Environmental factors affecting GHG dynamics in urban and forested soils

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Abstract

Urbanization causes negative and irreversible effects on the environment. One of the main ways urbanization can affect the environment is by its contributions to local and global climate change. Currently, more than half of the global population lives in urban areas and this number is expected to increase for the foreseeable future. As a result, impacts of urbanization on climate change are expected to increase as well. Changes in the soil environment are one of many ways that urbanization can contribute to climate change. Soils are a central component of the consumption and production of the three major greenhouse gases (CO_2, CH_4, and N_2O) that contribute to climate change. Soils can act as a sink or source for these greenhouse gases (GHG). Soil physical, chemical, and biological properties can influence soil processes which can in turn affect GHG dynamics. To understand the impact of urbanization on GHG dynamics, we reviewed the microbial processes responsible for GHG uptake in soils. Next, we identified the environmental factors that can alter these microbial processes and finally we quantified how urban vs. forested soils may differentially affect these environmental factors to estimate the effect of urbanization on GHG dynamics. We reviewed literature to identify the major environmental factors that impact GHG fluxes. Soil temperature and soil moisture affected the dynamics of each of the three major GHGs. Other factors like nitrogen availability, soil pH, and litterfall input were factors for specific GHGs. After identifying the important factors driving GHG dynamics, we found that certain environmental factors differed substantially between urban and forested soils, like soil temperature and soil pH, whereas other factors like litterfall input, nitrogen availability, and soil moisture were not. Regardless of the differences among soils or environmental drivers, urban soils will be increasingly important for GHG dynamics and climate change due to the increasing population. Further, a positive feedback of climate change affecting urban GHG dynamics via altered temperature and soil moisture conditions suggests a potential feedback loop between urbanization and climate change.

Introduction

Urbanization refers to the increasing amount of people living in urban areas, representing a shift in the population from rural areas to cities. According to the United Nations (2015), 54% of the global population lived in urban areas in 2014 and a projected 66% of the global population is projected to live in urban areas by 2050. This transition is triggered by social, economic, and political developments. Urbanization is accompanied by increasing the prosperity of individuals in the form of access to better education, employment opportunities, and social benefits and services. However, urbanization has also led to or enhanced destructive and irreversible effects on the environment, like climate change, air pollution, sediment and soil erosion, increased flooding, and loss of habitat (Wang, Chen, and Kubota, 2016).

In addition to urbanization, climate change represents another agent of global change that affects urban, rural, and pristine ecosystems alike. Over the period of 1880-2012 the globally combined land and ocean surface temperature has warmed by 0.85 °C per year on average. Each of the past three decades before 2013 have been warmer than the previous and the first decade in the 21st century is the warmest on record (Hartmann et al., 2013). In addition to warming temperatures, climate change is also expected to alter precipitation patterns, with many regions expected to experience more extreme events (Wuebbles et a., 2017). Although climate change is occurring on a global scale, it may be especially impactful in urban areas. Urban areas typically experience higher temperatures than nearby agricultural or natural areas because buildings and other infrastructure trap/store more heat than natural counterparts (Satterthwaite, 2007).

The combination of urban structures, hard surfaces, and shortage of vegetation all contribute to this increased temperature in urban areas known as the urban heat island effect (Stone, 2012). The urban heat island effect can be quantified by examining the difference in temperature between rural

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and urban areas located in close proximity to each other. For example, night-time temperatures in London are up to 7°C warmer than rural temperatures 20 km away (Watkins, Polmer, and Kolokotroni, 2007). Around the Baltimore metropolitan area, George et al. (2007) found that soil temperature of an urban site compared to the rural site was 0.7°C higher and the air temperature was 2.1°C higher. Another study by Heisler et al. (2007) found that during the summer the air temperature was at least 7°C warmer in downtown Baltimore than a nearby rural forested area.

The urban heat island effect can intensify changes in precipitation driven by climate change, as well. Jauregui and Romales (1996), observed a correlation between daytime urban heat island and intensification of rain showers during wet months (May-October) in Mexico City. They also performed an analysis of historical records showing the frequency of intense rain showers is correlated with the growth of the city (Jauregui and Romales 1996). Rising warm air, promoted by urban heat islands, may help produce clouds that result in more rainfall around cities. The different heights of the buildings in cities help form clouds by causing winds to converge, driving them upward (Shepherd, 2005). These changes in climate can significantly impact urban environments.

At the global scale, climate change is caused by the accumulation of greenhouse gases in the atmosphere. Greenhouse gases (GHG) trap heat in the atmosphere causing a greenhouse effect that alters global climate patterns. This greenhouse effect is a naturally occurring phenomenon, but human activities have increased GHG emissions, leading to a buildup of GHGs in the atmosphere. The three major greenhouse gases are carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O). NOAA's Annual Greenhouse Gas Index (2018) estimates CO_2 atmospheric concentrations at 405 ppm, CH_4 at 1850 ppb, and N_2O at 327 ppb for 2018. Of the total U.S. greenhouse gas emissions in 2016 (6,511 million metric tons of CO_2 equivalent), CO_2, CH_4, and N_2O accounted for 81, 10, and 6%, respectively (EPA, 2018). Before the Industrial Revolution, the carbon and nitrogen cycles were in balance, but human activities have altered the balance leading to an increase in the amount of CO_2, CH_4, and N_2O

released into the atmosphere each year. More GHGs are emitted to the atmosphere than can be taken up by global sinks (Pierzynski et al., 2009), which leads to the accumulation of GHGs and ultimately causes climate change. Urban areas are increasingly becoming an area of concern for climate change because they account for 67% of energy related global greenhouse gas emissions and this percentage is expected to rise to 74% by 2030 (World Bank Group, 2010). The large contribution of urban areas to GHG emissions, coupled with the projections of an increasingly urban global population in the coming decades, makes understanding GHG dynamics of urban areas critical to understand and combat climate change.

As one of the five forming factors of soil (Jenny 1941), climate plays a significant role on the properties of soil and vice versa. Soil physical, chemical, and biological properties can influence soil processes which can in turn affect GHG dynamics. Soils are capable of acting as a source of CO_2, CH_4, or N_2O via the processes of respiration, methanogenesis, nitrification and denitrification. In contrast, soils can also be a sink for CO_2, CH_4, or N_2O via photosynthesis, methanotrophy, and denitrification. Each of these processes can be affected by urbanization, potentially coupling two powerful global agents of change (urbanization, climate change) through soil processes. To understand the impact of urbanization on GHG dynamics, GHG dynamics in urban and forested soils were compared by looking into the various soil processes that affect the capacity of soils to act as a GHG source or sink. The goal of this paper is to identify the environmental factors that affect the source-sink dynamics of GHGs in soils and to discuss how urbanization affects these factors. Below, we discuss the three major GHGs, and identify major environmental factors driving the dynamics of each GHG. We attempt to establish how urbanization alters these environmental factors, ultimately linking urbanization with altered GHG dynamics.

Methane

CH_4 is formed through a process known as methanogenesis, which is carried out by a subset of microbial organisms known as methanogens. Methanogenesis is an anaerobic (i.e., occurs in the absence of oxygen) metabolic pathway that produces CH_4 as the terminal product. Methanogenesis is an example of anaerobic respiration, that microbial organisms perform in the absence of more energetically favorable electron acceptors such as oxygen, nitrate, or sulfate. The terminal electron acceptor in methanogenesis is carbon. CH_4 emission via methanogenesis is linked to two processes (Mach et al. 2015):

1. acetoclastic methanogenesis: $CH_3COOH \rightarrow CO_2 + CH_4$

2. hydrogenotrophic methanogenesis: $CO_2 + 4H_2 \rightarrow 2H_2O + CH_4$ (Mach et al. 2015).

Following methanogenesis, CH_4 can be broken down and used as a carbon and energy source via CH_4 oxidation (Serrano-Silva et al., 2014; Smith et al., 2003). Oxidation of CH_4 to CO_2 is performed by methanotrophs, and there are two forms of CH_4 oxidation. High affinity oxidation is when methanotrophs consume CH_4 at concentrations close to that of the atmosphere, less than 12 ppm (Jardine et al., 2004). The bacteria that are responsible for high affinity oxidation are unknown (Dunfield et al., 1999). Low affinity oxidation is when methanotrophs operate in CH_4 concentrations higher than the atmosphere. These methanotrophs don't use CH_4 from the atmosphere, but are able to absorb 90% of the CH_4 produced by methanogens in the same environment (Jardine et al. 2004). Any CH_4 produced by methanogenesis that is not consumed via methanotrophy will eventually be emitted to the atmosphere, making that environment a source of CH_4 to the atmosphere.

The global warming potential (GWP) index was developed to understand the impact of different GHGs on climate change. This index translates different GHGs into equivalent terms (converting all GHGs into CO_2-equivalent units), allowing direct comparison of the effect of different GHGs on climate change. The climate change impact of CH_4 is determined by the GWP of CH_4. The two determinants of the GWP are the ability of a gas molecule to absorb energy and the molecules lifespan in the atmosphere (Myhre

et al., 2013). CH_4 has a relatively short lifetime of 12.4 years (Myhre et al., 2013), but its ability to retain heat energy is high. Over the standard 100-year period, methane has a global warming potential 28 to 34 times higher than CO_2 (Myhre et al., 2013). To add to its climate change impact, CH_4 also oxidizes to CO_2 (Myhre et al., 2013), simply converting form into a different GHG. Given that there are different processes and organisms responsible for producing and consuming CH_4 in soil environments, an improved understanding of the environmental factors that affect CH_4 dynamics, and how these factors are affected by urbanization, is needed.

Soil moisture

Understanding the role of soils in climate change requires an understanding of specific processes and factors that affect GHG flux in soils. Soil moisture is one factor that influences soil CH_4 flux (i.e., the production or consumption of CH_4). In a study done by Wang et al. (2014), positive CH_4 flux was linearly related to water filled pore space of the soil. A positive flux represents net emissions of GHGs from soil, whereas a negative flux represents a consumption of GHGs. CH_4 uptake (consumption, a negative flux) in soils is high when the water filled pore space is low because the diffusion of CH_4 into the soil is higher in unsaturated soil conditions (Wu et al., 2010), leading to decreased (or negative) CH_4 fluxes. Higher rates of diffusion are seen in coarser and dry soils, showing that moisture and texture are highly influential in the uptake of CH_4 (Castro et al., 1995; Smith et al., 2000), potentially causing increased CH_4 flux. Further, the overall activity of microbes is determined by moisture conditions. Methanogens are more active under flooded conditions compared to dry soil conditions, largely due to increased anoxic conditions in the soils (Watanabe et al., 2009). CH_4 production is reduced by drainage because methanogen growth is suppressed and there is an increase in the methanotrophic population occurring simultaneously (Ma and Lu, 2011). CH_4 production occurs in anaerobic conditions and it correlates positively with soil humidity (Gao et al., 2014, Smith et al., 2003).

In general, the rates of CH_4 uptake in soils have decreased over the years in both urban and forested soils, subsequently increasing CH_4 flux rates. From the late 1990s to the mid-2010s, net CH_4 uptake decreased by 62% in urban forests and by 53% in rural forests in Baltimore, Maryland (Ni and Groffman 2018). This study suggested that the increase in CH_4 flux is driven by increases in precipitation and hydrological flux. As climate change and urbanization are projected to combine for increased precipitation frequency and intensity in urban areas (Jauregui and Romales 1996), it is likely urbanization increases CH_4 flux via increased soil moisture. Soil moisture concentration is affected by physical factors such as pore space and texture. Processes that alter these physical properties of soils in urban areas, such as compaction or organic soil amendments, will likely affect CH_4 flux as well.

Nitrogen availability

Processes that increase nitrogen (N) availability in soils, such as ammonium (NH_4^+) fertilization, impact CH_4 uptake in soils. There have been multiple studies that have shown that NH_4^+ fertilization can reduce CH_4 consumption by 50-85% in a variety of soils (Bowden, Stevens, Steudler, 2000; Steudler et al., 1989; Hutsch, Webster, Powlson, 1994; Moiser and Schmiel, 1991). The decrease in CH_4 uptake when NH_4^+ has been added to soil is linked with the catalyzing enzyme methane monooxygenase (MMO). There is substrate competition between NH_4^+ and CH_4 at binding sites of MMO in the first step of CH_4 oxidation pathway, this results in reduced CH_4 oxidation (consumption) due to enhanced NH_4^+ oxidation (Bedard and Knowles 1989; Topp and Pattey, 1997). Long term fertilizer inputs have also been suggested to cause a decrease in methanotrophs due to niche competition with nitrifying bacteria (Gulledge et al., 2004). In other studies, elevated NH_4^+ concentrations significantly reduce low affinity CH_4 oxidizers, but high affinity CH_4 oxidizers were not affected (Jang et al., 2011; Reay and Nedwell, 2004).

Despite these previous studies showing inhibition of CH_4 oxidation in response to nitrogen (N) fertilization, the relationship between N availability and CH_4 uptake is not conclusive based on results

from various studies. In a study of forest soil N and CH_4 dynamics, Zhang et al. (2014) measured N

deposition in rainfall at a rural, suburban, and urban forest site. Both the suburban and urban forest had

higher N deposition than the rural forest. Increased N availability in soils due to increased atmospheric N

deposition led to a decrease in CH_4 uptake at the urban forest site (Zhang, 2014). Other studies have

suggested that there is increased potential for atmospheric N deposition in urban areas (Lovett et al.,

2000; Cape et al., 2004), potentially impacting CH_4 dynamics in urban soils. However, N additions do not

always result in a reduction of CH_4 uptake by forest soils. One study of the effect of N additions at a

forest site showed that increased N availability did not affect CH_4 fluxes, although there were

contradicting trends found in different years of the study (Ambus and Robertson 2006). In 1995, the

coniferous forest site showed an increasing CH_4 uptake trend with N additions, but in 1996 the opposite

was true. Further studies need to be done to determine the consistency of the effect of N addition on

CH_4 uptake.

Nitrous Oxide

N_2O is formed as an intermediate product during the microbial processes of denitrification and

nitrification. Denitrification is an anaerobic process that reduces nitrate (NO_3^-) and nitrite (NO_2^-) to

nitrogen gas (N_2). NO_3^- and NO_2^- can be reduced to N_2O through a series of reductions by denitrifying

bacteria (Ji et al., 2015). N_2 is the final product of denitrification, but under certain oxygen (O_2)

concentrations the reduction of NO_3^- and NO_2^- could halt at N_2O because enzymes that carry out

different steps of the denitrification pathway (i.e., nitrate reductase, nitrite reductase, nitric oxide

reductase and nitrous oxide reductase) have different levels of O_2 tolerance (Korner and Zumft, 1989).

As an intermediate in denitrification, the amount of N_2O produced and the $N_2O/(N_2+N_2O)$ ratio is used to

understand N_2O fluxes from soils (Cuhel et al., 2010). The total amount of N_2O produced provides an

estimate of overall GHG flux, whereas $N_2O/(N_2 + N_2O)$ is a measure of denitrification efficiency (i.e., how

likely was it for denitrification to run to completion). Nitrification is another microbially-mediated

process that can produce N_2O. Nitrification is an aerobic process that oxidizes ammonia or ammonium (NH_4^+) to NO_3^- by microbes known as ammonia oxidizers with N_2O as a byproduct (Anderson, 1964). While both of these processes (nitrification, denitrification) can produce N_2O as a by-product, they also consume N_2O if run to completion. The known sink in the environment for N_2O is an enzyme called respiratory nitrous oxide reductase and it is found in denitrifying bacteria that reduce N_2O to N_2 (Thomson et al., 2015). It catalyzes the final stop in denitrification. However, if nitrous oxide reductase is exposed to O_2, it will result in loss of N_2O reductase activity (Thomson et al., 2015).

Similar to CH_4, the climate change impact of N_2O is determined by its global warming potential (GWP). N_2O has an atmospheric lifetime of 114 years and over the standard 100-year period has a GWP 265-298 times that of CO_2 (Myhre et al., 2013). In addition to warming potential, N_2O has been recognized to be the most important gas related to ozone depletion (Ravishankara, Daniel, and Portmann, 2009). This is especially true as chlorine and bromine emissions have decreased after the Montreal Protocol (an international treaty designed to protect the ozone layer by phasing out substances responsible for ozone depletion) was passed to eliminate anthropogenic emissions of chloroflurocarbons and other halogen-containing chemicals (Wang et al., 2014). N_2O reacts with O_2 radicals in the stratosphere to form nitrogen monoxide, which is involved in the depletion of the stratospheric ozone (Crutzen, 1981). Increasing N_2O concentrations in the atmosphere could result in a decline of 2-4% of the total ozone column by the end of this century (Portmann and Solomon, 2007).

Soil Moisture

Soil moisture poses both direct and indirect controls on the production of N_2O. N_2O production is controlled by soil moisture because soil water content acts as a barrier to O_2 transport through soils (Kroeckel and Stolp, 1986). As O_2 is an inhibitor for denitrifying enzymes (Knowles, 1992), increased soil moisture will reduce O_2 concentrations, providing anoxic environments for denitrification to occur.

However, the O_2 concentration that results in inhibition varies on species of denitrifying bacteria (Revsbech and Sorenson, 2013). Under anoxic conditions, N_2O production increases, but is then followed by a conversion to N_2 by the enzyme N_2O reductase (Firestone et al., 1980). N_2O consumption is related to the O_2 sensitivity of the enzymes of the denitrification pathway. The O_2 sensitivity of the enzymes is inversely proportional to the degree of substrate oxidation and increases in the order of: NO_3 reductase < NO_2 reductase < NO reductase < N_2O reductase (Dendooven and Anderson 1994, McKenney et al. 1994, Joye and Hollibaugh 1995). Conditions need to be anoxic for denitrifying bacteria. Therefore, at very high soil water content which limits O_2 diffusion, N_2O is reduced to N_2 (Ruser et al., 2006). In anoxic conditions, soils can act as a sink for N_2O (Letey et al., 1981).

Despite the importance of soil moisture on N_2O dynamics, a study of N_2O flux did not show a seasonal pattern (Groffman et al., 2009). The study did however show higher N_2O flux in the years 2003 and 2004, this was attributed to high amounts of precipitation those two years relative to 2001, 2002, and 2005. Precipitation in 2003 was 148% of the normal precipitation and in 2004 it was 123%. Despite this annual increase in precipitation and N_2O flux, seasonal changes in soil moisture were not associated with seasonal changes in N_2O flux (Groffman et al. 2009). In arid regions like Colorado, Kaye et al. (2004) found that urban grassland soils emitted 10 times more N_2O than native grassland soils, the main difference being the urban grassland soils received irrigation and fertilizer and the native grassland didn't. High N_2O fluxes were also measured in an urban turfgrass in Illinois in response to a large rainfall event following turfgrass fertilization (Horgan et al. 2002). Furthermore, N_2O fluxes from urban lawns were higher than native ecosystems, and this difference was particularly evident following irrigation of urban lawns (Hall, Huber, and Grimm 2008). Prior to wetting, the N_2O flux was 21 to 25 μg N m^{-2} h^{-1} and after wetting N_2O ranged from 18 to 80 μg N m^{-2} h^{-1}. This was four to six times higher than the native ecosystems. Overall, these results suggest that soil moisture is a major driver of N_2O emission in urban landscapes (Bijoor et al., 2008).

Soil pH

Soil pH is another important environmental factor known to influence N_2O production. Denitrification rates are positively related to pH; denitrification rates increase with increasing pH values up to an optimum pH (Cuhel et al, 2010). In a pure culture of *Pseudomonas* species, the optimum pH was 7.0 to 7.5 for denitrification (Knowles, 1982; Thomas, Lloyd, Boddy, 1994). Further, the $N_2O/(N_2+N_2O)$ ratio decreases with increasing pH, showing that the denitrification process is more efficient and runs to completion more regularly at higher pH (Tate, 1995). In contrast, Cuhel et al. (2010) found that the $N_2O/(N_2+N_2O)$ ratio increased with decreasing pH due to the changes in denitrification activity, but not in N_2O production. The composition of the denitrifying community is controlled by pH, meaning that the diversity, abundance, and transcriptional activity of ammonium oxidizing bacteria and archaea is influenced by pH (Nicol et al., 2008). Low pH values were reported to hinder N_2O reductase, the enzyme that reduces N_2O (Richardson et al., 2009). Both Maljanen et al. (2012) and Weslien et al. (2009) made the conclusion that high N_2O flux was most likely because of the low soil pH which limited N_2O reduction. These results all suggest that pH does not affect the entire denitrification pathway, but can affect the $N_2O/(N_2 + N_2O)$ ratio, ultimately meaning that the same amount of denitrification would produce more N_2O at non-optimal pH values. If soils are acidic, increasing pH will increase denitrification efficiency, if soils are basic, reducing pH will have the same effect. This could relate back to microbial denitrifiers being adapted to specific soil pH and any changes could impact the rate of denitrification (Simek and Cooper, 2002).

Urban soils are generally characterized as having higher pH values compared to forest soil. In a study done by Asabere et al. (2018), forest soils had pH values of 5.7 ± 0.3 and nearby urban soils had pH values of approximately 7. There are multiple reasons for elevated pH in urban soils: the application of calcium or sodium chloride on roads and sidewalks for deicing, irrigation by calcium enriched water, and the release of calcium from the weathering from buildings (Bockheim, 1974). Groffman et al. (2009)

sampled different types of soils in the Baltimore metropolitan area in the summer of 2000 and found

that forest soil pH was 3.9, whereas urban grassland soil pH was 5.9. Based on the impacts of pH on

denitrification efficiency, increasing soil pH should reduce N_2O flux in urban environments, but there are

some conflicting findings. For example, Van Cleemput and Samater (1996) found that the reduction of

NO_2^- to nitric oxide (NO) and N_2O can occur in slightly acidic soils and not at higher pH. Based on the

differences in findings, the effect of soil pH on N_2O flux needs to be further researched.

Soil temperature

Soil temperature is another factor that plays a role in microbial metabolism and N dynamics,

particularly influencing N_2O formation via nitrification and denitrification. Optimal temperatures for

nitrification range from 15-35 ℃ while it is inhibited at temperatures above 40°C and below 5°C

(Alexander, 1977). The optimal temperatures for denitrification range from 25 to 35°C (Kesik et al.,

2006). A variety of studies have shown that denitrification activity and temperature are positively

correlated (Maag and Vinther, 1996; Godde and Conrad, 1999). There are two primary reasons for

temperature influencing denitrification: the enzymatic processes in N_2O production are directly

increased by warmer temperatures, and temperature can indirectly influence N_2O production by

increasing soil respiration which can reduce O_2 concentration in soils and can lead to anoxic conditions

(Butterbach-Bahl et al., 2013).

The conversion of land from either natural or agricultural lands to urban land is associated with

changes in energy fluxes, water, nutrient cycling, albedo and heat capacity (Pataki et al., 2006; Pouyat et

al., 2007). The urban heat island effect (described in the introduction section, above) increases

temperatures of urban environments. That has the potential to influence N_2O flux from soils by altering

nitrification and denitrification. Surprisingly, N_2O fluxes associated with nitrification have decreased with

increasing temperature whereas denitrification associated N_2O fluxes show the opposite pattern (Maag

and Vinther, 1996). Higher temperatures enhance microbial respiration, which results in the depletion of O_2, thus favoring denitrification while inhibiting nitrification (Castaldi, 2000), this could mean a higher N_2O flux in urban soils versus forested soils because denitrification contributes more to N_2O flux than nitrification. However, this needs to be further studied as the effects of temperature on these processes are still unclear because of the few data available (Barnard et al., 2005).

Carbon Dioxide

As with other GHGs, soils act as both an important source and sink for CO_2, and CO_2 dynamics of soils are largely driven by the soil microbial community. CO_2 is removed from the atmosphere and subsequently synthesized into organic matter by carbon (C) fixing autotrophic organisms via photosynthesis. This process of converting CO_2 into organic matter represents a sink for CO_2 that is driven globally by CO_2-fixing plants and photo and chemoautotrophic microbes (Lu, Friedrich, Conrad, 2005). Once that C is fixed though, the C sequestration capacity of an ecosystem is strongly regulated by the C residence time in different pools (Schimel et al., 1994, Joos et al., 1996, Luo and Reynolds 1999, Thompson and Randerson 1999). Soil organic matter represents a major pool of C for terrestrial ecosystems, so changes in soil organic matter will greatly affect terrestrial C sequestration. If a large amount of fixed C is cycled rapidly through 'fast' pools of SOM then there will be minimal C sequestration. In contrast, if fixed C is cycled through 'slow' pools of SOM, then the carbon sequestration for that ecosystem is large (Xiao, 2015). Whether SOM is cycled through 'fast' or 'slow' pools depends upon a combination of physical, chemical, and biological factors unique to each ecosystem.

Soil respiration is a general term used to represent three separate (but related) soil processes that use fixed organic C as an energy source and subsequently release it as CO_2 soil: microbial respiration, root respiration, and faunal respiration. Soil microbial respiration is the aerobic microbial

decomposition of soil organic matter and is used by microbial communities to obtain energy and

produces CO_2 as a byproduct. Root respiration is also referred to as autotrophic respiration and this

includes associated rhizospheric microbial respiration. Soil faunal respiration and microbial respiration

not associated with the rhizosphere are referred to as heterotrophic respiration (Li, Xu, Zou, 2005).

Heterotrophic respiration uses organic matter produced previously that has been sequestered in the soil

for some period of time, resulting in the loss of accumulated carbon from the soil. In contrast,

autotrophic respiration uses C recently fixed by primary producers for energy, basically stopping this C

from being sequestered in the soil in the first place. Knowing the amount of heterotrophic respiration

allows for the quantification of the exchange of C between the atmosphere and soil or the rate of C

sequestration (Saurette, Chang, and Thomas, 2007). Global soil respiration data from 1990 to 2014

shows that the global heterotrophic respiration: total soil respiration ratio has increased from 0.54 to

0.63, this is in response to global environmental changes and is consistent across various ecosystem

(Bond-Lamberty et al. 2018). This increase in the heterotrophic: total respiration ratio shows that the

increase of CO_2 loss to the atmosphere is higher than the CO_2 uptake of plants, likely due to the

increased activity of microbes. Ultimately, this pattern suggests an overall reduction in C sequestration

and this sustained trend is important to note considering the role of CO_2 in climate change.

As described previously, global warming potential (GWP) is used to compare the effect of GHGs

over time and these are shown as CO_2 equivalents. Therefore, by definition the GWP for CO_2 is 1. CO_2 is a

long-lived pollutant compared to CH_4 and N_2O, some CO_2 can remain in the atmosphere for thousands of

years (Pierrehumbert, 2014). Before the Industrial Revolution, the CO_2 concentration was consistently

280 ± 10 ppm for thousands of years. In the decades following the Industrial Revolution, the

concentration has steadily increased reaching 410 ppm as of mid-2018 (ESRL, 2018). This increase in CO_2

concentration is caused by anthropogenic emissions with about three quarters of the increase coming

from fossil fuel burning and the rest from land use change (Prentice et al., 2001). This further increases

the concern of CO_2 emissions in urban areas because urban areas are largely responsible for anthropogenic emissions and they will likely continue to rise. Understanding the factors controlling CO_2 dynamics in soils, and the effects of urbanization on these dynamics are vital to mitigating future increases in GHG emissions from urban soils.

Litterfall input

Plant litterfall is a primary pathway for nutrients and organic matter to return to the soil in terrestrial systems. Litter provides the C source for soil respiration (Reynolds and Hunter, 2001) and the input of fresh organic matter (FOM) accelerates mineralization by priming the soil microbial community to process soil organic matter (Blagodatskaya and Kuzykavov, 2008). The mechanisms of priming are based off of two theories: FOM provides the energy source for microorganisms to synthesize extracellular enzymes that ease SOM mineralization (Kuzyakov Friedel, Stahra, 2000) and FOM stimulates microorganism growth and nutrient limitation will increase mineralization of otherwise unavailable SOM (Blagodatskaya and Kuzykavov, 2008). Any changes in the quality and quantity of litter inputs will alter the accumulation or loss of SOM (Boone et al., 1998). Increased inputs of decomposable organic matter lead to increases in CO_2 flux because microbes are stimulated by the organic matter (Kuzyaov, Friedel, Stahr, 2000). For example, a litter manipulation experiment showed that CO_2 flux increases when fresh litter is added and is reduced when fresh litter is excluded (Prevost-Boure et al., 2010). In addition to simply providing an organic matter source, total soil respiration increases from increased litter input because litter input promotes rhizosphere respiration and rhizosphere activity (Subke et al., 2004).

Litter decomposition in urban ecosystems is different from other ecosystems due to the distinct physical, chemical, and biological environment (Carreiro et al., 1999). Depending on the background and management of the urban site, the content of organic matter could be higher or lower than reference

ecosystems (Vodyanitskii, 2014). In urban areas plant litter is often removed for aesthetic reasons. This

removal of litter (i.e., leaves, grass clippings) affects the ability of the soil to retain nutrients and

sequester C (Sayer and Tanner 2010). Particularly during the early stages of urban construction, strong

contamination, or no vegetative cover on the site, can cause urban soils to have decreased organic

matter content (Vodyanitskii, 2014). However, there are urban areas that import C and stimulate

primary productivity by residential landowners fertilizing their lawn, adding compost or organic fertilizer

and mulch. As a result, this would lead to high rates of soil respiration in urban areas (Beesley, 2014;

Chen et al., 2014). The addition of soil amendments, like fertilizer and mulch, could lead to CO_2 flux that

could be up to twice as much as rural forests (Raciti el al., 2014). For forested soils, soil respiration

increased with litter input regardless of litter type (Cleveland et al., 2007; Fang et al., 2015). Overall,

litter inputs drive respiration and organic matter dynamics in soils, but urbanization can alter the

amount and quality of litter accumulating in soils.

Soil temperature

Soil temperature is a factor that is always discussed with soil respiration because it is such an

important driver of CO_2 dynamics. Many studies have shown a positive relationship between CO_2 flux

and soil temperature (Almagro et al., 2009; Dilekoglu and Sakin, 2017). This temperature dependency is

seen in the seasonal variation of soil respiration. In a 2-year study done by Tao et al. (2016) soil

respiration increased from February to August and decreased from August to February. The seasonal

variation of soil respiration is linked to the seasonal changes of the fine root biomass and photosynthetic

supply (Widen and Majdi, 2001) and temperature directly influencing the activities of soil

microorganisms (Tao et al., 2016). Hicks Piers et al. (2017) showed the positive relationship of

temperature and CO_2 flux by heating a soil at a depth of 100 cm by 4°C resulting in an increase in annual

respiration by 34-37%. This warming of 4°C reflects the IPCC (2013) prediction that whole-soil profile will

warm 4°C by 2100.

This temperature effect on soil respiration suggests that urbanization may be indirectly increasing soil respiration by increasing soil temperature (George et al. 2007). Indeed, Groffman et al. (2006, 2009) found that urban forest soils respire more than rural forest and temperature was included as one of the factors that could be responsible for this increase. Shen, Shi, and Jiang (2013) linked higher urban soil respiration rates to higher average soil and air temperatures in urban areas as a consequence of the urban heat island effect. Similar to CH_4 and N_2O, it is clear that soil temperature is an important factor controlling biological processes and subsequent greenhouse gas emissions in soils.

Soil Moisture

Soil moisture is another environmental factor that impacts CO_2 flux in soils. There have been studies that have indicated that the CO_2 response to soil moisture relationship is either linear (Epron et al., 1998) or exponential (Davidson, Belk, Boone, 1998). Soil moisture affects a variety of processes, such as diffusion, osmoregulation, and enzyme dynamics, which all in turn affect soil respiration (Moyano, Manzoni, Chenu, 2013). However, soil moisture is often associated with soil temperature. Wildung et al. (1975) found that season variation in CO_2 flux was attributed to the temperature effect (increase in temperature increases soil respiration), but only when there was sufficient soil moisture to permit significant soil respiration. During the late spring and summer months when temperatures were above 15°C and water content was above 6-10%, CO_2 flux increased with increasing soil moisture content (Wildung, Garland, Buschbom, 1975), but moisture changes had little effect on soil respiration at temperatures lower than 5°C (Flanagan and Veum, 1974). Similarly, Tao et al. (2016) saw that soil respiration was positively correlated with soil moisture when soil moisture is in the range of 18-25%, but outside of this range there was no relationship. Along with the warming seen during the months of December to June, soils are also moving away from a dry state to the optimal soil moisture range for soil respiration above 18% (Tao et al., 2016). Both the increase in soil moisture and the increase in soil temperature, can work in conjugation to one another to increase soil respiration. It's apparent that soil

moisture is an important driver of CO_2 dynamics, but there appears to be a strong interaction with soil temperature.

Urban soils exhibit variable moisture content due to a combination of different factors that could increase or decrease water content in soil (Pouyat et al., 2010). In highly compacted soils, which are common in urban environments due to various activities and management practices, infiltration can be restricted because of the hydrophobic surfaces, surface crust formation, and high bulk density (Craul, 1992). There are however areas that are irrigated that interfere with drainage and can result in higher water content (Craul 1992: Pouyat et al. 2007). For example, urban forest sites studied by Groffman et al. (2009) had a wider range of moisture content and moisture content was higher than the rural forested sites. The impact of irrigation on CO_2 flux can be massive, as seen in a study in Phoenix, AZ. In arid regions such as Phoenix, AZ, soil moisture was the largest contributor to CO_2 flux and it could be 50 times higher in desert environments with irrigation (Koerner and Klopatek, 2002). Depending on the management of urban soils (e.g., irrigation) CO_2 flux could increase in urban soils.

Summary: Urbanization effects on GHG

Urbanization appears to increase CH_4 flux based on soil moisture and N availability factors. In general, both urban and forested soils have decreased CH_4 uptake as a result of climate change and urbanization, subsequently increasing CH_4 flux. However, this decrease in CH_4 uptake was more pronounced in urban soils than forested. Thus, urbanization appears to exacerbate this increase in CH_4 flux. The same can be said for N_2O flux based on soil moisture and soil pH factors, urban soils had higher N_2O flux, however further research should be done as processes regarding soil pH are still unclear from different findings. For CO_2, urbanization appears to increase CO_2 flux based on litterfall input, soil moisture, and soil temperature factors. Urban soils exhibit higher amounts of respiration compared to forested soils and this will only increase as areas are increasingly urbanized. Overall, urbanization

appears to increase GHG emissions from soils due to multiple different factors and effects on the three major GHGs.

Conclusion

As discussed in this literature review, soil processes that drive GHG dynamics are affected by biological, physical, and chemical properties. Soil moisture was a major driving factor for all GHG fluxes. Increases in soil moisture drives the production of GHG by microbes due to increasing anoxic conditions, which are required for multiple microbial processes that produce GHG. Soil temperature was another factor that drives the production of GHG because higher temperatures are favorable for microbial activity. Other factors like N availability, soil pH, and litterfall input were important factors for specific GHG. Nitrogen availability affects CH_4 uptake because of substrate competition between CH_4 and NH_4, increased N would hinder CH_4 uptake. Low pH results in higher N_2O flux because lower pH hinders N_2O reductase, the enzyme that reduces N_2O. And increased litterfall input increases CO_2 production because of priming of SOM.

The impact of these factors, and how they are related to climate change, depended upon the surrounding landscape, with urban and forested soils having different environmental conditions leading to differences in GHG dynamics. Urban soils typically have higher temperatures and pH than nearby forested soils. However other characteristics like litterfall input, N availability, and soil moisture were not easily distinguished between urban soils and forested soils. These would vary from one specific area to another because management of these soils are not all the same. This was shown by studies with contradicting findings, which means that the direct effects of various factors should be furthered researched while controlling for the impact of other factors that have not been taken into consideration when previous studies were performed.

Regardless of these differences, one thing that is known is that the impact of urbanization on soils will continue to rise because of the rate of urbanization. Since urban areas already account for more than 50% of GHG emissions, urban soils will continue to be heavily linked to GHG dynamics in the future. And climate change will continue to play a role on GHG dynamics because of the link between climate change and urbanization. Climate change and urbanization will continue to push environmental factors like moisture and temperature to extremes which could imply higher GHG fluxes and a positive feedback loop because the accumulation of GHG will lead to more warming.

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