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Implications of climate change for managing urban green infrastructure: an Indiana, US case study

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Abstract

Urban areas around the world are increasingly investing in networks of urban forests, gardens, and other forms of green infrastructure for their benefits, including enhanced livability, sustainability, and climate change mitigation and adaptation. Proactive planning for climate change requires anticipating potential climate change impacts to green infrastructure and adjusting management strategies accordingly. We apply climate change projections for the Midwest US state of Indiana to assess the possible impacts of climate change on common forms of urban green infrastructure and identify management implications. Projected changes in Indiana's temperature and precipitation could pose numerous management challenges for urban green infrastructure, including water stress, pests, weeds, disease, invasive species, flooding, frost risk, and timing of maintenance. Meeting these challenges will involve managing for key characteristics of resilient systems (e.g., biodiversity, redundancy) as well as more specific strategies addressing particular climate changes (e.g., shifting species compositions, building soil water holding capacity). Climate change also presents opportunities to promote urban green infrastructure. Unlike human built infrastructure, green infrastructure is conducive to grassroots stewardship and governance, relieving climate change-related strains on municipal budgets. Resources for adapting urban green infrastructure to climate change are already being applied to the management of urban green infrastructure, and emerging research will enhance understanding of best management practices.

Keywords Indiana · Climate change impacts · Urban green infrastructure · Ecosystem services · Resilience · Urban forests

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1 Introduction

Cities both strongly drive and are highly vulnerable to planetary change (Grimm et al. 2008). Aging and energy- and resource-intensive urban infrastructures are increasingly fragile to extreme weather events and other global changes (Boyle et al. 2010; Wilbanks and Fernandez 2013). Population pressure exacerbates this fragility. World population will be two-thirds urban by 2050, and many regions are already highly urban (e.g., North America, 82%; Latin America and the Caribbean, 81%; Europe, 74%; Oceania, 68%; United Nations 2018). To sustain quality of life for growing urban populations while remaining within safe planetary biophysical boundaries (Röckstrom et al. 2009), today's cities are moving toward more resilient, self-sustaining systems.

Incorporating green infrastructure, broadly defined as networks of natural, restored, or cultivated green spaces managed for their associated ecosystem services, (Benedict and McMahon 2006; Allen 2012), is a growing strategy for designing cities that are more energy- and resource-efficient and also more livable, economically vital, and resilient to adversity (Secretariat of the Convention on Biological Diversity 2012). Urban areas around the world are increasingly investing in green infrastructure for carbon storage, flood control, thermal regulation, and other forms of climate change mitigation and adaptation (Demuzere et al. 2014; Matthews et al. 2015). Estimates of the economic value of such services range into millions of dollars for individual cities (Gómez-Baggethun and Barton 2013). Urban green infrastructure also provides important cultural services to residents, such as recreation opportunities, a sense of place, and community cohesion (Campbell et al. 2016).

As a living system, green infrastructure is inherently regenerative and less fossil fuel dependent than human built, gray infrastructure (e.g., buildings, machinery). Given sufficient biodiversity, green infrastructure can also be multifunctional and adaptive (Lennon and Scott 2014). Urban green infrastructure (UGI) nonetheless requires substantial management, with pruning, mowing, weeding, pest control, and irrigation among the activities needed to maintain optimal functioning. Municipalities need to anticipate and plan for changes in UGI management with ongoing climate change, just as is done for roads, bridges, and other forms of built infrastructure (Neumann et al. 2015). Although there is an abundant literature on the ecosystem services of urban green infrastructure (UGI) and its employment as a climate change adaptation strategy (Gómez-Baggethun and Barton 2013; Demuzere et al. 2014), managing UGI in a changing climate has received relatively little attention. Downscaled global climate models (GCMs) provide high-resolution climate projections useful for climate impact analysis at local scales and have been developed for the Midwest and Great Lakes region of the USA (Byun and Hamlet 2018). Using the Midwest state of Indiana as a case study, we apply fine-scale downscaled projections (Hamlet et al. 2019) to consider the impacts of climate change (70-year time horizon) on the biodiversity and associated ecosystem services (and disservices) of common forms of UGI and identify management implications. We emphasize urban forests, which include trees along streets and in backyards as well as stands of remnant forest (Nowak et al. 2001), because they are well appreciated as essential urban infrastructure warranting substantial municipal investment in urban forestry programs (Konijnendijk et al. 2006). Investments in other forms of urban green infrastructure are growing, however, as communities recognize the diverse benefits they provide. Therefore, we also consider urban food-producing spaces, green drainage systems, and prairies. Furthermore, as a traditional and popular form of urban green infrastructure valued primarily for their cultural services, we consider lawns. Although our focus is the state of Indiana, much of the information applies to other cities in temperate climates across the US Midwest and beyond, and our general approach is instructive for any urban area.

2 Indiana's climate, vegetative cover, and urban landscape

Indiana is part of the Midwest and Great Lakes region of the USA, with a four-season climate shaped by both cold Arctic and warm humid southern air masses and seasonal shifts in jet streams (Byun and Hamlet 2018). Historical vegetative cover was mostly hardwood deciduous forest, including beech, maple, ash, oak, hickory, and poplar (Potzger et al. 1956), but also included significant prairie, savanna, and swamplands (Jackson 1997). Forest cover dropped from greater than 90% to 4% between the time of European settlement (circa 1650) and 1900, recovering to approximately 23% by 2000 (Phillips et al. 2019). Likewise, most of Indiana's prairies, savannas, and swamplands were cleared, burned, or drained for agriculture, roads, and settlements as the state was colonized by pioneers in the eighteenth and nineteenth centuries (Jackson 1997), and croplands cover about 60% of land area today (Phillips et al. 2019).

By US census standards, about 6% of Indiana's land area (Nowak and Greenfield 2010) and approximately 70% of its population reside in urban areas (areas of at least 2500 people, U.S. Census Bureau 2012). Aside from Indianapolis and a handful of other larger cities (e.g., Fort Wayne, Evansville), Indiana has a remarkably even network of mid-sized communities (STATS Indiana 2016) and has retained a largely rural identity (Madison 2014). This pattern of extensive-not-intensive urbanization is distinctive among American states and suggests that while Indiana's urban areas will experience the full range of climate changes predicted statewide, the smaller sizes and relatively greater urban-rural linkages of many of its cities may lessen impacts of climate change, such as urban heat island effects and air quality issues. However, urban land in Indiana is projected to approach 17% of total area by 2050 (Nowak and Walton 2005), and urban climate change impacts can be expected to grow accordingly.

3 Climate projections across Indiana

Although Indiana has on average warmed about 0.6 °C (1 °F) over the last two decades (Frankson et al. 2017), this temporal change overlies a gradient of warmer to cooler temperatures that runs across the state in a roughly southwest to northeast direction (Hamlet et al. 2019). Fine-scale climate model projections indicate that this climatic gradient will be preserved as the state continues to warm (Hamlet et al. 2019). Below, we highlight changes in temperature and precipitation that Indiana urban areas can expect in the next 60–70 years, under the Representative Concentration Pathways (RCP) 4.5 and 8.5 scenarios ("medium" and "high"/"worst-case" emissions scenarios, Hamlet et al. 2019). We profile Evansville, Indianapolis, and Fort Wayne (cities in the southwest, central, and northeast of the state, respectively) for representativeness across the state's climatic gradient.

3.1 Temperature (T)

Relative to a 1971–2000 baseline, the average mean temperature in Indiana could increase by 3.3 to 5.6 °C (5.9–10.1 °F for RCP4.5 vs. RCP8.5), yielding more extreme hot days (Tmax > 35 °C (95 °F)), more days of frost-free growing season, fewer frost days (Tmin < 0 °C (32 °F)), and fewer extreme cold days (Tmin < −15 °C (5 °F)), with consequent shifts in USDA Plant Hardiness Zones (Hamlet et al. 2019). For example, compared to the historical baseline (1915–2013), Evansville could experience over one and one-half to nearly 3 months more of extreme hot days, Indianapolis over a month to two and one-third months more of extreme hot days, and Fort Wayne could experience nearly 1 month to 2 months more extreme hot days, particularly in summer (for RCP4.5 vs. RCP8.5, Hamlet et al., 2019, Table S1). Furthermore, projections show that the frost-free growing season could extend by over 3 weeks (Evansville, RCP4.5) to over a month in Indianapolis and Fort Wayne (either RCP, Fig. S9, Hamlet et al. 2019) and that the number of extreme cold days will be substantially reduced over most of Indiana (Fig. S6, Hamlet et al. 2019). USDA Plant Hardiness Zones are projected to increase one-half zone (RCP4.5) up to about one full zone (RCP8.5) compared to the historical baseline (1976–2005, Fig. S10, Hamlet et al. 2019).

Temperature-related climate changes could be exacerbated in urban areas due to the urban heat island effect, whereby building materials, heat sources, and other factors cause urban air temperatures to be approximately 1–2.8 °C (2–5 °F) higher than surrounding rural areas (EPA 2017). Urban areas, including Indianapolis, do in fact appear to be warming faster than surrounding rural areas, suggesting that climate change is interacting with urban heat island effects (Kenward et al. 2014). Urban heat islands are strongest at night (Kenward et al. 2014), so climate model projections for nighttime temperature increases in urban areas are likely to be conservative.

3.2 Precipitation

Climate models predict increases of 10–20% (RCP4.5) to 30–35% (RCP8.5) in winter and spring precipitation throughout Indiana compared to a 1971–2000 baseline, as well as increases in the number of days per year with heavy precipitation (> 25 mm) throughout the state, from over three-fourths of a day (RCP4.5) to about two more heavy precipitation days (RCP8.5) (Table S2 and Fig. S8, Hamlet et al. 2019). Snowfall is predicted to be reduced statewide and to become relatively infrequent in southern Indiana compared to the historical baseline (1915–2013) (Hamlet et al. 2019).

4 Climate change impacts to urban green infrastructure

4.1 Urban forest

Climate change is projected to affect the species composition of Indiana's urban forests as well as the services (e.g., climate and stormwater regulation, support of wildlife habitat) and disservices (e.g., insect pests and disease, production of allergens) that flow from such forests. Maple trees (*Acer spp.*) are commonly planted as street trees in Indiana, particularly *Acer saccharinum* (18%), *Acer saccharum* (7%), *Acer platanoides* (5%; although highly invasive, Indiana Invasive Species Council 2015), and *Acer rubrum* (4%; Davey Resource Group 2008). Although habitat suitability is expected to remain stable or increase for *A. saccharinum* and *A. rubrum*, habitat suitability for *A. saccharum* is projected to decline (Phillips et al. 2019). Furthermore, an overabundance of one family can leave urban forests vulnerable to insect pests and disease (Steenberg et al. 2017). For example, *Fraxinus pennsylvanica* and *Fraxinus americana*, which made up about 10% of Indiana's street trees as of last counting, are being lost to *Agrilus planipennis* (emerald ash borer) (Davey Resource Group 2016). Projected climate changes in the coming decades are expected to lead to additional losses, due to increases in insect pests from warming winters (Allen et al. 2010; Staudt et al. 2013) and pathogen infections from warmer temperatures and increased precipitation (Anderson et al. 2004). In addition, freeze-thaw events can weaken trees (Allen et al. 2010), increasing their vulnerability to secondary pests and disease (Ayres and Lombardo 2000).

Native tree species, such as *Quercus rubra*, *Quercus alba*, and *Carya ovata* that are projected to be marginally vulnerable in natural areas based on habitat suitability models (Phillips et al. 2019), are expected to be more vulnerable in urban areas than in natural landscapes in the state. Urban trees in North American temperate climates face additional stresses compared to trees in natural areas, such as stormwater runoff, road salt, air pollution (e.g., ground level ozone), and restricted growing conditions from being planted near roads and buildings (Mullaney et al. 2015). In particular, changes in precipitation patterns such as extreme drought and storms are projected to have significant and divergent impacts on different tree species (Fei et al. 2017). These additional stresses can exacerbate the impacts of climate change and reduce the capacity of species to adapt (Brandt et al. 2017). In addition, local climate phenomena like the urban heat island effect and alterations to urban soil are not included in broader-scale habitat suitability models, reducing the accuracy of model projections for urban trees. Despite the challenges they face, urban trees (particularly street and lawn trees) are also more likely to be watered and pruned, be treated for pests and pathogens, receive soil amendments, and experience less light competition compared with trees in rural or more natural settings, enhancing their capacity to adapt to some acute or chronic stressors associated with climate change.

Many of the trees planted along Indiana's streets and in parks and yards are cultivars or not native to the area, and species distribution model projections are not available for these trees (Phillips et al. 2019). Changes in habitat suitability for cultivars can be estimated by examining projected changes in USDA Plant Hardiness Zones and American Horticultural Society Heat Zones (Brandt et al. 2017). Some commonly planted species that cannot tolerate heat or require a significant chilling period to break dormancy are expected to be particularly vulnerable. *Syringa reticulata* and *Tilia cordata*, which are commonly planted street trees in the state, are recommended for planting up to heat zone 7 (61–90 days above 30 °C on average per year). Warmer temperatures in summer could shift much of the state to heat zone 8 or even 9 by the middle to later part of the century under a high emissions scenario, making conditions unfavorable to these species.

Urban trees can help regulate the climate of urban areas through shading, evaporative cooling, and windbreaks. It is estimated that urban trees in Indiana offset approximately $157 million in residential electrical and heating costs and help avoid the emission of 1.3 million metric tons of CO_2 (Nowak et al. 2017). However, the specific benefits will depend on a number of factors, including their placement relative to adjacent buildings, form, and distance as well as the assumptions made in modeling benefits (Ko 2018). In addition, trees placed on the southside of buildings can actually increase heating energy in northern areas (Hwang et al. 2016). If many of the dominant trees are damaged, lost, or stressed, these benefits (or costs) could be reduced. In addition, the projected increases in summer temperatures may increase the need for cooling, thus increasing the need for shade.

Urban trees can reduce pollution through the interception of particulate matter and uptake of gaseous pollutants, but play a complex role in urban air quality (Grote et al. 2016; Setälä et al. 2013). Research suggests that urban trees in Indiana removed approximately 8400 tons of pollution in 2010, with an estimated value of $63 million (Nowak et al. 2014). However, such effects may be relatively small, not easily generalizable, and offset by contributions to ground-level ozone and pollen production (Eisenman et al. 2019). Urban trees emit biogenic volatile organic compounds (BVOCs) from their leaves, which can serve as precursors to surface level ozone pollution (Simpson and McPherson 2011). Hotter summer temperatures can increase the production of BVOCs, reducing air quality (Peñuelas and Staudt 2010). In addition, warmer temperatures and increased carbon dioxide can increase airborne pollen, which can worsen seasonal allergies for people (Ziska and Beggs 2012).

Urban and community trees in Indiana store an estimated 13.76 million metric tons of carbon at a rate of 0.25 kg m^{-2} y^{-1} (Nowak et al. 2013), thus helping to mitigate greenhouse gas emissions into the atmosphere. However, the role of urban trees in the carbon cycle is more complex than their nonurban counterparts (Pataki et al. 2011). Some climatic changes, e.g., longer growing seasons, milder winters, and elevated atmospheric CO_2 concentrations, may enhance carbon storage rates by urban trees (Briber et al. 2015; Keenan et al. 2016). Other factors, such as fertilizer application and reduced competition for resources, may also contribute to higher rates of carbon uptake by urban trees (Zhao et al. 2016; Zhang et al. 2014). Warmer temperatures can also lead to increased carbon losses from urban ecosystems by increasing emission of CO_2 from soils (Decina et al. 2016) and causing heat stress in trees, reducing their ability to store carbon (Hardiman et al. 2017). More frequent and/or more intense storms and invasive insect pests could increase urban tree mortality and lead to carbon losses. Further, urban trees, particularly street trees, often require substantial maintenance (pruning, felling, chipping, hauling, etc.), activities which generate greenhouse gas and VOC emissions while also converting carbon stored as live tree biomass into forms that will quickly be released back to the atmosphere (e.g., wood chip mulch) (Horn et al. 2015). As a result, CO_2 cycles into and out of urban forests at rates faster than in nonurban forests, but the net effect of urban trees on atmospheric CO_2 concentrations may be neutral or even positive (Pataki et al. 2011). Accounting for urban forest carbon dynamics is essential for understanding the net carbon balance of urban ecosystems (Sargent et al. 2018).

Urban trees and other vegetation can also help regulate stormwater in urban areas (Berland et al. 2017, see also section 4.3, green urban drainage systems). Street trees in Indiana are estimated to provide approximately $24 million in stormwater management benefits (Davey Resource Group 2008). Heavy rain events have been increasing in Indiana and are projected to increase further, making the need for stormwater mitigation even greater. This is especially of concern in urban areas with high amounts of impervious cover and lack of riparian canopy cover. If some of the dominant urban trees are lost, this benefit could also be reduced.

Urban forests can support wildlife that in turn provides services. Native woody species, including oaks, poplar, maple, blueberry, and spruce, support high species richness of lepidopteran larvae (caterpillars), significantly more so than non-native species (Tallamy and Shropshire 2009). Caterpillars are a key food resource for terrestrial birds, including Neotropical migrants (Dunn and Garrett 1997). In addition to cultural services such as bird watching, birds help to regulate insect herbivores (Whelan et al. 2008; Mooney et al. 2010) and parasites, such as ticks (Samish and Rehacek 1999). Approximately 10% of migratory birds in the upper Midwest, among them warblers, flycatchers, and woodpeckers, are rare or declining (US Fish & Wildlife Service 2017). Urban forests and carefully designed greenways can provide valuable habitat to migratory (Rodewald and Matthews 2005) and forest-breeding (Mason et al. 2007) birds. The adult stages of Lepidoptera include butterflies and moths, which are an important group of pollinators for both crop and wild species (NRCS 2005; Rader et al. 2016).

Sustaining native urban forest diversity in the face of climate change will therefore help to sustain multiple ecosystem services. However, climate change may create mismatches in the timing of growth and development of plant hosts versus dependent trophic levels (Visser and Both 2005), reducing food availability for wildlife and associated ecosystem services.

4.2 Urban food-producing spaces

Industrial, highly centralized agriculture is both a large contributor to climate change and highly vulnerable to it (Schlenker & Roberts 2009; Weis 2010; Lengnick et al. 2015). In contrast, urban and regional agriculture can help to promote climate change resilience (Hellmann et al. 2010). Cities could reduce their carbon footprints and increase their food security through urban food production, particularly of fresh, whole foods such as fruits and vegetables (Lovell 2010). Urban food-producing spaces, including community gardens and orchards, rooftop agriculture, and urban and peri-urban farms, are highly multifunctional, providing food as well as supporting services (habitat for pollinators and other wildlife), regulating services (e.g., microclimate regulation, air purification, carbon storage, and stormwater regulation) and cultural services (e.g., foods connected with cultural heritage, recreation and relaxation; Lovell 2010). Indiana's capital city hosts over 150 food gardens or farms, and Purdue University Extension's urban farm incubator network and certificate program supports urban agriculture throughout the state (Purdue Extension Urban Agriculture 2017).

Although climate projections suggest longer growing seasons can be expected, they also suggest that Indiana will experience various climate change risk factors for agriculture in the USA, including higher temperatures, temperature fluctuations, excess precipitation, phenological mismatch between plants and pollinators, and increases in soil erosion, pests, and disease (Rosenzweig et al. 2002; Schlenker and Roberts 2009; Bartomeus et al. 2013; Walthall et al. 2013). These risk factors could be exacerbated in urban environments, due to the urban heat island effect, the high percentage of impermeable surfaces, habitat fragmentation, and pollution stress. Higher temperatures lead to greater irrigation needs, particularly in summer, and may promote northward range expansion of weeds (Walthall et al. 2013). Earlier springs lead to more variable spring temperatures and higher frost risk (Walthall et al. 2013) and potential asynchrony of plants and pollinators (Bartomeus et al. 2013; but see Harrison and Winfree 2015), of particular concern for fruit crops. Excess precipitation in spring increases risks of flooding crops, particularly in low areas (Rosenzweig et al. 2002). Milder winters and wetter springs can also enhance survival and/or population growth of animal pests and fungal or bacterial disease (Walthall et al. 2013), and associated flooding leads to soil disturbance and propagule dispersal, potentially promoting the spread of weeds and invasive plant species.

4.3 Green urban drainage systems

High impervious surface area makes stormwater control a common urban concern, given that roads and parking lots contribute substantial fractions of urban surface area (Frazer 2005). Whereas conventional stormwater control prioritizes flood prevention via engineered approaches such as drainage pipes, stream channelization, and storage basins, green drainage systems (e.g., riparian buffers, rain gardens, green roofs) use vegetation and soils to achieve simultaneous goals of flood prevention, water quality (pollution mitigation), wildlife habitat, and cultural services (e.g., recreational use) (Coffman 2000; Charlesworth 2010; Zhou 2014).

Rain gardens, bioswales, sand filters, constructed and natural wetlands, and natural riparian buffer areas have greater nitrogen removal efficiency than dry ponds and wet ponds (Collins et al. 2010; Bettez & Groffman 2012), even though the latter are more commonly employed (Collins et al. 2010). Green roofs are an underutilized means to retain and delay significant amounts of stormwater, while also increasing roof longevity, reducing energy demands for heating and cooling, mitigating urban heat island effects, reducing noise and air pollution (Getter and Rowe 2006), and promoting pollinator habitat (Tonietto et al. 2011).

Projected increases in overall precipitation and extreme precipitation events in winter and spring will impose additional floodwater stresses on green drainage systems. Urban riparian restoration is hampered by stormwater runoff that results in physical, biogeochemical, and biological degradation of urban stream systems ("urban stream syndrome"), reducing stream biodiversity (Violin et al. 2011) and ability to mitigate nitrate pollution (Gift et al. 2010). On the other hand, longer flood durations may result in lower emissions of the greenhouse gas nitrous oxide from urban riparian buffers (Jacinthe et al. 2012). Rain gardens (aka bioretention) help to reduce stormwater runoff and combined sewer overflows, particularly for smaller (< 22 mm) precipitation events, but could result in build-up of contaminants in sewers during dry periods that ultimately increase contaminant loads during subsequent precipitation events (Autixier et al. 2014).

4.4 Urban prairie

Urban prairie plantings range from curbside plantings and pocket prairies (aka "micro-prairies," Borsari et al. 2014) to larger tracts of several acres or more (e.g., Cotter 2017). Landscaping with native plants promotes native wildlife, including birds and pollinators (Burghardt et al. 2008; Tallamy and Shropshire 2009), supporting the increasingly crucial role of urban green spaces as centers of and corridors for biodiversity conservation (Derby Lewis et al. 2015). Local-genotype prairie grasses and forbs are readily available from Indiana native plant nurseries (Rothrock et al. 2016; INPAWS 2017). Greater canopy heterogeneity and deeper rooting depths suggest that prairie vegetation should capture and absorb more runoff than turfgrass, and some data support this (Bloorchian et al. 2016; Yuan et al. 2017; but see Steinke et al. 2009). Native grassland vegetation supports pollination and pest control services, consumption of methane, a greenhouse gas (Werling et al. 2014), and carbon sequestration (Tilman et al. 2006), although studies that quantify these ecosystem services for smaller-scale, urban plantings are needed. Like other forms of native landscaping, prairie plantings demand substantially fewer inputs of fossil fuels, water, fertilizers, and pesticides compared to conventional landscaping (Hostetler and Main 2010; Steinke et al. 2009). The hotter summers projected for Indiana (Hamlet et al. 2019) are likely to favor aggressive warm season grasses (Huang et al. 2001), potentially reducing diversity. In contrast, wet prairie plantings are vulnerable to greater spring flooding (Spence Restoration Nursery 2017).

4.5 Urban lawns

Turfgrass is a common form of urban greenspace – urban lawns are the most common irrigated crop in the USA – and turfgrass was estimated to cover 3843 km^2 in Indiana (Milesi et al. 2005), 4 % of the state's 92,789 km^2 of land area (US Census Bureau 2012). Conventional lawns are an esthetic norm (Larson et al. 2016) and support sports and other outdoor recreational activities but provide little wildlife habitat and support little biodiversity

(Hostetler and Main 2010; Aronson et al. 2017). Unshaded turf also has little cooling effect on surface temperature, even when irrigated (Pinctl et al. 2013; Chang and Li 2014). Although residential lawns have high carbon and nitrogen accumulation potential (Raciti et al. 2011), intensive management makes them net sources of greenhouse gas emissions (Townsend-Small and Czimczik 2010; but see Zirkle et al. 2011). In New York, reductions in snow cover caused urban lawns to retain less nitrogen and exhibit increased emissions of the greenhouse gases nitrous oxide and carbon dioxide (Duran et al. 2013); similar impacts could occur in Indiana with projected decreases in snow cover. Mown lawn may also be less effective at stormwater detention and retention than diverse perennial plantings (Yuan et al. 2017).

5 Managing urban green infrastructure in a changing climate

Meeting the challenges of climate change can involve managing for key characteristics of generally resilient systems as well as employing more specific strategies to adapt to particular environmental conditions (Fig. 1). Commonly cited characteristics of resilient systems include diversity (variation), redundancy (i.e., components sharing similar capabilities, as in functional groups of species, or different forms of green infrastructure yielding similar benefits), modularity (compartmentalization), and connectivity (linkages enabling flow of information or materials) (Levin et al. 2012). Diversity enables adaptation to changing conditions, whereas redundancy provides insurance against the loss of any one component. Modularity limits the spread of harms (e.g., disease, invasive species) while connectivity enables migration; the tension between these two characteristics is evident in the need to limit spread of invasive and pest species versus the need for north-south conservation corridors for beneficial organisms (Opdam and Wascher 2004). Invasive plants are common in urban areas (La Sorte et al. 2014), and their impacts are expected to increase with climate change (Staudt et al. 2013). These considerations emphasize the benefit of efforts to promote native species in landscaping and restoration, widely establish neighborhood greenspace programs, take invasive species out of trade, and engage in invasive species early detection and control efforts.

5.1 Specific management strategies

Climate change creates challenges associated with particular changes in environmental conditions. In Indiana, hotter, and potentially drier, summers may increase evaporative water losses; warmer, wetter winters and springs may increase pests, weeds and disease, and flooding damage; earlier springs may increase frost risk, especially for agricultural species; and earlier springs and longer growing seasons may result in earlier spring leaf out and later fall leaf drop. There are diverse management strategies to address these challenges, some of which are applicable across different forms of urban green infrastructure. For example, urban forests, food-producing spaces, and lawns may require increased irrigation and/or mulching, and/or shifts to more water-efficient species. The timing of tree, garden, and lawn care may also be affected (Derby Lewis et al. 2012). For example, pruning windows may shift because of changes in plant phenology (earlier spring leaf-out and later fall leaf drop) or disease-promoting conditions (wetter, warmer springs). Lawns may need to be mowed earlier or later in the year. Municipal forestry and public work departments may face increased needs for tree and branch removal following storms or outbreaks of pests and disease. Needs for pest or disease treatments may also increase. Cities that use street sweeping to remove leaf litter may

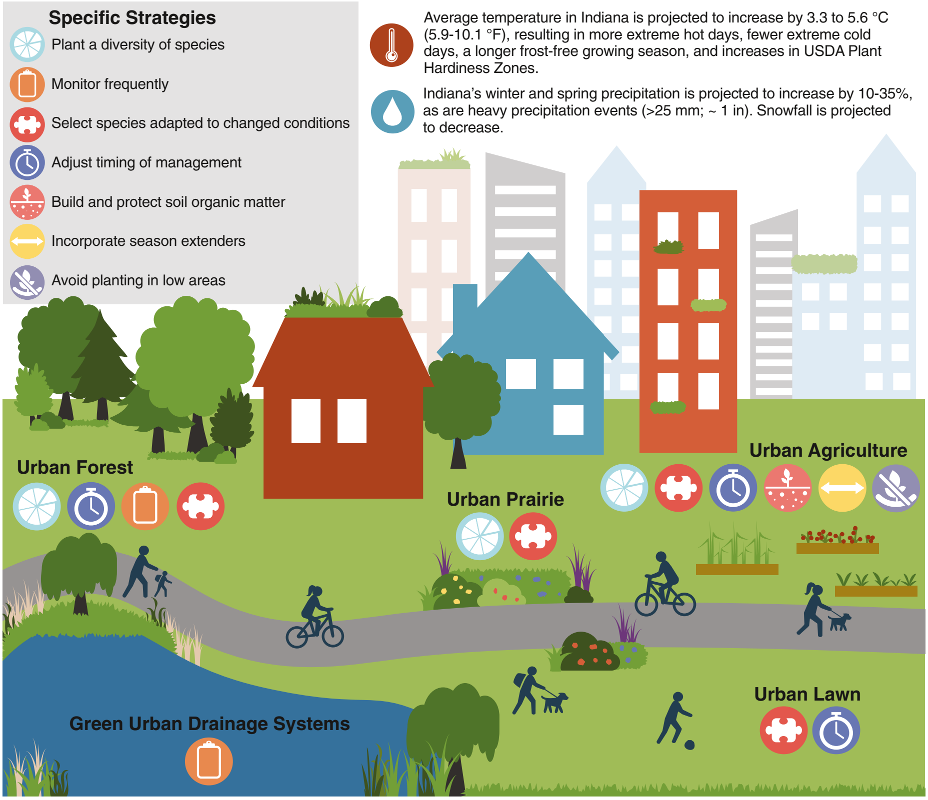

Fig. 1 Urban green infrastructure, including forest, food gardens, green drainage systems, prairies, and lawn areas, provides numerous ecosystem services. Based on "medium" and "high" emission scenarios (RCP4.5 and RCP8.5), projected changes in temperature and precipitation will pose management challenges for Indiana's urban green infrastructure, including storm damage, flooding, drought stress, pests, weeds, disease, invasive species, frost risk, altered habitat suitability, and changes in timing of growth. Theory suggests that promoting the diversity, redundancy, modularity, and connectedness of urban green spaces may make them more resilient to these changes. Various specific management strategies can also be employed, as illustrated. While key strategies for each form of green infrastructure are highlighted here, many strategies (e.g., building and/or protecting soil) are applicable across multiple forms of green infrastructure

need to adjust their schedules to align with the timing of leaf drop. Better and more timely monitoring of UGI condition may be necessary to optimize the timing of maintenance (i.e., for urban forests this means more consistent and timely tree inventories, for bioswales this might mean routine monitoring to keep them working properly).

A variety of strategies for urban forest management can be employed depending on whether the focus is on resisting change, building resilience, or facilitating transitions (Swanston et al. 2016). For example, a focus on retaining existing tree canopy could be achieved by treating susceptible trees with pesticides or fungicides to reduce the impacts of pests and disease. A focus on building ecosystem resilience calls for enhancing genetic, species, or structural diversity. In areas with significant canopy loss, the focus may be on transitioning to better adapted species or genotypes or new types of green spaces (see, e.g., McPherson et al. 2018).

Building soil quality through cover cropping and promoting crop and agroecosystem biodiversity may help to adapt urban agriculture to climate change (Kremen and Miles

2012; Lengnick et al. 2015). Cover crops protect against soil erosion and build organic matter, improving water holding capacity and resilience to dry and wet extremes (Hartwig and Ammon 2002). Mulching with compost materials has similar benefits (Cogger 2005; Saebo & Ferrini 2006). Diversifying crop varieties provides insurance against variable climate conditions, and overall agroecosystem biodiversity promotes stability of pollination and pest control services (Bartomeus et al. 2013; Lengnick et al. 2015). Avoiding planting in low areas (swales), which are more likely to experience frosts, is another key consideration, especially for orchard fruit production, which is highly sensitive to frost events that disrupt spring flowering (Lengnick et al. 2015). Season extenders, such as greenhouses, high tunnels, and hoop houses, can help to mitigate frost events and other climate risks, such as extreme precipitation events (Furman et al. 2013) and increases in pests and disease (Lamont 2009).

The expected increases in winter and spring flooding events with climate change reinforce calls for whole-catchment approaches to stormwater retention (Violin et al. 2011) and additional modeling to understand how multiple best management practices can help to mitigate extreme precipitation events (Autixier et al. 2014). Emphasis on synergizing green drainage systems with conventional stormwater control measures is needed (Zhou 2014).

Inclusion of a range of drought-tolerant species and varied burning and grazing regimes (Howe 1994) can promote urban prairie resilience to warmer summers and drought events. Furthermore, forb-dominated seed mixtures help to counter aggressive warm season prairie grasses (Rothrock et al. 2016) that may thrive with hotter summers. For wet prairie plantings, sedges and other species tolerant of long inundation periods are recommended (Spence Restoration Nursery 2017).

5.2 Economic considerations

Communities with more finances tend to have more staff and capacity to adapt green infrastructure management to climate change (Ordóñez and Duinker 2014). For example, the city of Indianapolis, the state's largest city, has only 1.4 park employees per every 10,000 residents, compared to an average of 4.8 across the nation's largest cities (Trust for Public Land 2017). Spending on parks is also lower than in other large cities, with $50 spent per resident per year compared to a national average of $83 (Trust for Public Land 2017). As more extreme weather events arise, spending on tree removal and disaster recovery may place further stresses on already-stretched budgets, reducing maintenance spending. Lack of maintenance could lead to further costs from weakened vegetation, creating a positive feedback loop (Vogt et al. 2015). However, grassroots efforts with a focus on shared governance may in fact be more important to the overall resilience of a community's urban forest than municipal budgets (Ordóñez and Duinker 2014).

5.3 Opportunities

Changing climate conditions may create opportunities for managing and promoting urban green infrastructure. Some plant species will grow faster in a warmer, more CO_2-rich environment. Fewer days below freezing and reduced snowfall could reduce the need for road salt, especially in southern parts of the state, reducing salt damage to sensitive species. While disservices and trade-offs (Pataki et al. 2011; Meerow and Newell 2017) need to be considered and addressed, the multifunctionality of green infrastructure often leads to co-benefits that could bolster support for its protection and expansion. For example, increasing tree cover to mitigate urban heat island effects (Davey Resource Group 2008) will also promote carbon storage, stormwater runoff reduction, and cultural services, including human health, coping capacity, and social bonding (Demuzere et al. 2014). Volunteer organizations in the state, such as Keep Indianapolis Beautiful, Inc., engage citizens in efforts to enhance urban forest and other green spaces and thus reduce vulnerability to the effects of climate change by increasing social adaptive capacity (Demuzere et al. 2014) and reducing direct negative impacts through the proper care and planting of future-adapted trees.

5.4 Resources

Resources are available to aid urban natural resource managers in adjusting urban green infrastructure management for climate change. The Chicago Wilderness Region Climate Action Plan for Nature, which includes 11 counties in northwest Indiana, guides conservationists in anticipating, adapting to, mitigating, and communicating about climate change in urban natural areas (Chicago Wilderness 2010; Derby Lewis et al. 2015). Forest Adaptation Resources: Climate Change Tools and Approaches for Land Managers covers climate change planning for natural and urban forests (Swanston et al. 2016). For example, this tool was used to develop an adaptation plan for urban forest management in Goshen, Indiana (Climate Change Response Framework 2018a). Websites maintained by the Indiana Invasive Species Council, Purdue Extension Entomology, and the Indiana DNR Division of Entomology and Plant Pathology offer resources for anticipating, identifying, and managing invasive and pest species of the state's urban and rural forest resources. Indiana University maintains the Environmental Resilience Institute Toolkit, an interactive online resource to help Indiana municipalities assess climate risks and review adaptation strategies, including those specific to urban green infrastructure.

5.5 Emerging research on Indiana's urban green infrastructure and climate change

Researchers are working to understand how urban forests are measured to assess risks and prepare for change (Climate Change Response Framework 2018b). Working with Indiana cities (pop. 6000+) that have current (2007-present) tree inventory data, urban forest vulnerability will be assessed using a suite of ecological and socioeconomic indicators. The extent to which urban forest plans consider environmental change, including resistance to change, building resilience, and adaptation to change, will be analyzed. Preparedness factors will be identified and used to construct a survey for urban forest stakeholders, to better understand their concerns about environmental changes, how to best prepare for these changes, and common barriers. Resilience-focused research on other forms of urban green infrastructure is also needed. As one example, researchers aim to develop and apply comprehensive social-ecological inventories of urban forests, food-producing spaces, and other urban green spaces, toward promoting city-wide resilience to climate change (Environmental Resilience Institute 2018).

6 Conclusions

Under both medium and high emission scenarios, downscaled climate projections for the midwestern US state of Indiana suggest warmer, wetter winters and springs and hotter, drier summers, posing numerous challenges for maintaining the biodiversity and ecosystem services of common forms of urban green infrastructure. Heat and drought stress, pests, weeds, disease, invasive species, flooding, frost risk, and changes in seasonal timing are among the prospective climate change impacts to Indiana's urban forests, food-producing spaces, green drainage systems, prairies, and lawns. There are general approaches and numerous specific strategies for managing climate change impacts to urban green infrastructure, some of which are already being applied locally. Our Indiana case study demonstrates that the information and tools are available for proactive assessment of impacts and development of practical management plans.

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