during summer (Sofer and Potchter, 2006) or during night hours (Saaroni and Ziv, 2010). Compared to UHI patterns in more temperate climates, cities like Phoenix, Arizona (USA), may have cooler summer daytime temperatures due to the higher vegetation cover than the surrounding semi-desert areas (i.e. oasis effect) (Brazel et al., 2000; Stabler et al., 2005). Contrary to the oasis effect, Georgescu et al. (2011) suggests that modification of adjacent conditions, such as land cover and soil moisture, may affect the diurnal cycle of near-surface temperature, for example eliminating the daytime urban cooling.

The UHI effect is often associated with the growing number and increased intensity of heat waves in cities, which cause heat stress and health risks on urban residents (e.g., Lemonsu et al., 2015; Murari et al., 2015; Schatz and Kucharik, 2015; Zhao et al., 2018). Integrating green spaces into the urban landscape has the potential to effectively mitigate the intensity of heat islands and improve human thermal comfort during hot conditions (Oke et al., 1989). The creation of patchy green areas within the urbanized environment generates so-called cool islands, thus mitigating the adverse effects of UHI and extreme heat events (Broadbent et al., 2017b; Gill et al., 2007; Spronken-Smith and Oke, 1998). Evidence of the role of urban green spaces in affecting the air temperature of urban areas and the human thermal comfort are reported in a growing number of studies (Table 2).

Vegetation affects the thermal energy balance in urban areas mostly through direct shading and evaporative cooling (Bowler et al., 2010; Coutts et al., 2016; Gunawardena et al., 2017; Holmer et al., 2007; Sanusi et al., 2016). Tree canopies intercept solar radiation and, through direct shading, prevent the energy storage and heating of the local land surface and air; this also helps to reduce buildings energy use (Akbari, 2002; Armson et al., 2012). This shading effect creates local cool areas beneath tree canopies and protects people from the direct exposure to the sun (Dimoudi and Nikolopoulou, 2003). For example, Shahidan et al. (2010) found that for species with high leaf area index (LAI), dense foliage covers, and multiple layers of branches and twigs, the reduction of thermal radiation through absorption and reflection is about 93%, thus producing only 7% radiant heat underneath the canopy. Therefore, vegetation makes a substantial contribution to human thermal comfort (Shashua-Bar et al., 2011).

Increasing interest is directed to evaporative cooling. Through transpiration, more radiant energy is used to increase latent heat rather than sensible heat, allowing the loss of water from plants and wet surfaces as vapor into the atmosphere (Oke, 1987). The achieved cooling potential is influenced by vegetation species (i.e., C3 and C4 photosynthetic metabolism) as well as plant characteristics (e.g., crown area, foliage density, and stomatal resistance) and soil water availability (Caird et al., 2007; Doick et al., 2014).

Several aspects can influence the thermal effect of urban green areas including their characteristics (e.g., size, vegetation composition, and irrigation practices), the nature of the surrounding landscape, and the climate (Shiflett et al., 2017; Spronken-Smith and Oke, 1998; Upmanis et al., 1998). The size of vegetated areas as well as their plant density and composition may affect the magnitude of the temperature difference with their urban surrounding (Cheng et al., 2014). For example, Potchter et al. (2006) found that an urban park covered by grass and a few low trees was warmer (up to 1°C) during the day compared to the surrounding built landscape, whereas a park containing a greater number of trees with a wide canopy reached a cooling effect of 3.5°C. Multilayer plant communities have the most significant effect on temperature reduction (~4.7°C) and relative humidity increase (~7.7%) (Zhang et al., 2013), as well as plants with higher foliage density (Yu and Hien, 2006).

Table 2: *Some results from recent literature on the cooling effect of green spaces*

Urban area/ *Climate*	**Period**	**Data Collected [1]**	**Urban green space**	**Size (ha)**	**Cooling Effect Findings**	**Reference**
Vancouver (Canada) *Marine West Coast*	August 1992	Ta	- 10 parks[2]: - 1 grass (dry) - 2 grass (moist) - 2 grass (irrigated) - 2 grass TB - 1 multi-use - 1 garden - 1 savannah - 1 forest	3÷53	Max PCI[3]: - 3.2 °C (D) - 4.5 °C (D) - 4.0 °C (N) - 3.8 °C (D) - 5.0 °C (N) - 4.6 °C (N) - 3.7 °C (N) - 4.0 °C (D)	Spronken-Smith and Oke (1998)
Sacramento (California) *Mediterranean with hot dry summer*	August 1993	Ta	- 10 parks[2]: - 2 grass (irrigated) - 2 golf course - 4 multi-use - 1 Savannah - 1 Forest	2÷15	Max PCI[3]: - 4.1 °C (N) - 5.1 °C (N) - 3.8 °C (N) - 5.4 °C (N) - 5.3 °C (N)	
Göteborg (Sweden) *Marine West Coast*	January 1994 to September 1995 Night-time	Ta	3 urban parks: - Gubberoparken - Vasaparken - Slottsskogen	 2.4 3.6 156	Max PCI[3]: - 1.7 °C - 2.0 °C - 5.9 °C - Park size is important for the magnitude of PCI.	Upmanis et al. (1998)
Tel Aviv (Israel) *Subtropical with hot and humid summer*	June 2002	Ta	- Park A: grass and few low trees, mostly without shade - Park B: dense medium size trees, partially shaded (65%) - Park C: trees with high and wide canopy, well shaded (95%)	2.5 35 28	- D[4]: warmer up to 1 °C - N[4]: cooler up to 1.5 °C - D[4]: cooler up to 2 °C - N[4]: cooler between 0.5-0.7 °C - D[4]: cooler up to 2.5 °C - N[4]: cooler between 0.5-1.2 °C - Cooling effect of trees is influenced by foliage density, trees height, and size/canopy volume. - Grass parks can be warmer during daytime.	Potchter et al. (2006)
Taipei (China) *Humid subtropical*	August to September 2003 December 2003 to February 2004	Ta	61 parks out of the 490 municipal parks: - 30 - 14 - 17	 < 1 0.5÷1 >1	Average PCI [3]: - Summer noon: 0.81 °C - Summer night: 0.29 °C - Winter noon: 0.57 °C - Winter night: 0.16 °C Parks as local heat islands: - Summer noon: 0.42 °C (14 parks out of 61) - Summer night: 0.39 °C (8 parks out of 61) - Winter noon: 0.59 °C (16 parks out of 61) - Winter night: 0.45 °C (12 parks out of 61)	Chang et al. (2007)
Lisbon (Portugal) *Mediterranean*	Summer 2007	Ta	Garden in a residential setting with a dense plant community	0.6	Max PCI[2]: 6.9 °C	Oliveira et al. (2011)
Phoenix (Arizona) Tropical and Subtropical Desert	October 2007 Early morning	- T_a RH	Small park with irrigated lawn and xeric surfaces within an university campus	~ 3	- Mean (maximum) PCI[3] intensities ~3.5 °C (~6 °C). Surface type affects PCI intensities with larger PCI over the irrigated lawn.	Chow et al. (2011)

(Continue from Table 2)

Urban area *Climate*	**Period**	**Data collected [(1)]**	**Urban green space**	**Size (ha)**	**Cooling Effect Findings**	**Reference**
Shenzen (China) Humid subtropical	November 2010 to October 2011	- T_a - RH	Plant community within 3 parks: - Multilayer - Palm mixed - Trees-grass - Bamboo grove	29-66	Temperature reduction - Humidity increase - 4.7 °C - 7.7% - 3.2 °C - 6.7% - 3.6 °C - 7.0% - 2.7 °C - 6.2%	Zhang et al. (2013)
Melbourne (Australia) Marine West Coast	May 2011 to May 2012 October 2011 to June 2013	- T_a - RH - W_s	Street trees in: - Deep urban canyon - Shallow urban canyon with very little tree canopy cover - Shallow urban canyon with dense tree canopy cover		- Street trees support daytime cooling during heat events in the shallow canyon by around 0.2 to 0.6 °C and up to 0.9 °C between 9-10 AM. - Maximum daytime cooling in shallow canyon is 1.5 °C. - In the deep canyon, the shading effect of the tall buildings mask the influence of trees.	Coutts et al. (2016)
Gothenburg (Sweden) Marine West Coast	Daytime and Night-time on warmer days of 2012-2013	- Stomatal Conductance - Transpiration Rate - Phosynthetic Active Radiation (PAR) - T_a - RH	- 7 of the most common street tree species in Gothenburg; - 1 plot for each specie containing 3 to 6 similar trees		- Midday energy loss due to tree transpiration: 206 Wm^{-2} (about 30% of incoming solar radiation converted in latent heat flux). - Night-time energy loss due to tree transpiration: 24 Wm^{-2}. - Sunlit leaves transpire 3 times more than shade leaves.	Konarska et al. (2016)
Adelaide (Australia) Mediterranean	February 2011	- T_a - RH - W_s	Suburb of Mawson Lakes: - pervious (including trees, grass, low vegetation, and bare soil) - open water - impervious	 374 (54%) 42 (6%) 260 (40%)	- Irrigation can significantly reduce microscale air temperature during heatwave conditions. - Average air temperature reduction during daytime ~2.3 °C.	Broadbent et al. (2017a)

(1) Ta: Air Temperature; RH: Relative Humidity; Ws: Wind Speed.
(2) Grass: open grass surface; grass TB: grass with tree border; multi-use: park with many different components (e.g., grass playing fields; garden: cultivated park with a mixture of grass, trees, and shrubs; savannah: grass surface with trees interspersed throughout.
(3) Golf course: open grass fairways lined by trees; forest: park that has a continuous tree canopy coverage.
(4) PCI, park cooling island: maximum urban temperature (Tu) – minimum park temperature (Tp).
(5) N: night-time cooling; D: day-time cooling.

Long-term analysis on diurnal and seasonal variations of climatic conditions in urban green spaces show a much higher cooling effect in summer than in winter and during the day than at night (Cohen et al., 2012). Several studies suggest that the cooling effect of vegetation also exists at night (Chow et al., 2011; Konarska et al., 2016). Vegetated areas may have an increase in outgoing long-wave radiations due to higher sky view factor (i.e., the proportion of visible sky above a certain observation point) compared to built-up areas; this helps the night-time cooling effect (Holmer et al., 2013). In addition, tree transpiration may increase the cooling rate around and shortly after sunset, in particular in warm climates and during heat waves (Konarska et al., 2016).

Local climate change within urban areas may affect biophysical and ecological conditions of urban vegetation. Increases in temperature associated with UHI affect the growing season length, with existing observations showing longer growing season lengths in urban environments than surrounding rural areas (Zipper et al., 2016). There are thus complex relationships between the effect of urban green spaces on local climate and how the local climate in turn affects urban vegetation (Zhou et al., 2016).

Beside atmospheric temperatures, precipitation patterns are also affected by urbanization. Early investigations in the 1970s (i.e. METROMEX project, see Changnon (1981)) found evidence of increased precipitation during summer months. Since then, assessing the role of urbanization on rainfall variability has become a topic of active research (Smith et al., 2012). Shepherd (2005) reviewed observational and modelling studies on urban-induced precipitation variability. It is generally recognized that urbanization can affect precipitation mainly through (1) enhanced convergence due to increased surface roughness, (2) destabilization of the boundary layers due to UHI, (3) increases in aerosols associated with land use influencing cloud condensation nuclei (CCN) sources, and (4) changes in the urban canopy (Yu and Liu, 2015).

In most cases, studies report increases in the precipitation amount during summer months (Collier, 2006), even though such estimates can have a high degree of uncertainty (Salvadore et al., 2015). A radar-based analysis using 91 thunderstorm cases in Indianapolis, Indiana (USA), found a strong urban impact on the intensity and structure of approaching thunderstorms (Niyogi et al., 2011). Burian and Shepherd (2005) found an increase in the average warm-season rainfall amount registered in the Houston, Texas (USA), metropolitan area due to urbanization, especially during afternoons. Other studies conducted in arid cities, such as Phoenix, found increases in mean precipitation of 12-14% from a pre-urban to post-urban period (Shepherd, 2006) and more-frequent late-afternoon storms (Balling and Brazel, 1987).

Modelling studies and satellite-based observations (Miao et al., 2011; Shem and Shepherd, 2009) confirm the crucial role of urban areas in determining storm movement and rainfall amount as a function of the degree of urbanization, the UHI effect, and air pollution. Rosenfeld (2000) pointed out new insights into the negative impact of air pollution on rain-forming processes in clouds. In fact, elevated concentrations of CCN in polluted clouds might lead to many small droplets that cannot easily grow in size to produce precipitation. However, the effects of urban areas on precipitation may depend on the climate regime and the geographical locations of cities. For this reason, in some regions rainfall patterns may decrease in response to urbanization (Zhang et al., 2014b). In Beijing, China, for instance, less evaporation, higher surface temperatures, and larger sensible heat fluxes lead to a decrease in precipitation. In such a scenario, increasing vegetation cover could help to produce more rainfall (Zhang et al., 2009).

3.2. THE URBAN WATER CYCLE AND THE ROLE OF VEGETATION

Following Lerner (1990) it is possible to identify two interconnected networks of hydrological pathways in the urban environment, namely a modified natural set of pathways and water supply-sewerage pathways (Figure 2). The pathways of the natural network include precipitation, evapotranspiration, runoff, infiltration, and groundwater recharge, which are often affected by the urban microclimate and land cover changes. The water supply-sewerage network includes groundwater extraction through boreholes, flow in water mains and storm-water flow in the sewage system, irrigation, and leakage from pipes, to mention the most relevant.

In the face of increasing urbanization, there is a growing need to restore the water cycle and water quality to a regime closer to that preceding the establishment of urban developments (Fletcher et al., 2013). This is aligned to the natural flow paradigm (Poff, 1997), which prescribes the maintenance of streams close to their natural regime (Bonneau et al., 2017).

The aim of this section is to give an overview of the pathways underlying the urban water cycle and the role that urban vegetation plays in influencing such dynamics. Moreover, it provides a review of the current status of the literature about the physiological response of urban vegetation to the altered environmental conditions caused by urbanization. We did not include a review of green technologies for stormwater harvesting, i.e., low impact development (LID), water sensitive urban design (WSUD), and sustainable urban drainage systems (SUDS).

Interested readers are referred to Bonneau et al. (2017), Fletcher et al. (2015), Golden and Hoghooghi (2018), and Wong and Brown (2009), just to name a few.

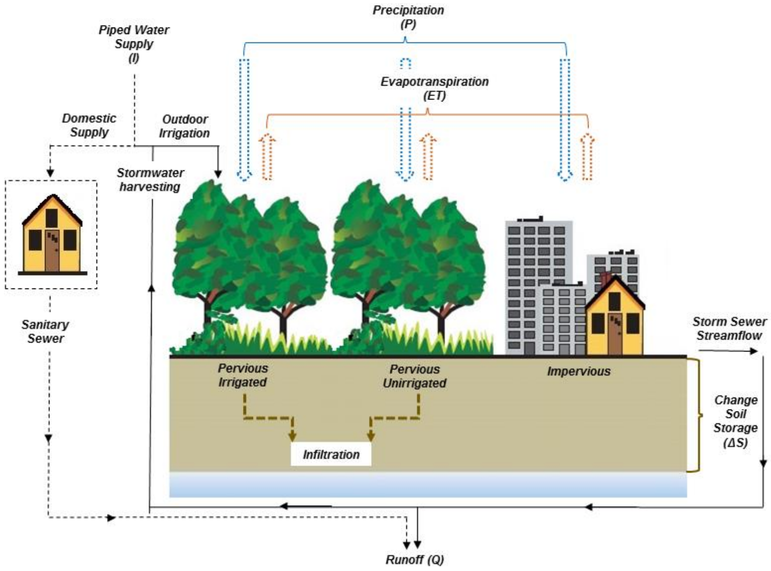


Figure 2 *Urban water balance (i.e., $P+I=Q+ET+\Delta S$) (Grimmond et al., 1986)*

The rates and volumes of surface runoff from urbanized watersheds are expected to be greater due to soil compaction, vegetation removal, and impervious surfaces (e.g., Endreny, 2005; Leopold, 1968; Xiao et al., 2007). In a study conducted in North Carolina, USA, for example, Boggs and Sun (2011) found that the mean annual discharge ratio (discharge/precipitation) was 75% higher in urban precincts compared to a forested watershed. The amount of impervious areas and their degree of connectivity with the drainage network are key factors in determining the increase of peak flows and surface runoff (Boyd et al., 1993; Shields and Tague, 2015). At the same time, it is generally accepted that the increase of urban impervious surfaces leads to a reduction in catchment evaporative losses (e.g., Barron et al., 2013; Dow and DeWalle, 2000), even though evapotranspiration (ET) in urban areas has complex patterns associated with landscape heterogeneity and changes in the urban microclimate (e.g., Litvak et al., 2017; Owuor et al., 2016; Price, 2011; Salvadore et al., 2015).

Several parameters were found to influence the interactions between urbanization and subsurface processes, including natural catchment characteristics (e.g., topography, vegetation) as well as anthropogenic factors (e.g., catchment imperviousness, spatial distribution of impervious surfaces, anthropogenic water inputs and outputs) (Hamel et al., 2013). It is also well known that the introduction of impervious surfaces and drainage infrastructure decrease infiltration and groundwater recharge (e.g., Hibbs, 2016; Shuster et al., 2005). However, several authors suggest that external anthropogenic water inputs, such as leakage from water supply systems, sewers, and stormwater drainage networks, as well as irrigation, may offset subsurface water losses (e.g., Lerner, 2002; Locatelli et al., 2017; Price, 2011). In particular, irrigation may play an essential role in groundwater recharge, being a conspicuous portion (i.e., 40-75%) of residential water use, especially in arid and semi-arid regions (Balling et al., 2008; Hilaire et al., 2008). In Austin, Texas (USA), for example, an estimated 8% of water that flows through city mains becomes groundwater recharge (Garcia-Fresca and Sharp Jr, 2005); this water and irrigation results in flows that represent at least 5% of total annual discharge (Passarello et al., 2012).

Urban vegetation has been identified as an effective measure to decrease runoff by increasing rainfall interception, soil permeability, and thus evapotranspiration (Berland et al., 2017; Yang et al., 2015). However, rainfall-runoff-infiltration processes associated with urban pervious green spaces are quite uncertain, being affected by the combined effect of soil compaction, vegetation type, topsoil removal as well as increased soil hydrophobicity and deposition of pollutants (e.g., Fletcher et al., 2013; Redfern et al., 2016).

The potential of urban trees to retain and store rainfall on leaves and branches is recognized to be extremely effective in reducing stormwater runoff because a large fraction of the water temporarily retained within the canopy is lost to the atmosphere by evaporation, a process known as canopy interception (Kermavnar and Vilhar, 2017; Xiao et al., 1998). For example, Livesley et al. (2014) estimated that eucalypt tree species can reduce rainfall reaching the ground by up to 45% in an urban street setting in Melbourne (Australia). However, canopy interception is strongly influenced by tree structure (e.g., leaf and stem surface areas, foliation period, species dimensions) and meteorological factors (e.g., amount, duration, intensity of rain events, evaporation rates) (Holder and Gibbes, 2017; Xiao et al., 2000). Xiao and McPherson (2002) found variations in the rainfall interception both depending on tree species (i.e., small canopy trees are much less effective in intercepting rainfall) and season (i.e., 14.8% of rainfall interception during a winter storm and 79.5% during a similar storm in summer for a large deciduous tree).

The ability of urban trees to be effective in mitigating the impacts of urbanization is strongly linked to the their health, stability, and longevity, which in turn depend on the ability of the root systems to grow in urban soil conditions to then acquire water and nutrients (Bijoor et al., 2012; Day et al., 2010a). Root growth and development in urban settings are often impaired by impenetrable or inhospitable soil layers, little suitable soil volume, and underground infrastructure, as well as by chemical contamination and excessive heat, thus influencing tree growth (Day et al., 2010b). In turn, tree roots alter the belowground environment through their impact on biological, physical, and chemical soil properties, thus affecting nutrient and carbon cycling, soil structure and infiltration processes. In particular, root systems increase the capacity and rate of soils to infiltrate rainfall along root channels, thus improving groundwater recharge (Bartens et al., 2008).

Biological factors such as tree size, rooting depth, and LAI also have a great impact on tree transpiration. Trees planted in urban landscapes are often exposed to high atmospheric evaporative demand because of the UHI, which results in increased temperatures and lower air humidity; this can raise tree transpiration rates, where sufficient soil moisture can sustain higher transpiration levels (Zipper et al., 2017). At the same time, limited rooting space, soil compaction, and water stress strongly affect stomatal regulation, and thus transpiration rates (Asawa et al., 2017; Chen et al., 2011; Cregg, 1995; Cregg and Dix, 2001; Litvak et al., 2011).

Vegetation response to urban conditions also strongly depends on the plant species and composition. For example, Peters et al. (2010), measuring sap flux in several dominant tree species in a suburban neighborhood in Minnesota (USA), found that evergreen needle leaf trees have higher total annual transpiration than deciduous broadleaf trees.

The coexistence of remaining native vegetation and introduced non-native species in urban green spaces may create unique biotic communities with potentially higher water demand, and different and more complex patterns of ET (Pataki et al., 2011a). In irrigated urban landscapes, soil moisture may be non-limiting, and vegetation can show higher transpiration rates largely controlled by stomatal responses to environmental conditions (e.g., Litvak et al., 2017; Litvak and Pataki, 2016). In southern California, for example, the combination of moist, irrigated soils and dry air often leads to higher urban ET than the surrounding natural ecosystem (Litvak et al., 2011). In the Los Angeles metropolitan area, where trees are often irrigated, their stomatal behavior can be consistent with the general relationship between transpiration and stomatal sensitivity in natural environments, as shown by Litvak et al. (2012).

In arid and semi-arid urban landscapes, plant-available soil moisture is the major driver for the viability of vegetation species, in particular for high water use, non-native plants (McCarthy and Pataki, 2010). For this reason, water-conserving landscape designs are encouraged as well as appropriate irrigation-scheduling depending on plant water demands (Breyer et al., 2018; Pataki et al., 2011c). Landscape design strategies include the conversion of more traditional mesic landscapes with xerophytic vegetation, which requires less water and it is recognized to be more water conservative (e.g., Holder and Gibbes, 2017; Martin and Stabler, 2002). In this context, McCarthy et al. (2011) suggested the water-use efficiency (WUE) (i.e., the ratio of water used in plant metabolism to water lost by the plant through transpiration) as a measurable indicator of the trade-off between the water use and the ecosystem services related to the growth rate to help in maximizing the growth while conserving water.

3.3. GREENHOUSE GAS EMISSIONS FROM URBAN SOILS

The environmental changes associated with urbanization, combined with human disturbances and landscape management, make urban areas hotspots for biogeochemical cycling and production of greenhouse gases (GHGs) (Decina et al., 2017; Galloway et al., 2008; Hutyra et al., 2014; Kaye et al., 2006; Livesley et al., 2016a; Pouyat et al., 2007).

Cities are commonly sources of CO_2, N_2O, and CH_4 as shown by eddy covariance observations above urbanized areas (Crawford et al., 2011; Famulari et al., 2010; Grimmond et al., 2004; Pawlak and Fortuniak, 2016). Human activities are mostly responsible for greenhouse gas emissions from urban areas, mainly from burning fossil fuels for electricity, heat, and transportation (IPCC, 2015; Kennedy et al., 2009). For example, Idso et al. (2001) found a strong but variable urban CO_2 dome in the Phoenix metropolitan area with a maximum peak in CO_2 concentration in the center of the city 75% greater than that of the surrounding rural area. Vegetated areas also have the potential to offset these anthropogenic emissions by reducing the level of atmospheric CO_2 as a result of their CO_2 uptake through photosynthesis (Kordowski and Kuttler, 2010; Nordbo et al., 2012).

Urban soils represent a major source of GHGs with the magnitude of emissions greatly influenced by urban environmental factors. Urbanization strongly affects soil carbon (C) and nitrogen (N) cycling and related GHG emissions (Lorenz and Lal, 2009; Pouyat et al., 2002). The exchange of CO_2, N_2O, and CH_4 between urban soils and the atmosphere strongly depends on factors that are greatly modified by urbanization, such as soil moisture and temperature, soil structure and texture, nutrient inputs and substrate availability (van Delden et al., 2016).

In urban environments, soils are often disturbed through removal and compaction, and the introduction of impervious surfaces, having strong influences on the soil structure and texture, C pools, and microbial community structure and function. Additionally, land management practices, such as fertilization and irrigation, have the potential to increase soil organic C accumulation and N inputs, which might result in increased GHG emissions, including N_2O and CH_4 (e.g., Kaye et al., 2004; Livesley et al., 2010). For example, in northern Colorado, urban lawns account for 30% of the soil N_2O emissions from that region, just occupying 6.4% of the land area (Kaye et al., 2004).

The impact of all these anthropogenic drivers on C and N cycles takes place through complex interactions at various levels, making it difficult to identify the contribution of individual factors. This section focuses on the exchange of the three major GHGs (i.e., CO_2, N_2O, and CH_4) between soil and atmosphere, and does not provide a complete review of the soil C and N cycles (see Chapter xx and Lal and Stewart (2017) for more details).

Several studies in the last two decades have been conducted to assess the magnitude of GHG fluxes from urban vegetated soils and the related effects of urban environmental changes. Some of these results are shown in Tables 3, 4, and 5 for CO_2, N_2O, and CH_4 respectively.

Variations in soil CO_2 emission rates across urban, rural, and natural environments are shown in Table 3. Long-term data collected within the Baltimore Ecosystem Study (Groffman et al., 2006; Groffman et al., 2009) showed marked seasonal patterns in the CO_2 fluxes with highest rates during warmer seasons. Lawns often have the highest CO_2 fluxes, especially when highly managed (i.e., watered and fertilized regularly) (e.g., Golubiewski, 2006; Kaye et al., 2005; Koerner and Klopatek, 2010; Pouyat et al., 2002). In arid and semiarid regions, such as the Phoenix metropolitan area, urbanization drastically changed the biologic structure of the ecosystem mainly due to water and energy subsidies, which in turn impact carbon stock and fluxes (Koerner and Klopatek, 2002; Koerner and Klopatek, 2010). In such ecosystems, soil respiration rates significantly increase compared to desert remnant areas because soil moisture is recognized to be the main factor controlling soil CO_2 efflux.

In terms of N_2O fluxes (Table 4), the climatic region plays a crucial role in the magnitude of such emissions. In humid areas, such as Baltimore, although urban grasslands can be heavily fertilized and irrigated, N_2O fluxes are not higher than those from urban forest plots; moreover, lower fluxes were found in the more heavily fertilized grass sites compared to less fertilized sites (Groffman et al., 2009). In dry regions, such as Colorado and Arizona, on the contrary, urban lawns exhibit higher N_2O fluxes compared to the native landscapes (see also Chapter xy(emissions)) due to the combination of fertilizer and water inputs (Hall et al., 2008; Kaye et al., 2004).

The use of fertilizer may not have an immediate effect on N_2O and CO_2 fluxes. For example, Raciti et al. (2011) found a significant increase in N_2O fluxes several days after fertilization. Livesley et al. (2010) obtained similar results by measuring N_2O emissions in lawn plots in Melbourne (Australia). In this specific case study the application of fertilizer caused a peak in N_2O emissions six days after its application. However, they found that irrigation and soil moisture content, rather than fertilizer, determine the long-term mean differences in N_2O emissions.

The net flux of CH_4 (Table 5) from soils is the results of anaerobic production and aerobic consumption of CH_4 (see Chapter xy and Wachinger et al., 2000). Soils in natural ecosystems, such as temperate forest, grassland, and desert, are normally CH_4 sinks; soils in urban ecosystems instead have low rates of CH_4 uptake (Costa and Groffman, 2013). Reduced CH_4 uptake were recorded in the urban forest sites within the Baltimore Ecosystem Study compared to the rural forest sites, while almost negligible fluxes were found in lawns (Groffman and Pouyat, 2009). Rural sites show higher CH_4 consumption compared to urban forest sites also in Guangzhou City metropolitan area (South China) (Zhang et al., 2014a).

CH_4 uptake is affected by urbanization through environmental changes in chemistry, climate, and land transformation (Costa and Groffman, 2013). In particular CH_4 uptake can be influenced by soil texture and moisture, mainly because of the rate of diffusion of atmospheric CH_4 into the soil. For example, coarse-textured and dry soils have higher rates of uptake due to enhanced diffusion rates. CH_4 uptake is also strongly affected by high levels of atmospheric CO_2 and N additions (i.e. N deposition), which exert an inhibitory effect. Finally, differences in the population of CH_4 oxidizing organisms may play an important role in the CH_4 dynamics.

Table 3: *CO_2 soil emission rates in $g_{CO2\text{-}C}\ m^{-2}\ h^{-1}$ for different cover types and climatic regions. Ranges have been often extrapolated from figures and then rounded for clarity (see original literature for more details).*

Region/*Climate*	**Land type**	**Rate**	**Comments**	**Findings**	**Reference**
Phoenix (Arizona) *Tropical and Subtropical Desert*	- Native desert - Abandoned agricultural - Xeric landscape - Mesic landscape (1) - Agricultural land - Golf courses - Landfills	0.02 0.04 0.04 0.30 0.25 0.33 0.59	Means June 2000 - May 2001	Lowest CO_2 emissions in the desert area, abandoned agricultural and xeric land. Highest rates of CO_2 in the landfills. CO_2 fluxes increase with increasing soil moisture.	Koerner and Klopatek (2002)
Fort Collins (Colorado) *Tropical and Subtropical Steppe*	- Urban lawns (2) - Dryland wheat-fallow - Irrigated corn - Native shortgrass	0.31 0.05 0.12 0.10	Annual means January - December 2001	Seasonal and annual variability in CO_2 fluxes. Higher CO_2 emissions and C allocation in urban lawns due to fertilization and irrigation inputs.	Kaye et al. (2005)
Baltimore (3) (Maryland) *Humid Subtropical*	- Urban forest - Rural forest	0.09 0.07	Means fall 1998 - fall 2002	CO_2 fluxes are strongly influenced by exposure to the urban land use and atmosphere.	Groffman et al. (2006)
Baltimore (3) (Maryland) *Humid Subtropical*	- Urban forest - Rural forest - Urban grassland	0.12 0.08 0.11	Means June 2001 - May 2004	High CO_2 fluxes in the grass plots driven by high productivity and high temperatures in the grass plots (lack of tree canopy).	Groffman et al. (2009)
Melbourne (Australia) *Marine West Coast*	- Urban lawn(4): - Rain only - Rain+Fertilizer - Irrigated - Irrigated+Fertilizer - Rain reduced - Mulch+Irrigated	 0.39 0.41 0.42 0.47 0.37 0.38	Means August-November 2007	No significant difference in CO_2 emissions among the lawn and mulched treatments.	Livesley et al. (2010)
Melbourne (Australia) *Marine West Coast*	Biofilter(5): - Cell I - Cell III	 0.10 0.10	Means March-May 2011 January-February 2012	Small values of CO_2 fluxes due to low levels of soil C in the biofilter (new sand dominated system).	Grover et al. (2013)
Singapore *Tropical Rainforest*	- Bare soil NC (6) - Bare soil WC - Turf soil NC - Turf soil WC	0.10 0.14 0.36 0.30	Means July - December 2012	Influence of management practices on CO_2 fluxes in urban tropical grassland. Bare soils: lowest mean CO_2 efflux rates. Turf with no grass clipping: highest mean CO_2 efflux rates.	Ng et al. (2015)
Boston (Massachusetts) *Hot Summer Continental*	- Urban forest - Rural forest - Urban lawn - Urban landscaped	0.11 0.19 0.29 0.13	Growing season means May - November 2014	Similar values between urban and rural forest soils considering growing season mean soil CO_2 efflux rates.	Decina et al. (2016)

(1) Grass lawns
(2) Well-managed with irrigation, mowing, and fertilization.
(3) Baltimore ecosystem Study Long-term Study Plots.
(4) 5 different lawn treatments with different water and nutrient management and 1 mulched and drip irrigated garden bed.
(5) Biofilter that treats stormwater from a multistory carpark; no fertilization. It is divided in three cells, but experiments were conducted in cell I (sandy loam) and cell III (80% sandy loam, 10% compost, and 10% hardwood mulch).
(6) NC: no grass clipping; WC: whit grass clipping.

Table 4: *N_2O soil emission rates in $\mu g_{N2O\text{-}N}$ m^{-2} h^{-1} for different cover types and climatic regions. Ranges have been often extrapolated from figures and then rounded for clarity (see original literature for more details).*

Region/*Climate*	**Land type**	**Rate**	**Comments**	**Findings**	**Reference**
Fort Collins (Colorado) *Tropical and Subtropical Steppe*	- Native shortgrass - Dryland wheat-fallow - Irrigated corn - Urban lawns[(1)]	2.20 1.80 23.20 27.60	Annual fluxes November 2000 to November 2001	N_2O fluxes from urban grassland are about 10 times larger than emissions from native grassland.	Kaye et al. (2004)
Baltimore [(2)] (Maryland) *Humid Subtropical Climate*	- Urban forest - Rural forest	14.30 7.00	Means fall 1998 - fall 2002	Finest textured soils lead to higher N_2O fluxes. Species change (plant or soil community) is a powerful driver of N cycling along urban to rural gradients.	Groffman et al. (2006)
Baltimore [(2)] (Maryland) *Humid Subtropical Climate*	- Urban forest - Rural forest - Urban grassland	40.54 6.22 25.14	Means June 2001 - May 2004	Although urban grasslands can be heavily fertilized and irrigated, N_2O fluxes are lower than urban forest plots. Water could be the key driver of N_2O emissions in urban landscapes. Highest N_2O fluxes rates during wetter months.	Groffman et al. (2009)
Phoenix (Arizona) *Tropical and Subtropical Desert*	- Urban lawns - Managed Xeriscapes - Remnant desert sites [(3)]	36.11 11.11 5.78	Means taking into account measurements before and after watering March 2001 May/June 2006	Higher N_2O emissions in urban lawns. Alternative urban landscaping practices (xeriscape) can reduce N_2O emissions and promote water conservation. Increased N_2O emissions in urbanized areas compared to the native landscaped due to the expansion of irrigated and fertilized lawns.	Hall et al. (2008)
Irvine (California) *Mediterranean*	- Urban lawns (ornamental lawns and athletic fields)	11.52-34.25	Annual means depending on fertilization rate	Great impact of fertilization on N_2O emissions.	Townsend-Small and Czimczik (2010)
Melbourne (Australia) *Marine West Coast*	- Urban lawn[(4)]: - Rain only - Rain+Fertilizer - Irrigated - Irrigated+Fertilizer - Rain reduced - Mulch+Irrigated	 18.00 15.67 28.67 28.00 9.00 14.13	Means August-November 2007	Irrigation and soil moisture content, rather that fertilizer and N status, influence long-term mean differences in N_2O emissions among treatments.	Livesley et al. (2010)
Melbourne (Australia) *Marine West Coast*	Biofilter[(4)]: - Cell I - Cell III	 13.70 65.60	Means March-May 2011 January-February 2012	Higher N_2O in cell III may be attributed to denitrification occurring within the saturated zone and the wet pockets within the unsaturated filter media.	Grover et al. (2013)

(1) Well-managed with irrigation, mowing, and fertilization.
(2) Baltimore ecosystem Study Long-term Study Plots.
(3) Protected Sonora Desert lands within the urban core.
(4) Five different lawn treatments with different water and nutrient management and one mulched and drip irrigated garden bed.
(5) Biofilter that treats stormwater from a multistorey carpark; no fertilization. It is divided in three cells, but experiments were conducted in cell I (sandy loam) and cell III (80% sandy loam, 10% compost, and 10% hardwood mulch).

Table 5: *CH_4 soil rates in $\mu g_{CH4\text{-}C}$ m^{-2} h^{-1} for different cover types and climatic regions. Ranges have been often extrapolated from figures and then rounded for clarity (see original literature for more details).*

Region/*Climate*	**Land type**	**Rate**	**Comments**	**Findings**	**Reference**
Fort Collins (Colorado) *Tropical and Subtropical Steppe*	- Native shortgrass - Dryland wheat-fallow - Irrigated corn - Urban lawns[(1)]	-20.43 -34.85 -11.42 -16.82	Annual fluxes November 2000 to November 2001	Highest CH_4 uptake in the native grasslands. Lowest mean CH_4 uptake rates in corn and urban lawns. CH_4 from urban soils is half of the flux from native grasslands.	Kaye et al. (2004)
Baltimore [(2)] (Maryland) *Humid Subtropical Climate*	- Urban forest - Rural forest	-42.19 -79.17	Means fall 1998 - fall 2002	Significant differences in CH_4 uptake between urban and rural land. CH_4 strongly influenced by soil texture and moisture. Coarse-textured dry soils have the highest rates of uptake due to enhanced diffusion of atmospheric CH_4 into the soil.	Groffman et al. (2006)
Baltimore [(2)] (Maryland) *Humid Subtropical Climate*	- Urban forest - Rural forest - Urban grassland	-33.75 -70.68 -1.06	Means June 2001 - May 2004	Highest consumption in rural forest sites. Negligible fluxes from urban lawns due to the inhibitory effect of N additions on CH_4 uptake ("lawn effect"). CH_4 is strongly affected by soil moisture with an effect on diffusion (essential for the movement of CH_4 from the atmosphere to the microorganisms that oxidize in the soil).	Groffman and Pouyat (2009)
Melbourne (Australia) *Marine West Coast*	- Urban lawn[(3)]: - Rain only - Rain+Fertilizer - Irrigated - Irrigated+Fertilizer - Rain reduced - Mulch+Irrigated	 -6.67 -4.17 -5.00 -5.00 -3.33 -31.25	Means August-November 2007	No significant difference between the long-term mean soil CH_4 exchange rates in any of the lawn treatments.	Livesley et al. (2010)
Melbourne (Australia) *Marine West Coast*	Biofilter[(4)]: - Cell I - Cell III	 -18.30 -3.80	Means March-May 2011 January-February 2012	CH_4 sink strength of the cell with the saturated zone (cell III) is lower than cell I. CH_4 uptake rates similar to other urban lawn systems. Biofilter cells have occasional large CH_4 emissions following inflow events.	Grover et al. (2013)

(1) Well-managed with irrigation, mowing, and fertilization.
(2) Baltimore ecosystem Study Long-term Study Plots.
(3) Five different lawn treatments with different water and nutrient management and one mulched and drip irrigated garden bed.
(4) Biofilter that treats stormwater from a multistorey carpark; no fertilization. It is divided in three cells, but experiments were conducted in cell I (sandy loam) and cell III (80% sandy loam, 10% compost, and 10% hardwood mulch).

4. Modeling urban ecosystems

Hydrologic modelling of urban environments is very complex because of the range of natural and anthropogenic processes involved. Although the nature and objectives of urban models cover a wide range (Masson, 2006), a large part of the hydrologic modelling in urban areas has focused on simulating storm water quantity and quality, often with lumped and semi-distributed models (Fletcher et al., 2013; Mejía et al., 2014; Petrucci and Bonhomme, 2014; Salvadore et al., 2015; Zoppou, 2001). Ecohydrological models, describing the water dynamics across the soil-plant-atmosphere continuum through coupling hydrologic and biophysical processes, have been mainly used to study natural and rural environments (e.g., Fatichi et al., 2016; Fatichi et al., 2014). However, the need to quantify impacts and reveal tradeoffs associated with the conversion of natural to urban landscapes has led to a growing interest in applying ecohydrological models to urbanized areas (Salamanca et al., 2018).

The objective of this section is to give an overview of available models in literature that have extended the analysis of hydro-biophysical processes from natural environments to urban areas at different scales. Three main scales are identified: (i) ecohydrological models that represent regional- to catchment-scale interactions between major biogeochemical cycles, water cycle, and vegetation dynamics, (ii) models that operate at the neighborhood scale to simulate soil-plant-atmosphere interactions, and (iii) models that simulate water and nutrient dynamics at the tree level.

We do not explicitly discuss urban storm water models, such as MIKE URBAN (www.dhisoftware.com/mikeurban), Storm Water Management Model (SWMM; Huber et al., 1988), Models of Urban Storm-water Improvement Conceptualization (MUSIC; Wong et al., 2002), Urban Runoff Branching Structure (URBS; Rodriguez et al., 2005), MOdel for Urban SEwers (MOUSE; Lindberg et al., 1989), and CANOE® (Lhomme et al., 2004), which are frequently used for the integrated management of urban runoff (Fletcher et al., 2013). Some of these models are also useful to predict water flow effects of LID, WSUD, and SUDS (Elliott and Trowsdale, 2007) as well as the water treatment performance of such systems (e.g., Daly et al., 2012; Kabir et al., 2017; Locatelli et al., 2017; Randelovic et al., 2016; Vezzaro et al., 2011; Vezzaro et al., 2012).

4.1. REGIONAL TO CATCHMENT SCALE

Applications of catchment scale ecohydrological models in urbanized environments are still rare, and few examples are reported here.

To investigate water fluxes at the regional scale over a long period where dramatic environmental changes in climate and land use happened, Liu et al. (2013) applied the dynamic land ecosystem model (DLEM) to the drainage basin of the Gulf of Mexico during the period 1901-2008. Coupling major biogeochemical cycles, water cycle, and vegetation dynamics, DLEM allowed for the investigation of spatial and temporal variations in evapotranspiration (ET) and runoff (R) due to climate and land use changes. The study shows that climate change is the dominant factor controlling the inter-annual variations of ET and runoff, with precipitation playing the major role in the variations of annual water fluxes over the whole region. Both changes in land use and climate led to a general reduction in ET during the study period with high spatial heterogeneity due to different microclimate conditions across the region.

Shields and Tague (2015) have applied the Regional Hydro-Ecological Simulation System (RHESSys) to an urbanized catchment in Santa Barbara, California (USA), to study the impact of impervious area connectivity on water and carbon fluxes in a heterogeneous urban precinct. Results show that the amount of impervious surfaces with a direct hydrologic connection to the drainage network (i.e., effective impervious area, EIA) has a significant impact on transpiration (T) and net primary productivity (NPP). In particular, by reducing the EIA fraction, the reductions in T and NPP associated with increased impervious areas and vegetation loss are significantly lower.

To simulate the effects of changes of urban tree cover and impervious surfaces on urban hydrology at the catchment scale, the U.S. Department of Agriculture (USDA) Forest Service developed UFORE-Hydro, now called i-Tree Hydro, a semi-distributed urban soil-vegetation-atmosphere transfer scheme (Wang et al., 2008). The model represents the watershed surface as impervious or pervious surfaces both with a certain percentage of canopy cover. The model was firstly applied to an urban catchment of about 15 km^2 in Baltimore to examine the tree interception under different vegetative and meteorological conditions as well as the tree effects on runoff generation. Results showed how trees significantly reduce runoff, in particular for low intensity and short duration rainfall events. i-Tree Hydro is part of the i-Tree software suite for urban forest analysis and benefit assessment. Within i-Tree, entire urban forests and street trees can be studied with i-Tree Eco and i-Tree Streets respectively, to estimate vegetation structure (e.g., urban tree growth) and ecosystem services (McPherson and Peper, 2012; Pace et al., 2018).

4.2. NEIGHBORHOOD SCALE

At smaller spatial scales, a more extensive literature exists on modeling the interactions between urban surfaces, vegetation, and the atmosphere.

Common urban canopy models available in the literature are radiative models centered on the energy balance of urban canyons or precincts. A review of the features of these models can be found in Grimmond et al. (2010) and references therein. A recognized limitation of these models is the description of latent heat fluxes associated with the presence of vegetation in the urban landscape. These fluxes are often not included (Krayenhoff and Voogt, 2007) or their description is simplified (Krayenhoff et al., 2014; Lee and Park, 2008).

Recently, urban trees have been embedded in some urban canopy energy balance models, linking these models to the soil water balance (e.g., Ryu et al., 2016). For example, Nice et al. (2018) modified the Temperature and Urban Facets in 3D (TUF-3D; Krayenhoff and Voogt, 2007) to create Vegetated Temperatures of Urban Facets in 3D (VTUF-3D). TUF-3D is a microscale energy balance model for urban environments, where energy fluxes are described via radiation, convection and conduction. VTUF-3D was developed by coupling TUF-3D to the MAESPA tree process model (Duursma and Medlyn, 2012), which couples canopy processes, such as radiative transfer, transpiration and carbon assimilation, to the soil water and energy balances. Once embedded into TUF-3D, MAESPA can account for the effect of trees due to shading of buildings as well as the role of urban trees in affecting soil and air temperatures via root water uptake, interception, and transpiration.

These energy balance models keep a certain level of simplicity and are mostly centered on the simulation of surface temperatures of buildings and paved areas, using a simplified description of atmospheric wind speed and temperatures. Computational fluid dynamics models (CFD), such as Large Eddy Simulation (LES) (e.g., Bou-Zeid et al., 2009; Giometto et al., 2016; Girard et al., 2017; Hertwig et al., 2017a; Hertwig et al., 2017b; Kanda et al., 2013; Toparlar et al., 2017), can provide detailed descriptions of wind speed and temperatures within the urban atmospheric boundary layer. Differently from radiative models, CFD models are computationally demanding and present challenges with the representation of geometrical features of the urban landscape. Nonetheless, recent studies with LES were able to include the geometry of street trees identifying their role in reducing pressure loads on buildings (Giometto et al., 2017; Giometto et al., 2016).

A well-established CFD model for urban precincts is ENVI-met© (Bruse and Fleer, 1998), which couples the incompressible Reynolds Averaged Navier-Stokes equations to advection-diffusion equations for specific humidity and temperature and includes spatially variable sources due to tree canopies. ENVI-met is largely used in microclimatic analysis that involve the study of urban vegetation and its potential to mitigate human heat stress (e.g., Lee et al., 2016; Morakinyo et al., 2017; Simon et al., 2018).

4.3. TREE SCALE

Very few studies exist in the literature focused on modeling the water dynamics in soil-plant systems in urban environments at the local tree scale, despite the importance of urban trees and their ecosystem services.

Several authors suggest the application of a stochastic model of the soil water balance, previously applied to natural and agricultural environments (e.g., Chapter 3; Daly et al., 2004; Laio et al., 2001; Vico and Porporato, 2011), to urban areas. Specific applications include the study of water dynamics in isolated and in-row trees (Vico et al., 2014) and irrigated and non-irrigated experimental landscaping treatments (Volo et al., 2014). These models, explicitly including rainfall unpredictability within a probabilistic framework, quantify the statistics of tree transpiration and water stress from the statistics of soil moisture, which depend on tree characteristics (i.e., species, size), planting design, root zone features, and precipitation and irrigation frequencies and amounts.

Having a model to explore planting design and irrigation scenarios on plant water stress can be particularly useful in desert urban landscapes, where plant-available soil moisture is a major driver for vegetation health, in particular for non-native plants (McCarthy and Pataki, 2010). For example, Volo et al. (2014) applied the model to the Phoenix metropolitan area to explore the effects of irrigation in terms of soil moisture dynamics, water balance partitioning, and plant water stress taking into account xeric (i.e., low water use vegetation) and mesic (i.e., turf grass and high water use trees) landscape design.

The model used by Vico et al. (2014) for assessing street tree water balance was recently coupled with the nutrient dynamics model proposed by Porporato et al. (2003) to explore the feedbacks between water and nutrient dynamics in the context of urban street trees (Revelli and Porporato, 2018). Model results (i.e., soil water and N concentration, water and N fluxes in and out of the system) can be used to quantify some ecosystem services provided by street trees, such as cooling effect, soil carbon sequestration, and storm-water management. In particular, the authors found that urban design (e.g., design of the soil compartment around the tress) and irrigation schedules must be optimized to enhance ecosystem services and minimize plant water stress.

5. Conclusion and future directions

Given the speed at which cities are expanding and becoming densely populated, there is a growing need to support sustainable uses of soil, vegetation, and water resources in urban areas.

It is well known that urbanization drives significant environmental changes at multiple spatial and temporal scales. Evidence of the influence of urbanization on local and regional climate are extensively documented, in particular in relation to the UHI effect and human thermal comfort. The impacts of urban development on hydrological and biogeochemical cycles have been quite well studied in the last two decades. Recently, more attention has been paid to the components of the urban water balance that are still highly uncertain (e.g., ET) and quite difficult to determine due to the complex heterogeneity of the urban landscape. Modeling tools are also available to assess the interactions between surfaces, vegetation, and the atmosphere in urban areas in particular from a climatic perspective.

Sustainable uses of soil and water resources in urban areas need to be supported by a better understanding of the ecosystem responses to the environmental changes associated with urbanization. Given these challenges, possible future directions for urban ecohydrology are suggested. The potential of UHI-induced changes to impact patterns of water, energy, and carbon cycling within cities is highlighted in several studies (e.g., Zipper et al., 2017). However, there is a need to improve empirical evidence of ET patterns in urban areas and to better understand the bidirectional feedbacks between urban vegetation and UHI, in particular in terms of evaporative cooling. Moreover, quantifying the carbon balance of urban areas, including both green and non-green areas, may represent a hot topic for future research, also taking into account how urban warming could affect carbon sequestration and emissions from urban soils.

A better understanding of the interaction between urban trees and the built environment is needed, in particular in terms of limited root growth due to soil compaction and infrastructure elements, and limited canopy growth due to radiation interception by buildings. Ecohydrological modeling of urban areas is perhaps the least explored research area. Several existing distributed models, which couple hydrological processes and vegetation dynamics (e.g., Tethys-Chloris, Fatichi et al. (2012); tRIBS-VEGGIE, Ivanov et al. (2008); CATHY-NoahMP, Niu et al. (2014)) could be extended to urban areas. Moreover, coupling radiative models with hydrological models and perhaps C and N cycle models at different scales, taking for example into account the UHI impacts, would bring an integrated and more complete evaluation of urban ecohydrological processes.

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