# Coupling biogeochemical cycles in urban environments: ecosystem services, green solutions, and misconceptions

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**Urban green space is purported to offset greenhouse-gas (GHG) emissions, remove air and water pollutants, cool local climate, and improve public health. To use these services, municipalities have focused efforts on designing and implementing ecosystem-services-based "green infrastructure" in urban environments. In some cases the environmental benefits of this infrastructure have been well documented, but they are often unclear, unquantified, and/or outweighed by potential costs. Quantifying biogeochemical processes in urban green infrastructure can improve our understanding of urban ecosystem services and disservices (negative or unintended consequences) resulting from designed urban green spaces. Here we propose a framework to integrate biogeochemical processes into designing, implementing, and evaluating the net effectiveness of green infrastructure, and provide examples for GHG mitigation, stormwater runoff mitigation, and improvements in air quality and health.**

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Humans are increasingly influencing biogeochemical cycles at a global scale (Vitousek *et al.* 1997; Rojstaczer *et al.* 2001). From the publication of "man's role in changing the face of the Earth" (Thomas 1956) to the recent syntheses on climate change (eg Solomon *et al.* 2007), the role of cities in altered biogeochemical cycles has received increasing attention; for the first time in human history, more people live in urban areas than in the countryside (UN 2010). As cities concentrate increasing numbers of people, they also concentrate and transform energy, materials, and waste in small areas. Although constraining populations to smaller areas may have advantages for land and other resource uses (Anderson *et al.* 1996), techniques and approaches need to be developed to design more efficient and sustainable cities.

One approach to improving urban sustainability is to use "ecosystem services" to remediate pollution and other environmental problems (McPherson *et al.* 1998; Bolund and Hunhammar 1999; Nowak *et al.* 2002; McPherson *et al.* 2005; Oberndorfer *et al.* 2007). The Millennium Ecosystem Assessment (MA 2003) provided a framework for categorizing the societal benefits of ecosystems into different services: provisioning services (which provide food and materials), cultural services (which provide aesthetic and psychological benefits), and regulating services (which moderate environmental conditions and quality; Figure 1). Each of these services relies on fundamental ecological processes that are recognizable and in most cases measurable by members of the scientific community. Here we focus on the climate-, water-, and atmosphere-regulating services provided by planned urban green space, including but not limited to urban forests, parks, and gardens. Regulating services in green space are intimately linked to many fundamental biogeochemical processes, which are the biological and chemical processes that cycle and transform carbon (C), nutrients (eg nitrogen [N] and phosphorus [P]), water, and other materials in the environment.

There is a growing body of literature about the poten-

### In a nutshell:

- Many commonly cited environmental benefits of urban greenspace are still poorly supported by empirical evidence, adding to the difficulties in designing and implementing green infrastructure programs
- To date, there is little data showing that urban greenspace can reduce urban greenhouse-gas emissions or air and water pollutant concentrations
- There is evidence to support substantial reductions in urban runoff, local atmospheric cooling, and improvements in human health that do not appear to be related to pollutant concentrations
- Studies of urban biogeochemistry are greatly needed to improve greenspace design and monitor its effectiveness in meeting local environmental goals in different regions and urban settings

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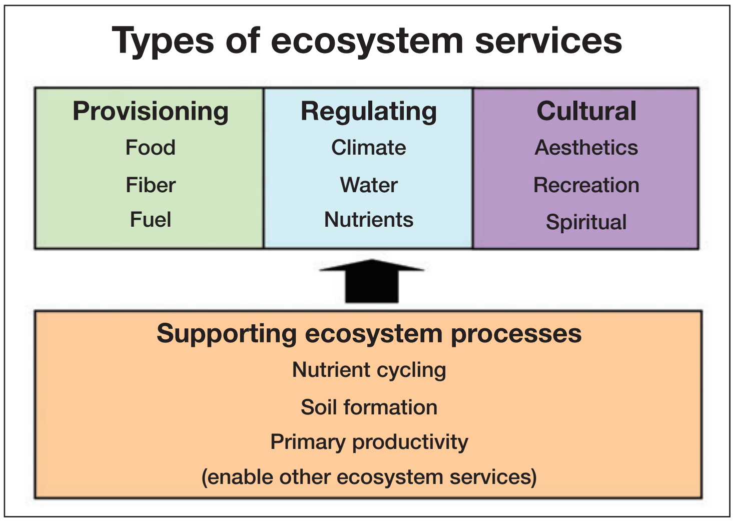


***Figure 1.** Ecosystem-services framework based on the Millennium Ecosystem Assessment (MA 2003).*

tial benefits of designed urban green space (Bolund and Hunhammar 1999; Nowak *et al.* 2002; McPherson *et al.* 2005; Oberndorfer *et al.* 2007), which we refer to here as "green infrastructure". These benefits are often generalized and undifferentiated by climatic zone, local vegetation and soils, local fiscal capacity, public interest, or institutional or cultural values (eg Schwab 2009; Lozanova 2010). Given the unexpected lack of empirical data evaluating the effectiveness of specific, place-based green infrastructure (Pincetl 2007; Park *et al.* 2009; Pincetl 2010a, b), we wish to draw attention to the gap between the anticipated benefits of green infrastructure and the

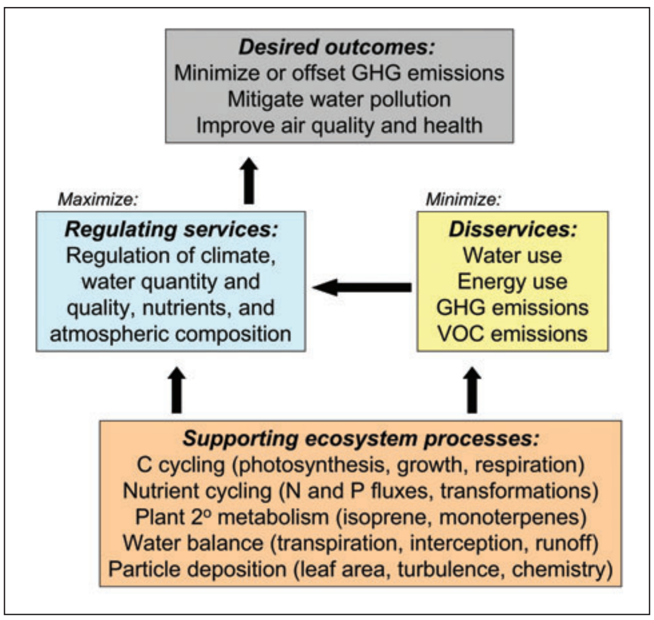


***Figure 2.** Framework for incorporating ecosystem services into improving environmental outcomes in cities. Both ecosystem services and disservices (benefits and costs of green space, respectively) must be identified for a given desired outcome. To quantify these services and disservices, we must further relate them to measurable supporting ecosystem processes.*

implementation and evaluation of its performance in specific contexts. In addition, we consider the potential environmental costs, or "disservices" – the negative consequences or tradeoffs of implementing green infrastructure. Examples of ecosystem disservices are urban plantings or landscape designs that increase allergens; promote invasive plants, host pathogens, or pests; inhibit human mobility and safety; or increase greenhouse-gas (GHG) emissions (Lyytimäki *et al.* 2008). We also highlight the role of biogeochemistry in improving our ability to quantify both ecosystem services and disservices as a means of evaluating the effectiveness of urban green space in meeting environmental goals.

By definition, ecosystem services have societal relevance: they provide benefits that humans want or need. For example, municipalities may wish to reduce or offset GHG emissions, decrease the volume or pollution load of stormwater runoff, or improve air quality. These desired outcomes must be identified and defined as a first step in evaluating the effectiveness of green designs. Once the desired environmental outcomes have been defined, the relevant ecosystem services can be identified – but to quantify these services, researchers must link them to measurable ecosystem processes. In the case of water regulation, removal of nitrate ($NO_3^-$) from stormwater is a desired regulating ecosystem service, but understanding and measuring urban aquatic nitrogen cycling (eg sources, sinks, fluxes) is necessary to quantify nitrate removals by proposed or implemented green infrastructure. These processes are often related to other coupled biogeochemical processes in C, water, or other nutrient cycles. Quantifying potential ecosystem disservices is also essential for assessing the net effectiveness of green infrastructure, and may also be related to one or more biogeochemical processes. For instance, a tradeoff of nitrate removals in streams may be the release of the GHG nitrous oxide ($N_2O$) from denitrification. The process of identifying desired environmental outcomes and ecosystem services (and undesired disservices), as well as their underlying biogeochemical processes, is demonstrated in Figure 2, through examples of (1) offsetting GHG emissions, (2) mitigating urban runoff and water pollution, and (3) improving urban air quality and human health. We discuss these examples further below.

## Greenhouse-gas emissions

Managers and planners in cities are increasingly concerned about climate change and its effects on residents because of direct threats such as coastal flooding and extreme heat events (Zahran *et al.* 2008). In response, many cities worldwide are voluntarily reducing GHG emissions (Betsill 2001) and signing on to agreements such as those developed by the Cities for Climate Protection program (www.iclei.org/index.php?id=10829) of the ICLEI Local Governments for Sustainability. Biological C sequestration in woody plants and in soils has been suggested as a potential mitigation tool for meeting these goals (McHale *et al.* 2007). Approximately 50% of live plant biomass is C, and dead plant biomass is incorporated into soil C pools. Increasing woody plant biomass and soil C removes and stores C that would otherwise contribute to global warming. Tree-planting programs to sequester C are an appealing option for climate-change mitigation because trees have additional benefits resulting from their local cooling and shading effects, provision of habitat for native and rare species, and cultural ecosystem services. Here we review the potential effectiveness of these programs specifically for mitigating the effects of climate change.

### *Regulating services*

There are two climate-regulating services of C sequestration: (1) direct removal of carbon dioxide ($CO_2$) from the atmosphere and (2) indirect effects of vegetation on local cooling through shading and transpiration in warm climates (McPherson 1992; Bolund and Hunhammar 1999; Nowak *et al.* 2002; Simpson 2002). Thus, coupling of C, water, and energy cycles is integral to impacts of urban vegetation on climate. Carbon storage in urban trees and soils (Nowak and Crane 2002; Pouyat *et al.* 2006) and other urban C pools, such as building materials, has been estimated for several cities in the US (Churkina *et al.* 2010). Estimates of C sequestration include uncertainties due to the unique forms of urban trees, which are often heavily pruned or open grown and may have restricted rooting volumes. Therefore, the equations that predict total tree biomass, which are usually derived from measurements of natural trees, do not apply. In addition, the specific location of trees relative to building aspect is an important determinant of whether trees will cool buildings in summer in temperate latitudes. Local cooling results in lower demand for air conditioning, which can reduce GHG emissions related to electricity generation (Akbari 2002; Simpson 2002). However, tree water use – an important component of local cooling – has been shown to be highly species specific in urban forests (McCarthy and Pataki 2010; Pataki *et al.* in press), and the water use of many urban species is unknown. In general, the exact location of trees as well as their size, planting density, management (especially irrigation), and species are important determinants of local cooling effects.

### *Ecosystem disservices*

Direct C sequestration is only one component of decreasing GHG emissions. Management of urban trees requires energy for planting, pruning, watering, fertilizing, repairing sidewalks and road surfaces, and removing debris (McPherson *et al.* 2005; Pataki *et al.* 2006). Also, emissions of non-$CO_2$ GHGs, including nitrous oxide, can be quite large from some types of urban land cover, such as lawns and turfgrass, relative to emissions from natural ecosystems (Kaye *et al.* 2004; Groffman *et al.* 2009). Even in largely intact embedded urban forests, modified climate and other disturbances can result in higher emissions and smaller sinks (methane [$CH_4$] uptake) of non-$CO_2$ GHGs (Groffman *et al.* 2006; Groffman and Pouyat 2009). In addition, in cities located in semiarid areas where urban landscapes are irrigated, local cooling effects may come at a considerable water cost. In California, for example, 30–70% of the municipal water supply is used for outdoor irrigation (Gleick *et al.* 2003). Water use by irrigated urban vegetation constitutes both an ecosystem service – because of the regulating effect of transpiration on the water cycle – and a disservice – because of the scarcity of water resources in many regions (Figure 3).

**Water tradeoffs in irrigated urban ecosystems**

*Ecosystem services (benefits)*

- Cooling of local climate
- Regulation of the water cycle
- Erosion control
- Fire breaks
- Runoff mitigation
- Supports other benefits (eg habitat, aesthetics)

*Ecosystem disservices (costs)*

- Depletion of scarce water resources
- Excess runoff contributes to water pollution and eutrophication

***Figure 3.*** *Ecosystem services and disservices related to irrigating green space in arid and semiarid cities.*

### *Net effectiveness*

Urban C sequestration estimates are rarely compared with urban GHG emissions to assess the potential importance of the former as a mitigation strategy, largely because the necessary data have been unavailable. The absence of these comparisons makes it very difficult for cities to evaluate whether they are meeting their GHG reduction goals (Pataki *et al.* 2006). However, Pataki *et al.* (2009) simulated the trajectory of $CO_2$ emissions under urbanization scenarios in Utah's Salt Lake Valley and found that doubling the tree-planting density would offset less than 0.2% of total annual $CO_2$ emissions after 50 years. A similar calculation comparing urban GHG emissions and primary productivity (the upper limit on biological C sequestration) is shown in Figure 4. These analyses are important because cities often have very limited resources to implement environmental programs such as GHG reductions (Betsill 2001; Pincetl 2010a). Although urban tree-planting and green-space programs have many benefits besides C offsets, direct C sequestration in plants and soils is not likely to be an effective means for reaching local GHG

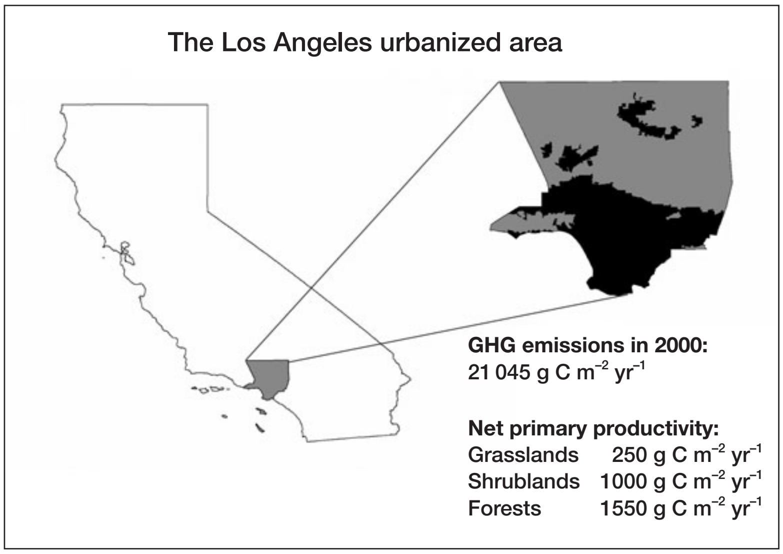


***Figure 4.*** *Urban areas (black) as defined by the 2000 US census within Los Angeles County, California (gray). GHG emissions from the urban area (Ngo and Pataki 2008) are compared with global average values of net primary productivity (NPP), an upper limit on biological C sequestration, for the ecosystems that occur in this region: grasslands, Mediterranean shrublands, and forests (Saugier* et al. *2001). Because much of the urban area is covered in built and impervious surfaces, the actual NPP of this region is likely to be lower than these values. In addition, only a fraction of NPP goes to long-lived C storage in wood and soil, so it is not possible for biological C sequestration to substantially offset GHG emissions in this urban area.*

reduction targets. The cooling effects of urban forests that lead to reduced energy use are likely to be more important than GHG reduction, but should be better quantified. The energy balance of several urban environments has been characterized (Grimmond and Oke 1999; Arnfield 2003), but previous studies often lacked explicit consideration of the biological processes that influence cooling, which are species-, location-, and management-dependent (Bush *et al.* 2008; McCarthy and Pataki 2010). Because urban areas contain distinctive mixes of species subject to disturbed environments with no natural analogs, there are few current datasets and modeling approaches to predict locally specific cooling effects. Though in general we know that urban green space often contributes to localized cooling, designing specific plantings and landscapes to optimize cooling effects and minimize costs, such as those from irrigation, still requires additional research (Table 1).

## ■ Urban water runoff and pollution

Cities often import water from surrounding areas in addition to converting land cover from vegetated surfaces to buildings, pavement, and other impervious surfaces. This land-cover change radically alters the pathways and magnitude of water and pollution flows into, within, and out of urban systems (Bonan 2002; Pouyat *et al.* 2007), all of which affect biogeochemical cycles (Walsh *et al.* 2005b; US EPA 2008a). Urban landscapes with 50–90% impervious cover can lose 40–83% of rainfall to surface runoff. In contrast, forested landscapes lose about 13% of rainfall inputs to runoff from similar precipitation events (Bonan 2002). The acceleration of water flow in cities adversely affects local water, energy, and biogeochemical cycles through land subsidence (Lerner 1996), saltwater intrusion from aquifer pumping (Dausman and Langevin 2005), erosion and degradation of stream channels (US EPA 2008a), decreased evapotranspirational cooling from land, and increases in water pollution (ie nutrient and contaminant loading) from direct discharge of pollutants and impairments in stream biogeochemical processes (Groffman *et al.* 2002; Sweeney *et al.* 2004; Walsh *et al.* 2005b).

### *Regulating services*

Ecosystem-services-based approaches have been used both to regulate the urban water cycle by reducing the amount of stormwater runoff and to improve water quality by removing pollutants from runoff. In urban streams, nutrient retention can be increased by adding coarse woody debris, constructing in-channel gravel beds, and increasing the width of vegetation buffer zones and tree cover (Booth 2005). Vegetated landscapes designed to absorb water – such as linear features (bioswales), green roofs, and rain gardens (eg Figure 5) – are other means of reducing both the amount of urban stormwater runoff and its pollution load (Clausen 2007; Shuster *et al.* 2008). For example, green roofs can retain 25–100% of rainfall, depending on rooting depth, roof slope, and the amount of rainfall (Oberndorfer *et al.* 2007). Green roofs may also delay the timing of peak runoff, alleviating stress on storm-sewer systems. Similarly, rain gardens and bioretention filters can reduce the volume of surface runoff (Clausen 2007; Shuster *et al.* 2008). However, few studies have demonstrated that these features improve water quality.

### *Ecosystem disservices*

Green-roof runoff may contain higher concentrations of nutrient pollutants, such as N and P, than are present in precipitation inputs (Oberndorfer *et al.* 2007), although more studies need to be conducted to confirm these findings. It may be costly to build or retrofit existing buildings and landscapes to support roof gardens or other forms of stormwater green infrastructure. Moreover, in arid regions, many green infrastructures require irrigation in the dry season, leading to various disservices (see Figure 3).

### *Net effectiveness*

The impacts of projects designed to improve stream nutrient retention have rarely been measured or have shown mixed results (Bernhardt *et al.* 2005). Natural stream habitats remove nitrate from waterways, but restoring these ecosystems in urban landscapes is difficult because of increased water-table depths, which can prevent establishment of an anaerobic zone for denitrification (Groffman *et al.* 2002). Engineered features designed to improve stream water quality tend to be short term, because excessive runoff can damage or remove gravel beds, debris, and/or planted vegetation within 1 or 2 years (Booth 2005). This suggests that a more permanent solution to stream water-quality problems lies in reducing runoff (Walsh *et al.* 2005a; Shuster *et al.* 2008; US EPA 2008b).

Rain gardens, bioswales, and other green infrastructures have the potential for reducing urban runoff. Dietz and Clausen (2005) provided the most thorough field test of both water retention and pollutant removal in rain gardens over a 1-year period: nearly 99% of water inputs were absorbed, even in winter. However, only 36% of N was retained, which is less than that of natural streams but greater than that associated with impervious surfaces. In that study, P discharge from rain gardens was greater than inputs, but this may have been due to initial soil disturbance, as P concentrations declined over time. Davis *et al.* (2001) conducted laboratory soil column experiments using synthetic (ie artificially produced) runoff and found similar results for N retention, but greater P retention (81%) than that in the field study by Dietz and Clausen (2005). Others report high nitrate fluxes from rain gardens, even when conditions are favorable for denitrification, a major pathway for nitrate removal (Seymour 2005). More field studies are needed to assess nutrient and contaminant transformations in stormwater mitigation features to understand how their designs can be effectively optimized.

## ■ Air quality and human health

Urban trees intercept the transport of air pollutants (Grantz *et al.* 2003) and, as a result, municipalities are interested in increasing urban tree cover to improve air quality and human health (Pincetl 2010b). We discuss research related to the effects of trees on air quality, which is directly linked to multiple biogeochemical processes and to other indirect effects of urban vegetation on human health. The impact of trees on atmospheric composition is related to ecosystem processes such as primary productivity and canopy physiology, but is also indirectly linked to human health via the provision of cultural benefits.

### *Regulating services*

Pollutant uptake by urban trees has been modeled through the use of tree physiology and calculated tropospheric ozone ($O_3$) and nitrogen dioxide ($NO_2$) uptake, and gaseous and particulate deposition to vegetation surfaces (McPherson *et al.* 1997; Bolund and Hunhammar 1999; Nowak *et al.* 2002; Simpson 2002). However, the effects of vegetation on local air quality or concentrations of transported atmospheric pollutants have not been well quantified. Escobedo and Nowak (2009) modeled air-pollutant uptake by various surfaces, including vegetation, in Santiago, Chile, and found that uptake was not significantly correlated with health metrics. In addition, less than 2% of particles with a diameter of 10 microns or less ($PM_{10}$) were removed in areas with the highest tree cover (26%), even though their model used a conservative resuspension rate (the amount of particulate matter returned to the atmosphere) of 50%, which has been questioned by Whitlow (2009). Based on these studies, the potential for vegetation to substantially improve air quality is probably limited (Panel 1). Urban vegetation could indirectly affect $O_3$ formation through reductions in the heat island effect, although this is also largely unquantified.

In contrast, the contribution of urban vegetation to human health has been documented. Compared with those recuperating in hospital rooms with windows facing a brick wall, post-operative patients with views of trees had shorter hospital stays, took fewer analgesic doses, and had fewer negative evaluations from nurses (Ulrich 1984). The presence of trees can reduce crime rates in public-housing complexes (Kuo and Sullivan 2001), increase safety and security (Kuo *et al.* 1998), and improve alertness in children with attention deficit disorder (Faber Taylor and Kuo 2009). The body mass index of children showed an inverse relationship to exposure to green space (Bell *et al.* 2008); proximity to green space also improved longevity of senior citizens (Takano *et al.* 2002) and reduced stress in individuals (Korpela and Ylén 2007). Moreover, Mitchell and Popham (2007) found that populations of individuals

**Table 1. Commonly discussed urban ecosystem services/disservices associated with biogeochemical cycles, with their potential magnitudes (relative to the scope of the associated environmental problem) and uncertainty levels**

| *Ecosystem service* | *Potential magnitude* | *Current level of uncertainty* |
|---|---|---|
| C sequestration | Low | Low |
| Net GHG emissions | Moderate | High |
| Local cooling | High | Moderate |
| Stormwater mitigation | High | Moderate |
| Water-quality mitigation | High | High |
| Air-quality mitigation | Low | High |
| General human health | Moderate | Moderate |
| *Ecosystem disservice* | *Potential magnitude* | *Current level of uncertainty* |
| Water use | High | Moderate |
| Net GHG emissions | Moderate | High |
| Source of allergens | High | Low |
| VOC emissions | Moderate | Moderate |

**Notes:** GHG emissions are listed as both a service and disservice because the impacts of plants or soils may be either positive (net cooling) or negative (net warming) in hot climates. VOC = volatile organic compounds, which are precursors to the formation of ozone pollution.

 

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***Figure 5.*** *A demonstration rain garden on the campus of Florida Agricultural and Mechanical University.*

below retirement age with greater exposure to green space had lower rates of mortality in general and a lower rate of mortality specifically from circulatory diseases.

### *Ecosystem disservices*

Urban plants can be a source of allergens and of pollution precursors, namely volatile organic compounds (VOCs), which are emitted in large enough quantities to influence urban $O_3$ concentrations (Chameides *et al.* 1988). VOC emissions are species and site specific and include isoprene (2-methyl-1,3-butadiene). Isoprene emissions from plants depend on light availability and temperature, but are found in relatively few species (Guenther *et al.* 1993; Kesselmeier and Staudt 1999; Lerdau and Gray 2003). Monoterpenes, which consist of two isoprene units, are also species-specific VOCs emitted by plants and can be induced by environmental stresses such as drought, mechanical damage, and invasive pathogens (Lerdau and Gray 2003). Information is available for helping individuals select common horticultural species having low VOC emissions (Guenther *et al.* 1993; Benjamin and Winer 1998; Kesselmeier and Staudt 1999). Potential interactions between urban pollution and allergens also exist: Ziska *et al.* (2003) and Singer *et al.* (2005) reported that pollen production and allergens in invasive plants, such as common ragweed (*Ambrosia artemisiifolia*), increased with elevated atmospheric $CO_2$ concentrations and temperatures in urban landscapes.

### *Net effectiveness*

Deposition and uptake of some pollutants (eg $O_3$, $CO_2$, and particulates) by agricultural crops and natural vegetation have been relatively well quantified (Bytnerowicz *et al.* 1999; Grantz *et al.* 2003; Cieslik 2004), but urban studies are uncommon and model estimates in cities are still unvalidated. For example, pollutant uptake by urban trees in Sacramento, California, was modeled and reported to have a specific monetary value of US$383 per 100 trees, after VOC emissions were subtracted (McPherson *et al.* 1998); however, actual pollutant uptake, deposition, and re-suspension rates have not been measured for urban vegetation (Whitlow 2009). In general, the removal of atmospheric pollutants by vegetation is one of the most commonly cited urban ecosystem services, yet it is one of the least supported empirically. Nevertheless, the presence of urban vegetation does appear to have important influences on human health that are not directly related to air quality, though these influences are not yet fully understood. In many cases, the extent to which these cultural and psychological effects are related to specific ecological or biogeochemical processes is unknown. Collaborations between ecologists, social scientists, and epidemiologists are needed to further explore the interactions among supporting, regulating, and cultural ecosystem services and disservices as they apply to human health.

## ■ Urban ecological engineering: challenges of designing and managing landscapes as urban infrastructure

We have highlighted some of the many tradeoffs – costs and benefits, ecosystem services and disservices – and uncertainties about these tradeoffs in designing green space to achieve environmental goals. Studies of multiple urban biogeochemical processes including C, water, and nutrient cycles are needed to quantify these tradeoffs. To incorporate this information into urban design and optimize green infrastructure choices, we also require locally and regionally specific tools linked to desired outcomes. Otherwise, ecosystem services may be perceived as a tool kit that can be implemented uniformly, regardless of location or of specific outcomes. Planting urban forests and rain gardens, creating bioswales, or constructing buildings with green roofs may be seen as uniformly positive actions that can be undertaken in a standard way, regardless of local climate, infrastructure, technology, or governance. These assumptions should be tested with careful cost–benefit analyses that include multiple-criteria decision-support tools and account for regional differences to ensure that disservices do not outweigh services (eg Mysiak *et al.* 2005).

We also lack an adequate understanding of the institutional and infrastructural requirements for implementing

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**Panel 1. Urban trees and air quality**

Section 205 of HR2454, The American Clean Energy and Security Act of 2009, states:

*The Congress finds that:*
*(4) shade trees have significant clean air benefits associated with them;*
*(5) every 100 healthy trees removes about 300 pounds of air pollution (including particulate matter and ozone) and about 15 tons of carbon dioxide from the air each year;*
*(7) in over a dozen test cities across the United States, increasing urban tree cover has generated between two and five dollars in savings for every dollar invested in such tree plantings (www.govtrack.us/congress/bill.xpd?bill=h111-2454).*

One would assume from this text that (1) our knowledge of the impacts of trees on air quality is adequate to formulate "good" policy and (2) trees appreciably reduce concentrations of harmful air pollutants. However, despite simulation models demonstrating the benefits of urban trees, their effects on air pollution remain empirically unquantified.

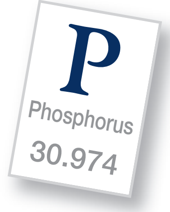
One of the presumed benefits of trees is particulate matter (PM) deposition onto canopies, which have a large surface area. However, particle deposition is affected by particle size, landscape roughness, canopy and leaf characteristics, and atmospheric turbulence. This complexity has hampered the development of a coherent theory for particulate deposition in canopies (Grantz *et al.* 2003; Hicks 2008). Reports of particulate deposition tend to be based on assumptions, models, or theories that are untested in urban settings. It is unlikely that even optimistic estimates of pollutant removals (uptake and deposition) will appreciably affect atmospheric concentrations in polluted cities. In contrast, tree canopies may reduce dispersion, causing locally elevated PM concentrations.

A commonly used model, Urban Forest Effects (UFORE) Model (www.ufore.org), estimates the effects of urban trees on particulate pollution. For New York City (NYC), UFORE predicted that – during the growing season – the forest removes 0.47% of PM matter, based on reported deposition velocities for particulates less than 10 microns in diameter ($PM_{10}$) (Nowak *et al.* 2002). If NYC were to add 1 million new trees to the urban forest, as is currently proposed (www.milliontreesnyc.org), particulate pollution removal would increase to 0.55% of $PM_{10}$ (there are currently ~6 million trees in the five-borough area). Thus, the additional 1 million trees would reduce PM by 0.02891 $\mu g\ m^{-3}$ to achieve a concentration of 36.97 $\mu g\ m^{-3}$. A decrease in $PM_{2.5}$ by 10 $\mu g\ m^{-3}$ has been estimated to add 0.61 years to human life expectancy (Pope *et al.* 2009). The net effect of planting 1 million trees would be to add 4.05 hours to the lives of NYC residents, based on PM reductions alone.

Although based on many assumptions, these calculations illustrate that assertions of the specific physical benefits of urban trees can be overstated. The pitfall in doing so is that the public receives the wrong message about how critical environmental and human-health problems must be solved. Tree-planting programs clearly have many benefits, but it is incumbent upon scientists to provide accurate and realistic estimates of both the ecosystem services and disservices of such programs.

ecosystem-services-based programs. Pincetl (2010a, b) showed that infrastructure for ecosystem services, even if well designed, poses numerous challenges for cities. For public works agencies that are responsible for meeting regulatory requirements, such as the US Clean Water Act, a decentralized ecosystem-services approach is difficult to implement because there is little supporting science to document its effectiveness. At present, neither science nor practice provides a means to predict how much bioswale capacity, for instance, would be needed to achieve regulatory compliance for criteria pollutants in a specific geographical area. In addition, the organizational structure of governmental agencies impedes implementation of large-scale green infrastructure. For example, separating water supply and wastewater treatment departments results in "black water" being considered as waste rather than as a resource that can be treated and reused. There are also barriers to social acceptance of green infrastructure. Methods of improving organizational structure and social acceptance should be developed, such as decentralizing municipal stormwater management and allocation of responsibility to homeowners (Shuster *et al.* 2008). In doing so, care must be taken not to imply that green approaches are cost- or maintenance-free, will solve all urban environmental problems, or will operate as expected without rational, site-specific design and placement. Rather, green infrastructure should be viewed as part of a suite of approaches, some of which can reduce costs associated with traditional built infrastructure, and others that have potential co-benefits, such as habitat restoration and cultural services.

Finally, budgetary constraints present additional limitations for implementing green infrastructure. In a time of tight budgets, municipalities have few resources to maintain existing infrastructure, let alone develop, implement, and test new designs for green infrastructures. Many cities have outsourced to non-profit organizations the implementation of ecosystem-services infrastructure programs such as tree planting, stream daylighting (bringing underground stream diversions to the surface), and construction of biofiltration projects. This can result in a complex set of public–private partnerships with little long-term stability on account of grant cycles, inadequate non-profit capacity, and other structural problems (Svendsen and Campbell 2008; Park *et al.* 2009). Implementing green infrastructure is still unproven, but small-scale, localized projects may permit their evaluation by scientists, recognizing the limitation of scale in their impacts.



Implementing larger-scale projects will help determine whether green infrastructure will have measurable effects on climate, air and water quality, and human health at a municipal scale.

## Conclusions

On the basis of current knowledge and uncertainties in biogeochemical cycles (C, nutrients, water, and pollutants), we summarize key urban biogeochemical regulating services as follows:

- Direct C sequestration by urban plants and soils is negligible as compared with urban GHG emissions; however, urban landscapes can have substantial local cooling effects that reduce energy use, but require site- and species-specific quantification.
- Bioswales, rain gardens, and other green infrastructure components reduce runoff, but further research is required to assess their effect on water quality and cost effectiveness, particularly at the watershed scale.
- The purported benefits of urban forests for improving air quality are poorly supported by empirical evidence in urban settings. In contrast, psychological benefits of green space and associated impacts on human health have been demonstrated, but the underlying linkages to biogeochemical and ecosystem processes need to be better understood.

Collectively, these issues reflect opportunities for integrating current biogeochemical science into the design and evaluation of green infrastructure related to GHG reduction, stormwater mitigation, and pollution remediation in cities. Implementing large-scale urban green infrastructure programs requires new knowledge about urban biogeochemical cycles, their role in ecosystem services and disservices, the associated uncertainties, and governmental structures that reflect improvements in knowledge of urban ecosystem function.

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