PROJECT SUMMARY

Overview:

Urban nature shapes and is shaped by the intersection of urban social and ecological systems. It provides habitat for biodiversity and forms networks with grey infrastructure that determine the movement of organisms, water, materials, and energy and provides a range of benefits for residents. However, key uncertainties exist about how social and ecological factors interact to shape urban nature: whether ecological theories developed in non-urban ecosystems predict patterns and processes in highly modified and managed urban systems; which theories best inform management to improve environmental outcomes in the face of stressors such as pollutants and climate change; and how patch-scale processes influence long-term processes at watershed and landscape scales. Furthermore, the value of and access to urban nature benefits depend on dynamic social, ecological, and technical contexts, and are not equal for all urban residents. The MSP Urban LTER (encompassing the seven-county Minneapolis-St. Paul, Minnesota metropolitan area) aims to illuminate the dynamic and diverse relationships between urban nature and people, towards better understanding how the urban ecosystem is changing in the face of rapid environmental and social change and to inform approaches for improving environmental outcomes for all residents.

A diverse team will focus on four interrelated questions: (1) How does biodiversity of urban nature interact with the broader biophysical, social, and technical contexts to mediate response of long-term ecological structure and function to urban stressors? (2) How do the ecological structure and function of urban nature interact with social and technical factors to influence urban climate, hydrology, and water quality of watersheds and lake ecosystems over annual to decadal timescales? (3) How are decisions about urban nature, community wealth, and well-being coupled over space and time to affect social inequities; how can governance institutions be changed to better address equity, such that environmental outcomes and human well-being are improved for all urban residents? And (4) How can long-term social-ecological research engage inclusively with diverse urban communities, particularly Black, Indigenous, and other People of Color (BIPOC), for more equitable and meaningful scientific and community outcomes? A wealth of existing long-term data, opportunities for comparative studies of watersheds draining to hundreds of lakes and ponds, across >100 municipalities and 33 watershed entities, create a rich mosaic for studying long-term effects of land cover and use, governance, and management on urban ecosystems. Highly polycentric governance and pronounced social and environmental disparities allow transdisciplinary research on complex long-term social and ecological relationships.

Intellectual Merit:

The proposed research illuminates the dynamic and diverse relationships between urban nature and people to improve understanding of social-ecological responses to environmental and social changes that are as rapid as any in recent history. By advancing understanding of how pollutants, biodiversity, land cover, habitat fragmentation, and drainage network properties affect urban nature processes in the face of such change, research will test whether ecological theories developed in non-urban ecosystems can predict patterns and processes in highly modified and managed urban systems. The project will shed light on patterns of social disparities in human relationships with urban nature and how such disparities can be addressed through institutional and policy change and greater inclusivity in long-term research.

Broader Impacts:

Schoolyard LTER activities will help diverse middle school learners meet Minnesota's new science standards through an urban ecology field trip at the Bell Museum, and workshops and toolkits to support teachers in teaching science standards using outdoor activities in their local school yards. REU students will be recruited through a consortium (80% minority, 40% Native) that provides support through activities and professional development. MSP will nurture academic-community partnerships, especially new and meaningful partnerships with BIPOC communities, through inclusive participatory research to understand mechanisms underlying socioeconomic disparities in urban nature burdens and benefits. MSP will enhance research infrastructure through development of models of pollinator dynamics, remotely sensed tree diversity, urban drainage networks, and urban lake dynamics, and through creation of a public data portal to synthesize, make accessible, and visualize societally relevant environmental and ecological data towards improving human health and urban nature benefits.

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Appendix Items:

*Proposers may select any numbering mechanism for the proposal. The entire proposal however, must be paginated. Complete both columns only if the proposal is numbered consecutively.

PROJECT DESCRIPTION

I. RESULTS FROM PRIOR NSF SUPPORT

Hobbie, PI, Nelson, co-PI. ***NSF Macrosystems Biology and Neon-Related Science, Collaborative Research: MSB-FRA: Alternative ecological futures for the American Residential Macrosystem.*** DEB-1638519. $530,420 to UMN from 1/1/17-12/31/20. This project investigated the factors that drive change in the American Residential Macrosystem and the ecological consequences of alternative futures for yard management across six U.S. metropolitan areas. The project's *Intellectual Merit* included advancing the understanding of the factors shaping diversity in residential yards, by showing that (1) spontaneous non-native plant species contribute to homogenizing urban yard floras across cities, (2) homeowners' landscaping priorities are only partially related to their yard vegetation at the continental scale, (3) even though more plant and bird species were present overall in yards managed for wildlife compared to either high-input or low-input yards, all yards were still homogeneous in terms of diversity across cities compared to natural areas, (4) across cities, ordinances clearly provided guidance about what plant species are or are not allowed and about specific requirements regarding the size or dimensions of impervious surfaces and plants, but not about factors such as neatness, and (5) homeowners have complex relationships with their yards, that fluctuate from being cooperative, oppositional, and negotiative. *Broader Impacts* included support for four postdocs at UMN and numerous undergraduate students. Products include 11 papers published, in press, in revision, or in review (indicated with[SEH.KCN] in References Cited). Data are being archived and made accessible through EDI.

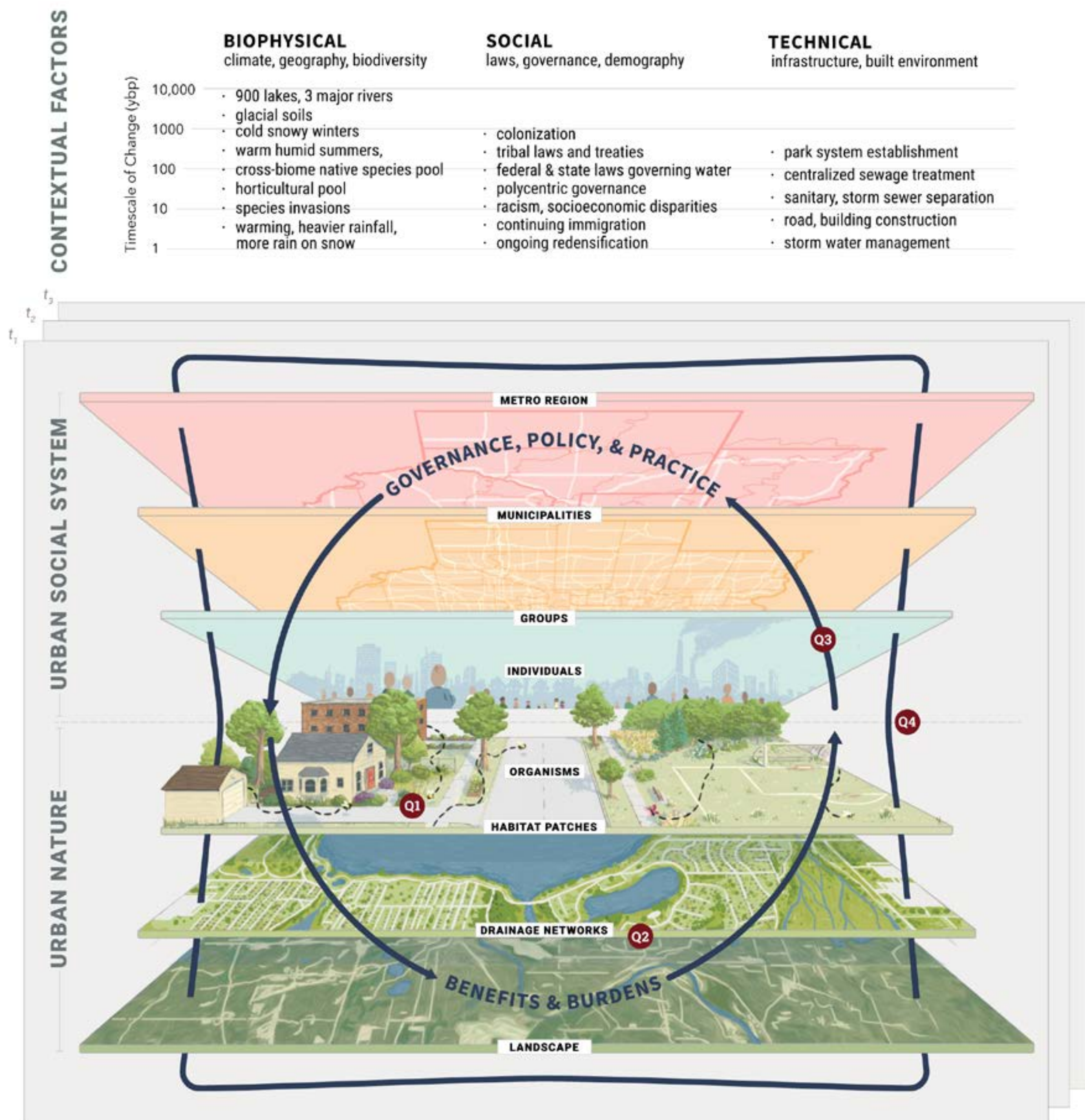
Finlay, co-PI. ***NSF WSC Category 2, Collaborative: Climate and human dynamics as amplifiers of natural change: a framework for vulnerability assessment and mitigation planning.*** DEB-0543363, PI: E. Foufoula. $451,459 direct to Finlay, of $2,526,005 at UMN from 9/1/12 to 8/31/18. This project investigated how geological history, climate change, and intensive agriculture affect water quantity and ecosystem integrity in watersheds. The project's *Intellection Merit* included developing a framework to assess the vulnerabilities of a natural-human system to guide decision-making in agricultural watersheds towards eco-hydrologic sustainability and resilience. Results highlighted the importance of specific places, times, and processes in determining how human- and climate-induced changes to intensively managed landscapes propagate through river networks and impact downstream waters. *Broader Impacts* included support for five postdocs, two graduate students and 14 undergraduates (for Co-PI Finlay), and generation of information used by agencies to develop restoration and management plans that address interactive effects of climate change and agricultural intensification within the Upper Midwest. Products include 10 published papers (indicated with[JCF] in References Cited) plus three in advanced preparation. Project data have been made available via UMN's digital conservancy.

Co-PIs Feng and Keeler have not received NSF funding within the past five years.

II. CONCEPTUAL FRAMEWORK AND RESEARCH GOAL

Urban nature in all its diverse forms – yards, parks, lakes, streams, engineered stormwater controls (Box 1) – is shaped by and shapes the intersection of urban social and ecological systems. Urban nature provides habitat for biodiversity and forms networks with grey infrastructure that determine the movement of organisms, water, materials, and energy through the urban ecosystem. Urban nature also provides a range of benefits for residents (Keeler et al. 2019). However, key uncertainties exist about how social and ecological factors interact to shape urban nature (Pickett et al. 2020). For example, can ecological theories developed in non-urban ecosystems predict patterns and processes in highly modified and managed urban systems? Which theories best inform management to improve environmental outcomes for diverse urban residents in the face of dynamic stressors such as pollutants and climate change? How do patch-scale processes that vary across urban green-blue-grey networks influence long-term ecological, hydrological, and climatic processes at the watershed or landscape scale?

Box 1. We define ***urban nature*** broadly to encompass all "green" and "blue" spaces in cities, including vegetated, pervious surfaces, as well as lakes, ponds, and rivers. By our definition, urban nature crosses public-private boundaries, including private yards; public parks; vacant lots; commercial green space; green infrastructure, defined as "all natural, semi-natural and artificial networks of multifunctional ecological systems within, around and between urban areas" (sensu Tzoulas et al. 2007); bioswales, detention and retention ponds, and other vegetated "Best Management Practices" (BMPs) and "Low-Impact Development" (LID) strategies for managing stormwater; and "nature-based solutions" aimed at mitigating and managing impacts of global change (Nesshöver et al. 2017).



Box 2. The conceptual framework for the MSP Urban LTER, with research questions (Q1-4) mapped onto it. We conceive of the MSP Urban Ecosystem as a coupled ecological and social system operating within the milieu of diverse influential and interacting biophysical, social, and technical contextual factors (top). Contextual factors are dynamic – some have changed recently (springs have become wetter, specific acts of racism have led to social uprising) and others changed hundreds (major biomes intersecting) to thousands (glaciers retreating) of years ago. Within this context, the MSP Urban LTER aims to determine the long-term coupled dynamics of urban nature (bottom) and the urban social system (middle) in the face of rapid environmental and social change. We examine this coupling across organizational scales of urban nature from diverse **organisms** in **habitat patches** within stream and stormwater **drainage networks** in **landscapes** with abundant surface water; and of the urban social system from diverse **individuals** acting in **groups** within numerous (117) **municipalities** within complex governance systems and institutions at the **metropolitan region**, with seven counties and 33 watershed management entities, and a metro-wide governing body, the Metropolitan Council. Our research addresses how biodiversity at the organism to habitat patch scales, and habitat fragmentation and connectivity mediate long-term responses of ecological structure and function to urban stressors such as toxins, pests, pathogens, and climate change (**Q1**); how configuration and connectivity of urban nature habitat patches and impervious cover at the drainage network and landscape scales influence long-term hydrology, urban climate, and water quality (**Q2**); how ecological, hydrological, and climate processes of urban nature create benefits and burdens for diverse human communities over time, and in turn how governance, policy, and practice can change to improve equity of urban nature decisions (**Q3**); and how the long-term process of growing inclusive relationships for knowledge creation and practice change scientific and community outcomes in the urban ecosystem (**Q4**).

Furthermore, although urban nature is often touted for providing benefits to urban residents, the value of and access to those benefits depend on dynamic social, ecological, and technical contexts (Keeler et al. 2019), and are not equal for all urban residents. Urban nature is more accessible to those who are wealthy, white, and able-bodied (Wolch et al. 2014, Pradhananga et al. 2019), and urban residents may even be harmed by disservices and green gentrification (Wolch et al. 2014, Taguchi et al. 2020). Understanding the historical and contemporary social factors that shape urban nature and cause unequal access to urban nature benefits could help reverse and prevent inequities, but such information is currently lacking. Diverse stakeholders experimenting with new approaches, advocates who change policies, and researchers who increasingly engage with community partners may be able to address this challenge.

Focusing on the seven-county Minneapolis-St. Paul urban ecosystem (**MSP**) and guided by our Conceptual Framework (Box 2), our goal is **to illuminate the dynamic and diverse relationships between urban nature and people, towards better understanding how the urban ecosystem is changing in the face of rapid environmental and social change, and to inform approaches for addressing inequities and improving environmental outcomes for all residents.**

III. MSP URBAN DOMAIN

Biophysical, Social, and Technical Context. MSP lies at the transition between two major biomes (tallgrass prairie and deciduous forest) and has abundant freshwater, with three major rivers, ca. 900 lakes, and many thousands of ponds (Fig. 1). MSP has cold, snowy winters and warm, humid summers, mean annual temperature of 8°C, and mean annual precipitation of 76 cm (period 1981-2010), 15-20% of which falls as snow (NCEI, Asheville, NC). MSP is located on traditional and contemporary Dakota lands, is the center of the Dakota origin story, and is home to numerous Indigenous sacred sites. MSP also has strong Ojibwe ties and connections to reservations and treaty territories in northern Minnesota. The current population of 3 million is 73% white and 27% Black, Indigenous, and other People of Color (BIPOC), including one of the largest and most diverse urban American Indian populations and a large population (12%) of recent immigrants, mostly from Southeast Asia and North Africa (Minnesota Compass n.d.). MSP has highly polycentric governance, including seven counties, 117 cities (population >1000), and 33 watershed-based governing entities, and forms the center of the larger Minneapolis-St. Paul-Bloomington Metropolitan Statistical Area. Regional governance, coordinated through the Metropolitan (Met) Council, has planning and policy-making authority and provides essential services. MSP has made historical investments in its parks and extensive highway, road, and public transportation systems, centralized sewage treatment, and separate sanitary and storm sewers.

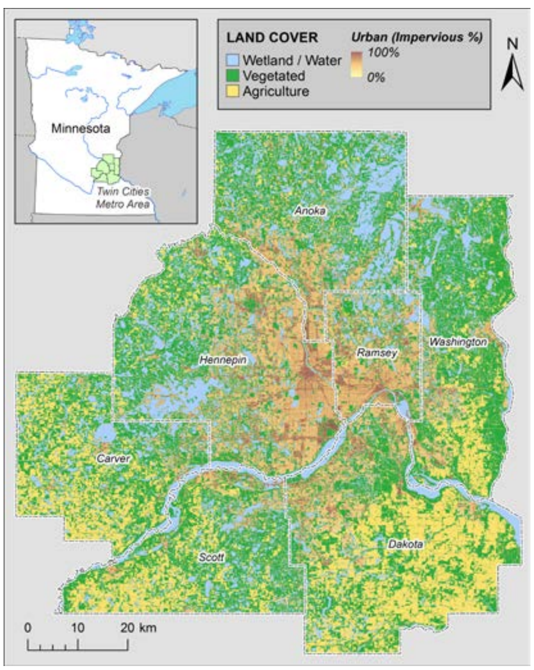


Figure 1. Map of MSP LTER area. Seven-county Twin Cities (Minneapolis-St. Paul) metropolitan area, including major water, land type, and impervious cover.

Appropriateness of MSP to represent urban ecosystems of the USA for long-term research. The seasonally cold and increasingly wet MSP climate represents a meaningful contrast to the hot, arid climate of the Central Arizona Phoenix (CAP) LTER. The flora of MSP is typical of temperate, mesic U.S. cities, and shares many species and lineages with northern and eastern cities (Pearse et al. 2018). Minnesota has some of the greatest and most persistent racial and socioeconomic disparities in the U.S. in test scores, graduation rates, college readiness (Grunewald and Nath 2019), income, homeownership, and health outcomes (City of Minneapolis 2019). These disparities recently led to an eruption of local, national, and international social uprising and calls for justice for Black people, following the murder of George Floyd. Racial and socioeconomic disparities extend to environmental conditions, with, for

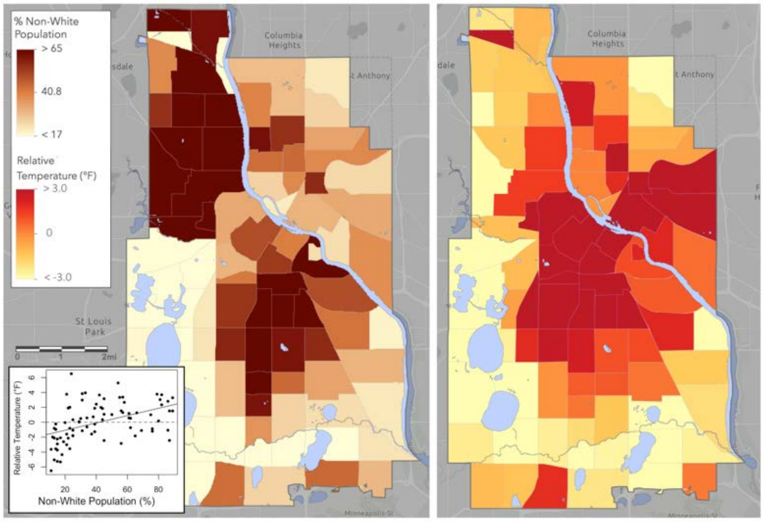


Figure 2. Comparison of racial composition and heat exposure of Minneapolis neighborhoods. (Left) Percent non-white population is calculated as the total population minus the non-Hispanic white population, divided by the total population using 2013-2018 American Community Survey data. (Right) Relative temperature exposure, based on Landsat 8 satellite-derived summertime land surface temperature estimates. (Insert) Graph of relative heat exposure (°F) vs. percent non-white population by neighborhood. Dashed line represents the average temperature, and solid line represents the linear relationship ($y=0.05x-2.05$, Adjusted $R^2=0.19$, p-value<0.005). Heat is also related to income: relative heat=-0.00005*income + 372.7, Adjusted $R^2=0.48$, $p<0.005$.

example, greater urban heat island (UHI) effects in communities of color and low wealth (Hoffman et al. 2020) (Fig. 2), underscoring the need to evaluate how such disparities came to be and how they can be addressed, and to strengthen community-based participatory research partnerships between environmental scientists and BIPOC communities.

A region in transition. MSP is undergoing rapid social and ecological change that will likely intensify in response to recent social events and climate change. MSP has rapidly diversified (from 91% white as recently as 1990 to 73% currently), and its growing population is stressing aging infrastructure, as is typical of cities established in the late 1800s. After decades of decline, the population is rising in the urban core, increasing housing demand. Lack of affordable housing and recognition of the consequences of discriminatory housing policies, including redlining and racial covenants, have recently led to novel zoning policies to incentivize multi-family housing in Minneapolis (City of Minneapolis 2019), while the periphery continues to expand into agricultural landscapes.

Given its ecotonal location, MSP will likely undergo marked ecological change in the face of climate change, biological invasions, and pollutants. Climate is changing more quickly in the Upper Midwest than in any other part of the contiguous U.S., with rapid warming exacerbated by an increasing UHI (Stone Jr 2007, Smoliak et al. 2015), declining snow cover, more intense rain and wind (Walsh et al. 2014) (Fig. 3), and more rain-on-snow events contributing to flooding (Berghuijs et al. 2016). Nitrogen (N), phosphorus (P), and road salt degrade surface waters, with 183 lakes "impaired" by excessive nutrients and 22 by high chloride (Fig. 4) (Minnesota Pollution Control Agency 2020), and frequent beach closures because of stormwater runoff leading to high pathogen loads or harmful cyanobacteria blooms. Invasive pests and pathogens create high costs for prevention, treatment, and tree removal (Haight et al. 2011). For example, emerald ash borer (*Agrilus planipennis*)

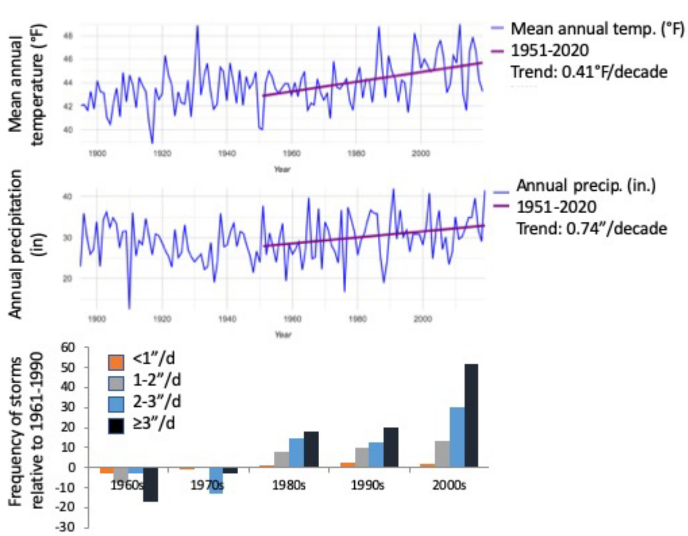


Figure 3. Recent trends in MSP temperature and precipitation. Data are for Hennepin County from NCEI.noaa.gov and analyzed at county level by MN DNR State Climatology Office (top, middle) and for Upper Midwest from U.S. Historical Climatology Network (bottom, figure courtesy of Peter Snyder).

and the fungal pathogen oak wilt (*Bretziella fagacearum*) are spreading through the urban forest (Fahrner et al. 2017) killing ash and red oaks, common trees throughout MSP.

Unique opportunities for comparative study. The MSP LTER will take advantage of watersheds draining to hundreds of lakes and ponds, across >100 municipalities and 33 watershed governance entities, which create a rich mosaic for studying the long-term effects of land cover and use, governance, and management on lakes with varying physical and ecological characteristics. Highly polycentric governance and pronounced social and environmental disparities allow transdisciplinary research on complex long-term social and ecological relationships and outcomes for urban residents and ecosystems.

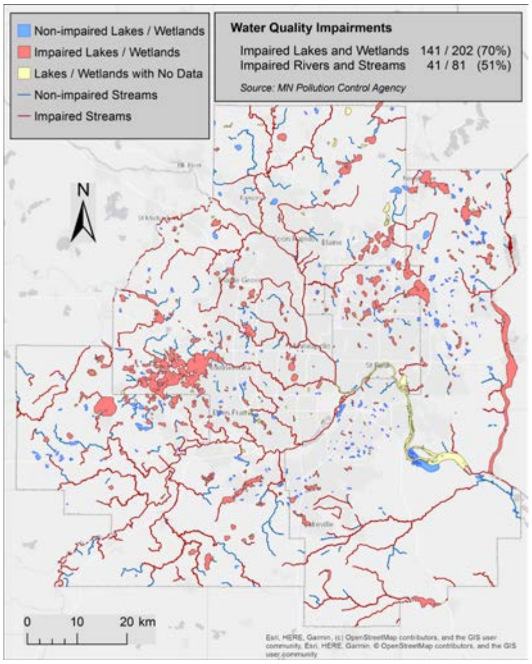


Figure 4. Impairment status of lakes and streams in MSP.

Abundance of available long-term data. MSP is extraordinarily rich in long-term data collected by diverse entities, including by LTER collaborators (see letters of collaboration). For example, ca. 30 stormwater sites and ca. 22 streams in MSP have event-based and seasonal stormwater flow and chemistry data, and 11 sites have year-round data, readily accessed for analyses and publication (e.g., Janke et al. 2017). Many entities have been collecting data on lake chemistry, chlorophyll, and clarity, in many cases >20 years, and on streams and rivers. In fact, MSP has the longest records of water quality for stormwater, streams, and a representative diversity of urban lakes in the world (Soranno et al. 2017). The U.S. Forest Service collects urban Forest Inventory Analysis (FIA) and national landowner survey data. Minnesota has a separate state agency, Minnesota Geospatial Information Office, for collection of geospatial data. Three University of Minnesota (UMN) centers (Minnesota Population Center, Polar Geospatial Center, USpatial) collect and curate long-term demographic and spatial data. Collections of the Bell Museum and the National Lacustrine Core Facility extend records into the past.

Co-location of MSP and UMN. The land-grant UMN is centered in MSP, and all senior personnel have conducted research in MSP, many in past collaborations. The large, diverse team, with expertise ranging from ecology to hydrology, engineering, sociology, and public policy, is held together by collaborations, shared service to UMN, and mentorship of students and postdocs. Nearly all investigators reside in MSP, increasing the ease of convening scientific meetings and participating in **Broader Impacts** activities. Researchers have established relationships with diverse entities that can be leveraged for co-production of knowledge, and translation of research into policy, education, and community-engaged scholarship.

IV. BACKGROUND AND RESEARCH QUESTIONS

To achieve our overall research goal, we will focus on four interrelated questions (Box 2):

Question 1. How does biodiversity of urban nature interact with the broader biophysical, social, and technical contexts to mediate response of long-term ecological structure and function to urban stressors?

Question 2. How do the ecological structure and function of urban nature interact with social and technical factors to influence urban climate, hydrology, and water quality of watersheds and lake ecosystems over annual to decadal timescales?

Question 3. How are decisions about urban nature, community wealth, and well-being coupled over space and time to affect social inequities; how can governance institutions be changed to better address equity, such that environmental outcomes and human well-being are improved for all urban residents?

Question 4. How can long-term social-ecological research engage inclusively with diverse urban communities, particularly Black, Indigenous, and other People of Color, for more equitable and meaningful scientific and community outcomes?

Following we provide the background and conceptual framing for each of these major research questions.

Q1. Biodiversity of Urban Nature. Urban nature hosts a diverse assemblage of species, including spontaneously occurring and intentionally cultivated native and non-native species (Padullés Cubino et al. 2019), whose populations occur in habitats of varying suitability within highly fragmented landscapes. These diverse organisms likely vary in their susceptibility to urban stressors, including elevated concentrations of pollutants, abundant pests and pathogens, and extreme temperature and hydrological impacts that are exacerbated by UHI phenomena and high impervious cover (Hobbie and Grimm 2019). Different historical and contemporary factors, including management regimes, lead to diverse plant and animal community composition and different levels of habitat connectivity within and among particular types of urban nature (e.g., lawns, parks, forests). **Question 1** aims to understand the role of this biodiversity in urban nature habitat patches, and the configuration and connectivity among patches, in influencing long-term ecological responses to unique urban stressors, including UHI-exacerbated warming and disturbances such as storms, pests, and pathogens (Grimm et al. 2017). We will investigate biological mechanisms (e.g., related to life history) explaining why organisms in urban environments vary in their responses to complex suites of toxins, and whether diverse assemblages of species, arising from intentional management actions or other factors, confer enhanced resilience and resistance in the face of perturbations, and thus increase urban nature benefits of biodiversity. Work under Q1 further addresses the effectiveness of specific management actions targeting wildlife, and whether ecological theory can improve management in fragmented urban landscapes. We focus on (1) insect pollinators because they represent a tractable system for studying ecological and evolutionary responses to toxins; are experiencing worldwide declines; are important for urban and peri-urban agriculture; have high intrinsic value for urban residents, including Indigenous people; and are the focus of local nascent and novel management efforts; and (2) urban tree canopies because of the importance of leaf area, canopy cover, and litterfall in influencing urban climate, hydrology, and water quality and providing benefits for urban residents such as shade, evaporative cooling, higher property values, enhanced well-being, and cultural identity. Thus, Q1 addresses LTER Core Areas 1-primary production, 2-population dynamics and trophic structure, 4-inorganic inputs and movement of nutrients, and 5-disturbance.

Pollutant Effects on Ecological and Evolutionary Processes. Urban ecosystems have a unique chemical signature of elevated heavy metals, salts, N, and P (Snell-Rood et al. 2015, Kaushal et al. 2020). Toxins of anthropogenic environments shape the ecological communities of microbes, plants, and animals in human-dominated environments (Yang et al. 2006, Moron et al. 2012, Hassall 2014) and their subsequent development and evolution (Shochat et al. 2006). However, it is unclear why some species perish and others thrive in the face of suites of elevated toxins and nutrients. Comparative studies across species suggest that combinations of phenotypic plasticity and certain life history traits may predispose some species to thrive in anthropogenic environments. For instance, both behavioral plasticity and increased reproductive events favor survival in cities (Moller 2009, Maklakov et al. 2011, Sayol et al. 2020) and both physiological plasticity and lifespan can cause variable responses across species to climate change (Dalgleish et al. 2010, Seebacher et al. 2015). Understanding these interactions has implications for restoration of urban ecological communities in ways that reduce toxin loads and improve the well-being of both human and ecological urban communities.

Biodiversity and Ecosystem Functioning. Theory, experiments, and observations indicate that more diverse ecological communities are more productive and resistant to climate extremes (Isbell et al. 2015a). Urban ecological communities include a mix of species shaped by a complex suite of human and non-human factors, including climate, the regional species pool, human preferences and wealth, management, regulations, and, for plants, availability from nurseries (Avolio et al. 2018, Pearse et al. 2018, Roman et al. 2018, Cavender-Bares et al. 2020). For example, factors such as race, income, and homeownership have been linked to urban tree cover and diversity (Fan et al. 2019, Locke et al. 2020), but how differences in tree diversity contribute to the long-term resistance and resilience of the urban tree canopy in the face of disturbance, and thus to more sustained benefits for urban residents, is unknown.

Habitat Quality, Fragmentation, and Pollinator Dynamics. Urbanization destroys, fragments, and alters wildlife habitat, and has contributed significantly to steep declines of insects (Sánchez-Bayo and

Wyckhuys 2019), including bees (Winfree et al. 2009, Vanbergen et al. 2013). However, cities also provide critical insect habitat, primarily through management of urban green spaces (Aronson et al. 2017). Maintaining and adding beneficial habitat patches to the urban landscape can increase connectivity between isolated populations (e.g., Rudd et al. 2002). For example, insects such as bees are central place foragers, and according to theory, maximize foraging efficiency when visiting discrete food patches to procure nectar and pollen (Olsson et al. 2015). The abundance and diversity of bees foraging in a patch thus depends on the distance from that patch to nesting habitat (Lonsdorf et al. 2009), and suitable nesting substrates (well-drained soils for ground-nesting bees and pithy stems or soft wood for cavity-nesters; Cane 1991, Michener 2000) can act as a limiting resource for urban bee populations (Potts and Willmer 1997). Thus identifying how best to manage the patch size, composition, configuration, and connectivity of habitat patches within the urban matrix, and how communities and populations change over time will significantly advance urban biodiversity research (Lepczyk et al. 2017).

Q2. Watersheds and Lakes. Research under **Question 2** builds on **Q1** by increasing the scale and scope of inquiry. Q2 will address how characteristics of urban nature, including those of the habitat patches studied under Q1, interact with management and climate to influence hydrologic processes, UHI effects, and the transport and fate of materials, at scales ranging from individual habitat patches to watersheds to landscapes. Q2 further addresses consequences of transported materials for the structure and functioning of lake ecosystems and the distribution of UHI and lake benefits for urban residents across MSP. Thus, Q2 addresses LTER Core Areas 1-primary production, 3-organic matter accumulation, 4-inorganic inputs and movement of nutrients, and 5-disturbance.

Effects of Land Cover on Hydrology and UHI. Urban vegetation provides many benefits to urban residents including mitigation of the UHI effect (Smoliak et al. 2015), reduced runoff (Armson et al. 2013), and filtering of pollutants (Morani et al. 2011), among others. However, these benefits are not distributed equally among urban residents (Hoffman et al. 2020). On the other hand, urban vegetation is a major source of nutrient and organic matter (OM) pollutant to urban watersheds (Hobbie et al. 2017). Within urban nature patches, fluxes of water and materials largely are governed by local physiological (e.g., evapotranspiration, ET) and physical (interception, infiltration, litterfall, erosion) processes that partition the flow of water between ET (which can provide evaporative cooling), runoff, and groundwater (Charlier et al. 2009). Thus, hydrologic fluxes depend on interactions between vegetation type, topography, soils, and climate. For example, different vegetation types partition water fluxes depending on characteristics such as leaf area, rooting depth, and stem architecture (Johnson and Lehmann 2006). Little work has explored the tradeoffs between the environmental benefits and burdens of various types of urban vegetation (e.g., Small et al. 2019a) and how these tradeoffs vary over time.

Along with hydrologic processes, land cover affects surface temperature (Baker et al. 2002, Hart and Sailor 2008, Bounoua et al. 2015, Ziter et al. 2019) and local precipitation (Haberlie et al. 2015, Niyogi et al. 2017). However, each city's unique land cover mosaic hinders development of generic relationships to predict UHI (Brazel et al. 2000, Sailor and Fan 2002, Jenerette et al. 2006). Therefore, combining fine-scale empirical datasets with detailed numerical models is essential to predict spatially explicit feedbacks among UHI and urban nature and determine the configuration of urban vegetation (i.e., cooling by ET) that will mitigate MSP's UHI and increase long-term resilience to climate extremes.

Urban Drainage Networks and Transport of Pollutants. At the watershed scale, the effects of urban vegetation on water fluxes and on mitigating stormwater runoff depend on its spatial configuration in relation to existing grey infrastructure and the built environment (Meierdiercks et al. 2017), which dominate flow paths. Fluxes of water interact with sources of urban pollutants to determine their transport through urban stream and stormwater drainage networks. For example, inputs of biologically reactive forms of nutrients and other pollutants to urban drainage networks are dominated by atmospheric deposition, fertilizer, road salt, and pet waste (Hobbie et al. 2017). These sources may enter streets and drainage networks directly (like atmospheric deposition and road salt), or move into streets via overland flow, erosion, and tree nutrient uptake and subsequent leaf litterfall on the street. Thus, understanding material and water fluxes requires consideration of how diverse types of urban nature are positioned along with impervious surfaces to form hydrologic flow paths.

The transport of water, nutrients, and OM into streams, ponds, lakes, and rivers also depends on stormwater management strategies. Management approaches targeting source reduction have focused on regulations and practices reducing use of lawn fertilizer P and road salt. However, P that accumulated in soils and sediments prior to fertilizer P restrictions (i.e., legacy P) may be mobilized and contribute to water quality impairment (Bennett et al. 1999, Motew et al. 2019) and road salt application has led to an accumulation of chloride (Cl) in urban soils, groundwater, and surface water (Dugan et al. 2017, Kaushal et al. 2018b). Street sweeping reduces the fluxes of litterfall nutrients and OM entering streets and storm drains, but current practices are insufficient to prevent leaf litter transported to storm drains from being a major source of nutrient pollution and impairing functioning of stormwater management structures. Further down the flow pathway, stormwater management structures can retain nutrients, trapping sediment-bound P, infiltrating dissolved N, and removing N through microbial denitrification; however, the effectiveness of these management strategies depends on urban development and climate contexts. For example, variation in the relative cover of tree canopy interacts with roads to affect stormwater nutrient concentrations and overall nutrient loads (Janke et al. 2017). The dynamic network connectivity of impervious roads in urban watersheds determines partitioning of runoff and solutes into surface water or groundwater (Baruch et al. 2018, Blaszczak et al. 2019). And how social equity has been considered in making management decisions and allocating management resources is unclear.

In addition to management effects, climate change affects transport processes directly – through altered seasonality, intensity, and frequency of precipitation events, such as rain-on-snow or heavy rainfall events – and indirectly through its effects on vegetation composition and productivity and thus on transpiration. Furthermore, climate change (combined with salt accumulation) increases the stratification in stormwater ponds, causing anoxic conditions that release dissolved P from sediments (Taguchi et al. 2020). The interactive effects of climate change, management, and configuration of habitat patches in stormwater drainage networks in mesic regions remain underexplored.

Consequence for Lake Ecosystems. In many urban areas, lakes sustain critical ecological communities and functions and provide important benefits for urban residents by supplying fresh water and food, regulating water and regional climate, and providing aesthetic and recreational opportunities (Phaneuf et al. 2008). Yet, the quality of urban waters is widely impaired by eutrophication (excessive algal growth caused by high nutrient inputs) (Dubrovsky et al. 2010), chemical contaminants, and other stressors (Baker and Newman 2013). In north-temperate North America, where lakes are an important feature of many landscapes, more intense rainfall events will further contribute to impaired water quality by increasing runoff, reducing infiltration, and accelerating soil erosion and transport of nutrients to lakes (Jeppesen et al. 2005, Duan et al. 2012) thereby reducing the effectiveness of current runoff management practices (infiltration structures such as rain gardens, trenches, and stormwater ponds). Effects of warmer winters are especially poorly known, but in northern climates, they will increase freeze-thaw events, releasing nutrients from soil and litter during periods when capacity of plant and microbial nutrient uptake is low, accelerating erosion and P loss from lawns (Bierman et al. 2010). Because of the tight coupling of urban land and water via impervious surfaces and storm drains (Kaushal et al. 2008), pulsed inputs of heat, salt, and nutrients from more intense runoff have immediate impacts on lake nutrient cycles.

Urban water quality responses to watershed processes, management, and climate change also affect and depend on in-lake processes, such as stratification and mixing, oxygen regime, and sedimentary processes. For example, warming increases N removal through enhanced denitrification (Veraart et al. 2011) and may increase P release from sediments if oxygen declines (Jacobson et al. 2010). These processes, combined with climate-driven increases in P-rich stormwater runoff (Jeppesen et al. 2005, Duan et al. 2012), will enhance algal productivity, and shift algal nutrient limitation from P- to N-limitation. N limitation favors noxious cyanobacteria, a major contributor to eutrophication and harmful algal blooms (Patoine et al. 2006, Wagner and Adrian 2009, Elliott 2012, Huber et al. 2012, Kosten et al. 2012). Effectiveness of in-lake management such as alum treatments to bind P and removal of P-rich aquatic macrophytes will depend on lake characteristics such as depth and mean residence time. Rapid changes in temperature, nutrients, and toxins in urban runoff provide a unique opportunity to study coupling of climate and water quality within the context of diverse lakes, landscapes, and management.

Q3. Urban Nature, Social Inequities, and Governance.

Decades of research on the distribution of urban environmental benefits and burdens arising from urban nature show that environmental hazards are more likely located in poor and marginalized communities, whose members experience higher rates of air and water pollution, and live in neighborhoods with fewer and lower-quality recreational amenities (Bullard 2000), and less access to biodiversity (Leong et al. 2018). Research under **Question 3** explores the historical and current factors that contribute to such disparities in MSP and how advocacy for policy change and practice working across differences can address those inequities. Thus, Q3 addresses LTER Core Area 6-social, economic, or cultural processes, linking to Core Areas 1-5.

Distribution of Benefits and Burdens of Urban Nature. Socioeconomic disparities reveal how environmental investments have historically been funded, designed, and implemented to benefit wealthier and whiter communities to the detriment of BIPOC communities (Pulido 2000, Grove et al. 2018). Such disparities have been codified through property values, where environmental gains in white communities allow for intergenerational transfer of wealth for white homeowners and compound the racial wealth gap in highly segregated cities (Crompton 2005). Concurrently, in historically marginalized neighborhoods, poorly maintained stormwater infrastructure, siting of environmental hazards, and degraded air and water quality negatively affect people's health and well-being, decreasing property values, and leading to cycles of vacancy and further value decline. Such reinforcing feedbacks contribute to the racial disparities in wealth and well-being that define U.S. cities today (Sampson 2017).

Past research documents patterns of inequities along racial and class lines in the spatial distribution of biodiversity and urban ecosystem services (Lerman and Warren 2011, Wilson 2020). Racial segregation can be explained by a suite of national policies and laws that govern banking and lending and interact with local laws and policies to shape housing markets. For example, the legacy of discriminatory mortgage lending or "redlining" is still apparent in the distribution of tree canopy in many cities (Hoffman et al. 2020, Locke et al. 2020), and racist housing policies and diminished household wealth interact to concentrate poverty in degraded environments (Pulido 2000). Far less is known about other potential feedback loops, including accumulation of wealth and benefits associated with investments in urban nature in majority-white neighborhoods and the burdens of such investments in the form of gentrification and displacement in historically marginalized neighborhoods (Checker 2011, Wolch et al. 2014). A recent meta-analysis of U.S. and European cities found that investments in urban nature can increase property values by up to 20% (Bockarjova et al. 2020). More research is needed to understand the contextual factors that determine where and when improvements in different types of urban nature (lakes, parks, lawns, tree canopy) have race-dependent impacts on the accumulation of wealth and well-being in cities.

Improving Equity and Human Well-Being through Changing Urban Nature Governance. In human-dominated ecosystems, such as urban areas, governance is a key driver of ecological processes (North 1990, Ostrom 2005), where environmental governance refers to "the set of regulatory processes, mechanisms, and organizations through which political actors influence environmental actions and outcomes (Lemos and Agrawal 2006, p. 298)." Governance thus encompasses formal institutional processes, such as the creation and enforcement of laws and the structure of public and private organizations, as well as informal processes such as social norms and kin- and friendship-based social networks. Practice comprises the multiple behaviors of working together to create change: from considering "day-lighting" a stream buried in a storm drain to talking about values with someone who looks "different". Practice in governance includes political ideas that confirm rather than challenge current governance (Brodkin and Kaufman 2000), or, conversely, may bring new expectations for civil society. However, when practice brings novel ideas into society over time, and thereby transforms governance, both environmental outcomes and effects on human well-being are unclear. A detailed scientific understanding of the mechanisms that link governance with ecosystem processes, and how governance itself transforms through interactions among people and nature, remains limited, largely because there are few long-term datasets measuring relationships between ecosystems and governance.

Although abundant research exists about how specific factors such as social networks, articulated expectations, policy advocacy, social movements, and social learning contribute to transforming social-ecological processes (Koontz et al. 2015, Crutchfield 2018, Weible and Sabatier 2018), past studies have been limited by small sample sizes, failure to account for multiple variables (Matias et al. 2017), and short time frames. Political advocacy plays a key role in social transformations, but the role of advocacy and why it is successful are rarely incorporated into models of social and ecological change (Han and Barnett-Loro 2018). Also, most studies of social transformations do not incorporate valid measures of ecological processes (Scott 2015, Gerlak and Heikkila 2019). The LTER framework enables long-term data collection on social and ecological transformations to overcome these challenges, and MSP enables rich comparative study because of the unusually large number of distinct governance units (e.g., cities, watershed districts), non-government organizations, neighborhood communities, businesses, and social entrepreneurs, all contributing to decisions about environmental practice and governance.

Q4. Inclusive Social-Ecological Research. The rich and growing literature on community-based participatory research (CBPR), including community-engaged scholarship and participatory action (Viswanathan et al. 2004, Ortiz et al. 2020) offers principles, theoretical frameworks (Tremblay et al. 2017), measures, and best practices (Lemos et al. 2018, Massuel et al. 2018, Wyborn et al. 2019) for improved relationships between researchers and community partners and more equitable science outcomes. However, CBPR literature and our past work lack systematic and longitudinal examinations of community and academic outcomes. Moreover, most evaluative work has been unidirectional and summative, focusing on assessments of a research project's impact on the community after work is complete.

Under **Question 4**, we aim to meaningfully engage and support diverse community partners and interests, including BIPOC, in (1) setting clear academic and community goals; (2) recognizing and discussing the relevance of race, ethnicity, class, and culture in social-ecological research; (3) identifying multiple criteria for assessing long-term success; and (4) conducting formative evaluation of community and academic outcomes using multiple methods and multidirectional assessments of long-term change. In addition to evaluating community outcomes, we will evaluate the effects of CBPR on the researcher, the research project, and the research institution. We recognize that successful community-academic partnerships require a concerted effort, take time to develop, and necessarily transform over time. We also acknowledge that communities are a ubiquitous part of urban ecosystems, underscoring that long-term relationships are critical to this work. Research under Q4 addresses LTER Core Areas 1-6 as it lies at the intersection of biophysical science, social science, and community engagement (Fig. 5).

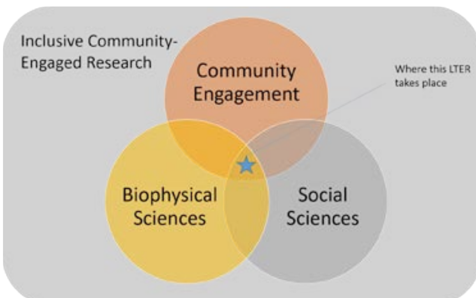


Figure 5. Position of MSP LTER research at the intersection of biophysical, social science, and community engagement research.

V. RESEARCH ACTIVITIES

Below, under each primary question, we define specific research questions, articulate our hypotheses, and describe the models and long-term experiments and observations we will use to test them.

Q1. How does biodiversity of urban nature interact with the broader biophysical, social, and technical contexts to mediate response of long-term ecological structure and function to urban stressors?

Q1.1. How are pollutants distributed spatially in MSP and how do these pollutants affect population dynamics, community structure, and ecosystem and evolutionary processes in urban ecosystems?

Hypothesis 1.1.1. Urban toxins will vary spatially with patterns of anthropogenic inputs, such that heavy metals (lead, cadmium, zinc, copper, nickel) and salts will be elevated within the urban core compared to suburban areas because of higher traffic volumes and a history of leaded paint and arsenic contamination. In contrast, nutrient pollutants characteristic of lawns (e.g., N) will be similar in urban and suburban areas.

Hypothesis 1.1.2. Species vary in their ability to cope with complex suites of urban toxins, such that (a) species with evolutionary histories with naturally occurring toxins have cross-tolerance to a range of novel urban toxins, and (b) species with life history traits associated with high fecundity and rapid dispersal have relatively high tolerance of suites of urban toxins.

Hypothesis 1.1.3. Management (mulching or removal of leaf litter) and composition of urban plant communities shape the long-term distribution of toxins throughout the soil profile through their effects on earthworm activity: availability of higher Ca leaves in forbs of pollinator plantings (**Q1.3**) and intact leaf litter promote activity of Lumbricid earthworms and movement of heavy metals into deeper soil levels.

Background and Past Results. MSP's social, biophysical, and technical contexts interact to create unique patterns of pollutant distribution, increasing nutrients and toxins to often-stressful levels (Fig. 6). For example, road salt use has created widespread and chronically high Na and Cl regimes (Kaushal et al. 2018a). Engineered materials elevate metals, some essential but stressful at high doses (zinc, copper), some toxic at any level (lead, cadmium), and many that interact chemically with other urban stressors (low oxygen, high temperature). Although evolutionary biologists have identified many specific adaptations to individual stressors, such as salts or metals (Ji and Silver 1995, Ahmad et al. 2012), it is less clear how organisms cope with simultaneous changes in multiple toxins that exert diverse physiological effects.

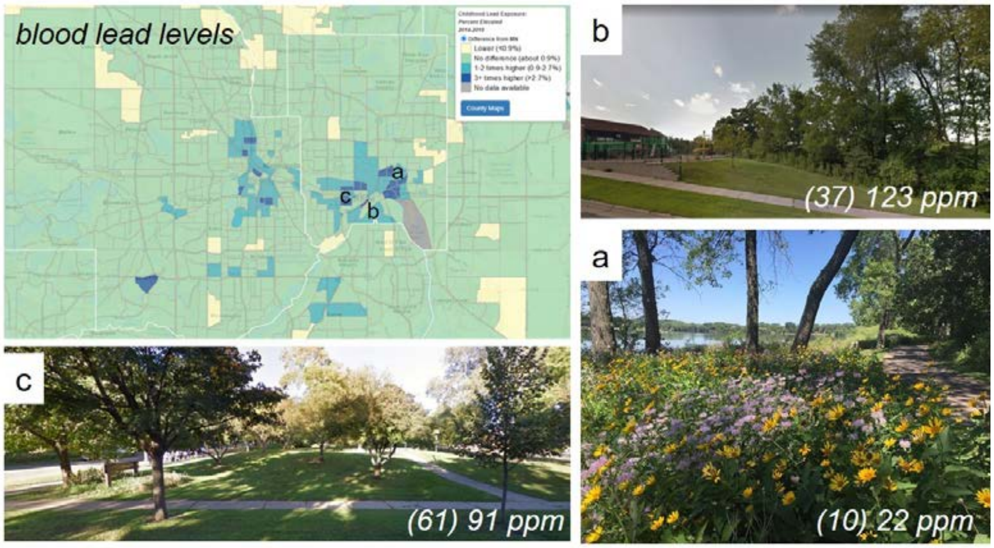


Figure 6. Example of site variation with respect to lead. (Upper left) Blood lead levels across the cities of Minneapolis and St. Paul (MN Department of Health). Child blood lead is generally correlated with soil lead levels (Mielke and Reagan 1998). Photographs show current variation in urban park management for three parks in St Paul in areas of high lead levels. Numbers indicate soil lead levels ((mean) max) measured in playgrounds in these parks in 1989 (Mielke and Adams 1989). We will make use of variation in park management (leaf litter management, pollinator plantings) (e.g., a and b vs. c) to test the hypothesis that leaf litter availability affects earthworm activity and movement of heavy metals into deeper soil layers. In addition, LTER will digitize and integrate thousands of additional soil metals data and make them publicly accessible via the data portal.

We focus on two sets of traits that may result in adaptations to suites of toxins. First, life history traits associated with rapid generation time and population growth are often associated with adaptation to polluted sites (Posthuma and Vanstraalen 1993, Agra et al. 2011). Although life history has been linked to stress tolerance in plant ecology (Ponge 2013, Pierce et al. 2017), it is unclear how this theory maps onto tolerance of urban toxins. Second, generalized physiological stress mechanisms may underlie the ability to cope with suites of urban toxins because upregulation in response to one condition results in cross-tolerance to other stressors (Snell-Rood et al. 2018). Evolutionary histories in toxic environments should result in general adaptation mechanisms to deal with a range of toxins, potentially pre-adapting organisms to toxic urban environments (e.g., Li et al. 2007, Efferth and Volm 2017), however, this hypothesis has yet to be tested. We will study insect pollinators, making use of comparative life history data across species (Swanson et al. 2016) and work exploring generalized stress responses as pre-adaptations to novel toxins (Sikkink et al. 2020). Our focus on interactions between insect pollinators and toxins allows us to link with habitat-scale ecological questions (**Q1.3**) and with **Q3** and **Q4**, given the importance of insect pollinators to Indigenous and other human communities in MSP.

Ecological communities of polluted sites can also change toxin levels, with ecological and human health implications (e.g., Chen et al. 2019). For example, plants, microbes, and fungi take up heavy metals, reducing or altering their bioavailability (Khan et al. 2000). Plants that hyperaccumulate metals are often used in bioremediation (Shah and Nongkynrih 2007, Mosa et al. 2016), but disposal of metal-enriched plants is problematic. An alternative approach is to move toxins into deeper and less accessible soil layers. Earthworm activity, especially of anecic worms such as *Lumbricus terrestris* that feed at the soil surface but burrow deeply, can redistribute surface pollutants to lower soil layers, potentially reducing surface toxicity (Sterckeman et al. 2000, Fernandez et al. 2007, Rillig et al. 2017) (but see Sizmur and

Hodson 2009). Lumbricid earthworms prefer calcium (Ca)-rich leaves (Hobbie et al. 2006, Holdsworth et al. 2008), suggesting that management via deliberate planting of Ca-rich species could significantly impact earthworm activity and thus movement of heavy metals into deeper soil layers. For instance, the leaf Ca content of many trees (*Tilia*, *Acer* spp.) and prairie forbs is higher than that of grasses (Watanabe et al. 2007). Thus, leaf litter management (e.g., mulching or removal), and/or species selection for plantings could impact surface soil heavy metal content over time due to earthworm activity.

Thus, in our research, we will explore two key knowledge gaps related to urban toxins: (1) the mechanisms, including the roles of evolutionary history and life history traits, underlying variation in tolerance of insect pollinators to elevated levels of multiple toxins simultaneously, and (2) whether toxin levels in surface soils can be reduced by exploiting preferences of some earthworm species for Ca-rich plant material.

Research Methods. We have three aims under **Q1.1**: (1) to create a spatially explicit and publicly accessible data portal on the distribution of toxins across MSP; (2) to quantify ecological community responses to variation in urban toxins over space and time; and (3) to address how plant communities affect distribution of heavy metals through soils. First, we will create a public toxins data portal at the **landscape scale**, supplementing existing soil heavy metal and salt data with new measurements from underrepresented locations. Existing soil metals data include over 150,000 data points from thousands of soil testing sites in MSP, but are not accessible online. The data portal will additionally integrate aquatic toxins data from stormwater and lakes across MSP (**Q2.1-2.2**). We focus on elemental toxins because levels of N, metals, and Cl are straightforward to quantify and substantial (but inaccessible) data already exist. The LTER information manager will be primarily responsible for creating and updating (as data become available) a GIS-based online data portal and will work in collaboration with topical experts (e.g., Senior Personnel Jelinski, Finlay) to correct for differences across studies in methods when data are merged. This data portal will be valuable for addressing basic ecological and evolutionary questions and societally relevant questions about areas at risk for lead poisoning, Cl stress, or excessive algal growth.

Second, we will quantify ecological community responses to variation in toxic heavy metals at the **habitat patch** scale (Grandjean 1983, Robards and Worsfold 1991), although the data portal will provide opportunities for future studies of other pollutants. Using data on occurrence of heavy metals, we will select sites (N=40) with a range of toxin levels (Fig. 6). Sites will include open habitats (<50 % canopy cover) within lawns and parks, and we will coordinate with **Q1.3**, **2.1**, and **3.2** to maximize site overlap and sampling. We will sample bumblebee and butterfly communities annually in May-June and again in July-August using both transect-based counts of all sighted individuals of bumblebees and butterflies and collection of focal species using sweep nets (N = 4 bumblebee species, N = 10 butterfly species; we will coordinate bumblebee sampling with **Q1.3** on shared sites). We will use four 1-m^2 quadrats/site to measure the plant community. On collected individuals, we will use body size to assess condition, as body size tends to reflect larval nutritional stress in insects (Nijhout 2003), and measure body heavy metal content (ICP-MS on bee hindlegs). Relationships between body condition and heavy metal content will be integrated with existing species-specific trait data (Swanson et al. 2016) to relate heavy metal sensitivity to species' life history and

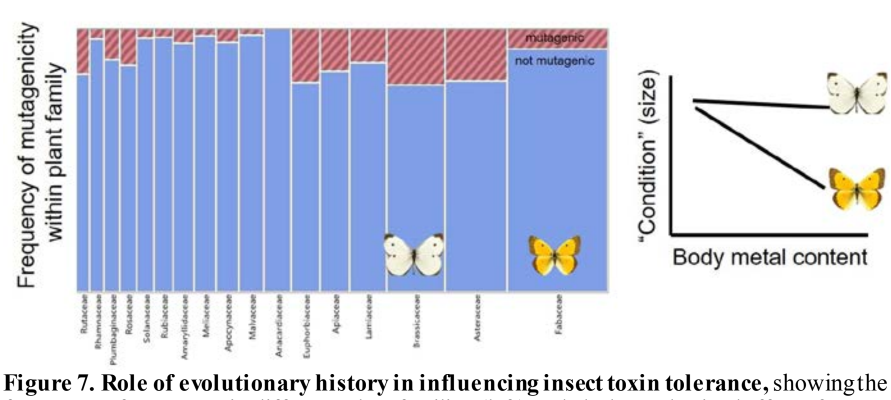


Figure 7. Role of evolutionary history in influencing insect toxin tolerance, showing the frequency of mutagens in different plant families (left) and the hypothesized effect of evolutionary history with mutagens on sensitivity of body condition to metals (right). Organisms vary in their evolutionary history with different toxins, e.g., butterflies that feed on larval host plants in the family Brassicaceae have more of a coevolutionary history with mutagenic plant defenses than butterflies that feed on Fabaceae or grasses (left). We will quantify toxin-tolerance with species-level reaction norms relating metal exposure and condition (right). Snell-Rood unpublished.

evolutionary history.

Finally, we will address how Lumbricid earthworms mediate effects of management and species composition of plant communities on distribution of heavy metals through soils over semi-decadal to decadal timescales. For each site, we will sample at least two different microenvironments in years 1 and 5 to test if plants (grass vs. forb) or litter management intensity (e.g., mulched or litter removed versus no litter removal, Fig. 6) affect the depth distribution of metals and worms in soil over time, using soil cores to measure depth distributions of metals (Taylor and Griffin 1981) and sampling the earthworm community with liquid mustard extraction methods (Hale et al. 2005).

Expected Results. We expect that heavy metals and salts will be distributed heterogeneously across MSP, with higher concentrations where road density is high. Nutrients may be distributed more homogeneously, but with fine-grain heterogeneity resulting from skewed fertilizer practices (Fissore et al. 2011). We expect that species with faster life histories (high fecundity, rapid dispersal) and more generalized physiological stress responses (based on evolutionary history or genomic data) will be more tolerant of urban toxins. For example, Snell-Rood has shown that butterfly species that feed on Asteraceae and Brassicaceae have a pronounced evolutionary history with dietary mutagens and should have the machinery to cope with suites of toxins resulting in less decline in body size with increased body and site toxin levels (Fig. 7). Finally, we expect site management to affect depth distribution of heavy metals over time. In particular, use of forb-rich pollinator plantings and leaving leaf litter in place should both promote earthworms relative to traditional lawns, resulting in less toxic surface soils.

Q1.2. How do urban tree canopies of differing diversity – resulting from contrasting management and legacies of past housing and investment policies – influence resilience and resistance in the face of climate change, invasive pests and pathogens, land-use change, pollutants, and other stressors?

Hypothesis 1.2.1. Intentionally designed tree canopies that are selected and managed to align with future climate conditions will have greater resistance and/or resilience to disturbances than those that are designed to align with current or past climate conditions.

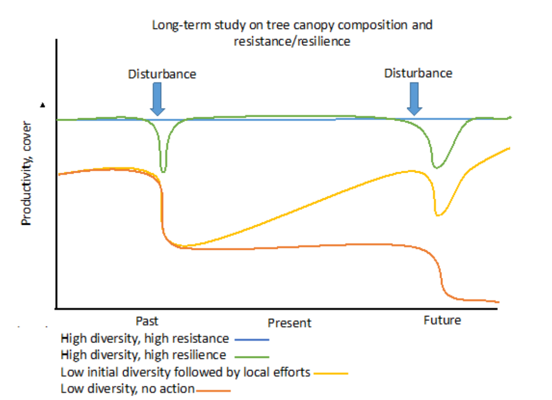


Figure 8. Hypothesized relationships between the resistance or resilience of tree productivity and diversity in response to hypothetical disturbance. We hypothesize that more diverse tree canopies will have higher resistance or resilience than less diverse canopies. Less diverse tree canopies will be more common in lower income areas, and will have less resistance and resilience, but local actions taken to increase diversity and cover will enhance resilience and/or resistance to disturbance.

Hypothesis 1.2.2. Tree canopies with higher taxonomic, age class, and genetic diversity will have higher resistance and resilience (of productivity and canopy cover) to pests, pathogens, extreme weather events and heat, and high soil salinity (where resistance is the capacity to remain essentially unchanged and resilience is the return to a previous state following disturbance, Fig. 8) (Isbell et al. 2015b).

Hypothesis 1.2.3. Socio-economic factors linked to legacies of racially motivated housing policies are major predictors of current tree canopy structure and diversity, and, consequently, their resistance and resilience to perturbation.

Hypothesis 1.2.4. The dominant drivers of variation in tree community composition and diversity – and subsequent resistance and resilience to disturbance – will vary with the grain size and spatial extent of analysis. For example, tree cover and composition will co-vary with past discriminatory housing policies at neighborhood scales, but with soil type, governance, and land use history at stream and stormwater drainage network to landscape scales.

Background and Past Results. As in grasslands, greater diversity in forests has been linked to higher productivity and biomass, in both experimental studies (Grossman et al. 2018) and across gradients of tree diversity in forest inventory analyses (Liang et al. 2016). However, whether greater diversity in forests also leads to greater stability has been challenging to ascertain. Most tree diversity experiments have not been undertaken long enough to allow evaluation of diversity-stability relationships, and forest inventory

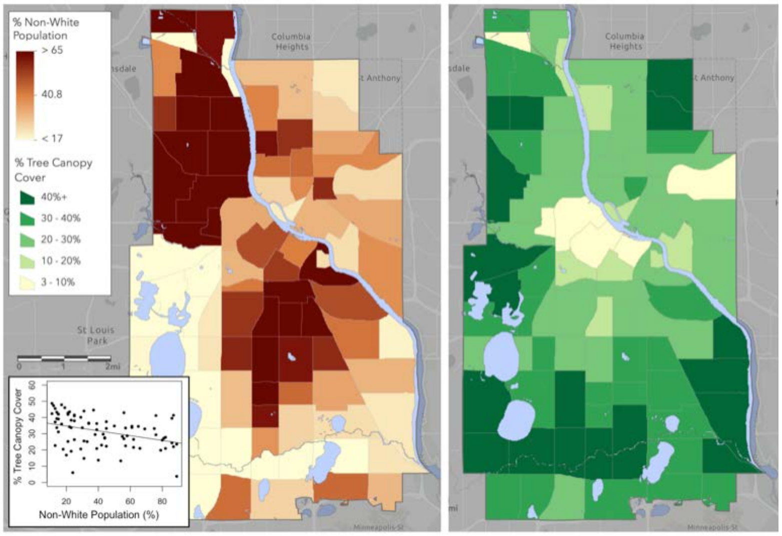


Figure 9. Comparison of racial composition and tree cover of Minneapolis neighborhoods. (Left) Percent non-white population is calculated as the total population minus the non-Hispanic white population, divided by the total population using 2013-2018 American Community Survey data. Tree canopy cover by neighborhood estimated from high-resolution (1m) tree canopy estimates conducted using LiDAR imagery (Knight et al. 2017). (Above) Graph of tree canopy cover vs. percent non-white population by neighborhood, with solid line representing the linear relationship ($y=-0.15x+37.7$, Adjusted $R^2=0.12$, p-value<0.005). Tree cover is also related to income: (% tree canopy=0.0002*income + 19.29, Adjusted $R^2=0.37$, $p<0.005$).

analyses are often snapshots in time. In cities, social factors are likely to be important in influencing canopy dynamics, along with diversity. For example, a recent analysis showed that in 37 U.S. cities, racially discriminatory lending policies were strongly associated with tree canopy cover, with canopy cover in predominantly U.S.-born white neighborhoods twice that in neighborhoods dominated by racial and ethnic minorities (Locke et al. 2020). Such patterns are evident in the city of Minneapolis (Fig. 9).

We aim to address key gaps in our understanding of urban tree diversity-stability relationships: (1) whether canopy *diversity* reflects socioeconomic patterns as has been shown for canopy cover, (2) whether patterns of tree diversity are related to resilience and resistance to disturbance and climate extremes, as has been documented in grasslands, and thus to long-term canopy dynamics, and (3) whether policies aimed at reversing social disparities will also influence canopy cover and diversity over the long-term. We will address these uncertainties for urban tree canopy because of its importance for influencing urban climate, hydrology, and water quality (**Q2**), and ecosystem benefits (**Q3.1**) such as shade and evaporative cooling.

Research Methods. To test how intentionally designed tree canopies can better withstand long-term disturbance at the **habitat patch** scale (H1.2.4), we will take advantage of a newly installed replicated experiment in an urban park in St. Paul, the first urban affiliate site in the adaptive silviculture for climate change (ASCC) network (Nagel et al. 2017). Scientists and local managers worked collaboratively to design three contrasting management treatments for areas with massive canopy loss resulting from emerald ash borer (Hammes et al. 2020). Twenty-four permanent 0.04-ha plots were installed in 2019 and planted with either (1) a mix of five tree species native to the park, (2) a mix of eight species native to southern Minnesota and northern Iowa that are projected to be adapted to near-term climate conditions (30-60 years) based on habitat suitability modeling, (3) a mix of eight species native to the U.S. Midwest that are expected to be adapted to long-term projected climate conditions (60-80 years) based on habitat suitability modeling, or (4) control (no planting). Plots will be measured annually for biodiversity, productivity, and evidence of disease and survival of planted trees using protocols developed for the ASCC network (Nagel et al. 2017).

To test whether resilience and/or resistance are related to canopy diversity at larger spatial scales (H1.2.2), we will relate patterns of canopy diversity and composition across MSP to tree productivity over time, using long-term satellite records. We will use remote sensing data going back nearly four decades (Landsat, since 1984) to understand long-term trends and variation in the Normalized Difference Vegetation Index (NDVI) and other spectral indicators of productivity at coarse spatial scales. Forthcoming satellite imagery will enable mapping and monitoring of tree diversity and composition at finer spatial scales through time, once the Surface Biology and Geology and Copernicus Hyperspectral Imaging Mission for the Environment (CHIME-Sentinel 10) are launched. A suite of annually collected, remotely sensed imagery will enable us to establish maps of canopy cover and composition at varying levels of composition and accuracy through the "NextView" partnership between Maxar, Inc. (previously

DigitalGlobe, Inc.), the largest provider of commercial high-resolution satellite imagery, and the National Geospatial-Intelligence Agency, which provides no-cost access to Maxar imagery for federally funded researchers. Minnesota also has statewide growing season 1-m National Agricultural Imagery Program datasets (2003, 2008, 2009, 2010, 2013, 2015, 2017, 2019), 0.6-m spring imagery (various dates 2010-2015), and lidar. In addition, the UMN Remote Sensing and Geospatial Analysis Laboratory (RSGAL, Senior Personnel Knight, Director) has created multiple geospatial datasets, including 30-m land cover/use classifications of MSP (2000, 2007, 2011), and 1-m statewide canopy height models (2010, 2019). Ongoing RSGAL work will produce 1-m urban forest cover maps for 300 municipalities.

We will stitch together geo-referenced tree inventory data from individual municipalities, plot-level data from recent studies conducted in MSP, Twin Cities Urban Forest Inventory and Analysis data, and data collected previously for yards and for relatively unmanaged natural areas within MSP as part of past NSF-funded studies (Knapp et al. 2012, Padullés Cubino et al. 2019) to calculate measures of diversity within and across neighborhoods and municipalities (linking to **Q2** and **Q3**). Where we lack ground-based compositional data, and to measure diversity over time, we will develop statistical models to predict species, lineages, and/or functional trait composition from remotely sensed hyperspectral data (from the DESIS sensor on the International Space Station and from 1-m multi-spectral (8 band) WorldView imagery) (Foster and Townsend 2004, Singh et al. 2015, Pontius et al. 2019, Wang et al. 2019, Sapes et al. In preparation). We will collect additional ground-level data for model validation and use the model to develop a fine-scale map of tree diversity and composition for regions of MSP.

To evaluate how and whether diversity, composition, and canopy response to disturbance vary with socioeconomic factors (H1.2.3) at the **drainage network** scale (Box 2), we will test for associations of canopy diversity, productivity, resilience, and resistance with past and present median household income, level of education, racial diversity, and other factors from U.S. Census data at the census block level. We will also consider associations with past discriminatory housing policies, including discriminatory lending policies (i.e., redlining) and racial housing covenants, as described in activities under **Q.3.1**.

We will test how drivers of canopy composition, diversity, and responses to disturbances and extreme climate vary with grain size and spatial extent (H1.2.4) using a modeled composition map and long-term trends in productivity and cover from remote sensing data at two scales. Within the cities of Minneapolis and St. Paul, we will examine how effects of recent disturbances (2011 tornado in north Minneapolis, summer 2013 storms across Minneapolis and St. Paul, emerald ash borer outbreak from 2009-present) vary across census blocks to relate economic, racial, and tree species compositional differences to canopy resistance and resilience (linking to **Q3.1**). We will also examine how neighborhood-scale actions to increase canopy cover and diversity following these events affect resistance and resilience moving forward (linking to **Q3** and **Q4**). At the broader MSP **landscape** scale, we will examine how differences in management approach and investment in urban tree canopy planting and maintenance across municipalities have affected and will continue to affect resistance and resilience (linking to **Q3.2**).

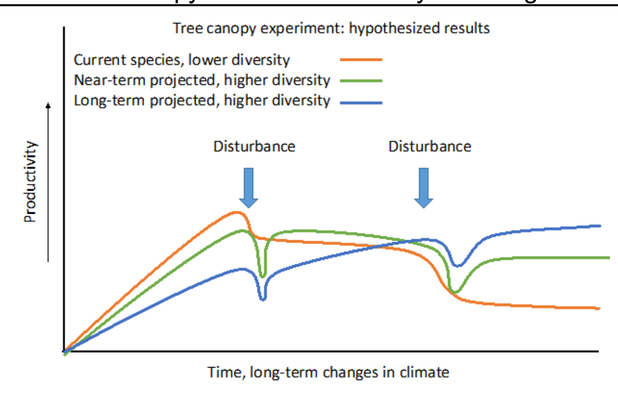


Figure 10. Hypothesized changes in productivity in an intentionally designed tree canopy experiment. Low diversity treatment with species adapted to current climate (current species) will have higher productivity initially, but higher diversity treatments that incorporate species from just south of MSP (near-term) or further south (long-term), are projected to gain suitable habitat and thus will have higher resilience following disturbance, and higher productivity in the long-term.

Expected results. In our long-term experiment, at the **habitat patch** scale, we expect locally adapted species will have greater survival and productivity over the short term (1-10 y), but that canopies designed based on habitat suitability modeling will better withstand disturbance and be more stable over the long term (10-50 y, Fig. 10). Within the urban core (cities of Minneapolis and St. Paul), we expect that tree canopy resistance and resilience will be driven by past legacies and present conditions of racial and other socioeconomic differences across neighborhoods (as characterized under **Q3**). Historically underserved neighborhoods with high local action and engagement will see increases in tree canopy cover and tree taxonomic and phylogenetic diversity, resulting in long-term resistance and resilience to disturbance. At the **landscape** scale, differences in tree canopy composition, resistance, and resilience will be driven by the interaction of underlying biophysical factors, land use history, and past and present management decisions and investments across local governance structures.

Q1.3. How do the amount, type, and configuration of urban nature affect insect community structure and population dynamics?

Hypothesis 1.3.1. Pollinator abundance and diversity will vary with different types of land management and socioeconomic status, whereby green spaces managed for pollinators and located in more affluent neighborhoods (with greater floral diversity) will support more diverse and abundant pollinator communities and ecosystem services.

Hypothesis 1.3.2. Density, spatial configuration, and design of land managed for pollinators will influence pollinator abundance, diversity, and population dynamics, whereby larger, more abundant, and more connected habitat managed for pollinators will support more abundant and diverse bee communities, and support viable populations over time.

Background and Past Results. Over 40% of insect species, including bees and other pollinators, are experiencing widespread decline, with some species threatened with extinction (Winfree et al. 2009, Vanbergen et al. 2013, Sánchez-Bayo and Wyckhuys 2019). Such declines have cascading impacts on human well-being and local, regional, and global ecosystem processes (e.g., Gallai et al. 2009, Ollerton et al. 2012). Although these declines are largely driven by habitat loss from urban development and agricultural intensification, ecologically oriented design and management of urban spaces have the potential to help offset declines and transform cities into refugia for pollinators and other wildlife (Hall et al. 2017). Such an ecological urban transformation can potentially reduce inequities in benefits of urban nature (i.e., reverse the ‘luxury effect’, whereby more affluent neighborhoods support higher biodiversity) (Leong et al. 2018), with implications for delivering ecosystem services (Lerman and Warren 2011).

One important target for urban habitat transformation is traditional turfgrass lawns, which cover more than 163,000 km^2 of U.S. urban area (Milesi et al. 2005). The primary motivation for the redesign of traditional lawns is to create an aesthetically pleasing landscape that is easy to maintain (Larson et al. 2016). However, research in MSP suggests that alternative, pollinator-friendly habitats such as “flowering bee lawns” (i.e., turfgrass seeded with pollinator-friendly flowers) can have similar aesthetic and practical value (Ramer et al. 2019). Furthermore, research in eight MSP city parks found that florally enhanced bee lawns supported more diverse and distinct bee communities compared with lawns without enhancement (Wolfin 2020) (Fig. 11). Widespread interest in supporting pollinator habitat on the part of lawmakers and private citizens alike led to the creation of the “Lawns to Legumes” grant program by the state of Minnesota, a program first implemented in 2020 that provides funding and other resources to homeowners to plant turfgrass lawns with pollinator-friendly flowers, to create bee lawns. However, the long-term consequences of this program for pollinator populations and communities has yet to be determined.

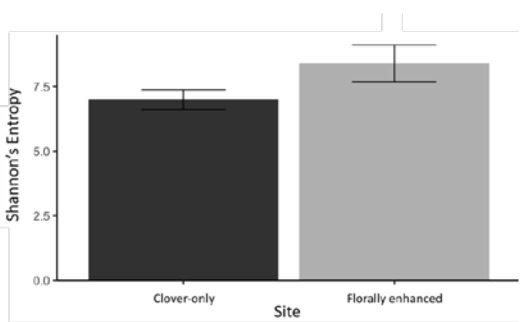


Figure 11. Effectiveness of bee lawns for supporting pollinators. Exponential Shannon's index of entropy at clover-only and florally enhanced parks in Minneapolis, Minnesota, 2016-2018. Parks with bee lawns (i.e., florally enhanced) supported greater bee diversity (p= 0.046) than clover-only parks (i.e., traditional turf lawns). Data from Wolfin 2020.

We aim to address two interrelated knowledge gaps relevant to managing insect pollinators in cities: (1) how turfgrass-to-bee lawn transformation interacts with local habitat features (e.g., floral abundance, bare soil, and ground cover), landscape features (e.g., impervious surface and fragmentation), and social factors (e.g., income and race) to influence community structure, population dynamics, and pollination by insect pollinators; and (2) how effective bee lawn programs are for providing urban habitat for pollinators and increasing pollinator abundance, diversity, and pollination services.

Research Methods. We will combine long-term monitoring, experimentation, and modeling at **habitat patch** to **landscape** scales to assess interactive effects of land management and social factors on bee abundance and diversity. First, we will compare bee lawns (n = 40) and traditional turfgrass lawns (i.e., fertilized and not managed for bees; n = 40) (H1.3.1), coordinating site selection with **Q1.1**, **1.2**, **2.1**, and **3.2**. We will identify and seek permission from households that have implemented bee lawns by partnering with local seed distributors (Twin City Seeds, MN Board of Water and Soil Resources; see letters of collaboration), locating yard sites broadly across socioeconomic gradients throughout MSP. Annually, we will sample for bees twice/year (May-June and July-August), and identify and quantify bee communities (richness, abundance, and diversity) using pan-traps set for 24 hours (30 traps/parcel per sampling round) and a 15-min sweep-net survey (Lerman and Milam 2016). We will annually sample plant communities (species richness, floral abundance and diversity, bare ground cover) within study lawns to test bee community-plant community associations (12 1-m^2 quadrats/lawn). To quantify soil effects on plant and bee communities, and to integrate with **Q1.1**, we will compare soil characteristics (compaction, moisture, texture, OM, pH, nutrients, metals) between bee- and non-bee lawns. In a subset of yards (n = 20 bee lawns, n = 20 traditional lawns), we will quantify pollination services using an assay that compares fruit set in potted tomato plants introduced to yards for short periods (Potter and LeBuhn 2015).

Second, we will use three long-term experiments to determine how density, spatial configuration, and nesting habitat of bee lawns influence pollinator abundance and diversity (H1.3.2). Expt. 1: In year 2, we will expand research to eight city parks that have participated in past bee lawn research (Wolfin 2020), again in coordination with **Q1.1**, to test how bee lawn size and density influence bee communities. At each park, we will establish a new long-term experiment that varies the numbers and sizes of bee lawns in a park, spanning the size range found in yards surveyed for H1.3.1, and with some parks having only one patch and others having multiple patches. We will sample soils, plants, and bees as described above. Expt. 2: In year 3, we will install a second long-term experiment, consisting of artificial "bee hotels", i.e., nest boxes for cavity-nesting bees such as mason bees (*Osmia* spp.) and leafcutter bees (*Megachile* spp.), to test whether nesting site availability influences population dynamics and community structure in lawns. We will install bee hotels at half of the bee lawn sites (n=20) and half of the turfgrass sites (n=20) using a full-factorial design. Each winter thereafter (when bees are in diapause), we will measure bee hotels for % occupancy (Fortel et al. 2016) and test for associations between bee abundance and diversity on lawns and bee hotel occupancy. Expt. 3: In year 4, after developing relationships with neighborhood associations and community organizations, we will establish an experiment similar to Expt. 1, but in private yards and public spaces (medians, roundabouts, boulevards) within neighborhoods, to test whether effects of size and configuration of bee lawns on pollinators depend on contextual factors such as impervious cover and socioeconomic gradients (as identified in H1.3.1) at the **landscape scale**.

We will use the results of observations and experiments to parameterize and validate the InVEST pollinator abundance model (Lonsdorf et al. 2009, Davis et al. 2017, Sharp et al. 2018). Urban InVEST is a suite of models aimed at aiding decision-making by visualizing trade-offs among different management options. The InVEST pollinator model predicts how changes in lawn management affect urban bee abundance and diversity at broad scales. Briefly, this spatially explicit model determines a bee fitness index as a function of the density and quality of nesting and forage resources within foraging distance of the nest. The model applied to a **landscape** predicts the relative abundance of bees visiting flowers at each **habitat patch**. We will parameterize InVEST using the observational and experimental studies described above and use machine learning to best explain patterns of abundance for each bee species (Groff et al. 2016). Using InVEST will allow us to estimate the floral value of bee lawns compared to traditional turfgrass lawns given the broader foraging landscape surrounding the experimental areas. Finally, we will integrate with **Q3.2** to jointly investigate how bee lawn policies, programs, and practices contribute to equitable expansion of ecosystem services across socioeconomic gradients.

Expected Results. We predict bee lawns will increase bee abundance, diversity, and pollination services relative to traditional turfgrass lawns. Further, we predict that effect sizes will depend on lawn size and density, and other landscape features (fragmentation, other available floral resources), many of which will be correlated across socioeconomic gradients. By experimentally testing bee responses to different sized patches and their configuration within parks, we can identify mechanisms shaping urban nature. By applying these results back to neighborhoods, we can advance understanding of how bees move across the urban landscape, and the degree to which bee lawn enhancements can meet species requirements. Our integrative and long-term research will assess filtering processes (nest sites, floral resources) structuring bee populations and communities (Aronson et al. 2017). By targeting neighborhoods along socioeconomic gradients, we explicitly test whether inequities exist with access to urban nature. Further, results will identify the distribution of pollination services across MSP, with implications for wildflower populations and urban agriculture (Theodorou et al. 2020) and thus help inform how to manage lawns for multiple ecosystem services (e.g., biodiversity, pollination, stormwater management). Research has direct applicability for bee lawn program managers by providing detailed information on optimal soil properties, plant mixes, lawn size, and locations for supporting pollinators.

Q2. How do the ecological structure and function of urban nature interact with social and technical factors to influence urban climate, hydrology, and water quality of watersheds and lake ecosystems over annual to decadal timescales?

Q2.1. How does urban vegetation influence nutrient runoff, moderate the urban heat island effect, and deliver ecosystem services through its spatial configuration and functional connectivity?

Hypothesis 2.1.1. At the habitat patch scale, effects of urban vegetation on runoff and nutrient and OM export are determined by structural attributes (e.g., proximity to impervious surfaces, biomass, canopy cover, infiltration capacity) during rain events and snowmelt and by functional differences (e.g., transpiration) between rain events. Effects of urban vegetation on the UHI effect are determined by both structural and functional differences.

Hypothesis 2.1.2. At the watershed scale, effects of urban vegetation on runoff, export of pollutants, and the UHI effect are determined by spatial distribution and connectivity to the stormwater drainage network.

Hypothesis 2.1.3. Over the long-term, effects of climate and management on the transport of water, nutrients, OM, and other pollutants through urban watersheds are mediated by legacy accumulation of nutrients and contaminants in soils and sediments of residential landscapes, stormwater management structures, and aquatic systems.

Background & Past Results. Previous studies in MSP have (1) quantified the UHI effect at daily, seasonal, and annual timescales (Smoliak et al. 2015), and (2) shown that urban vegetation structure interacts with climate, management, and stormwater drainage network properties such as road density and impervious cover to affect fluxes of water and nutrient pollutants in urban drainage networks. For example, ET rates in recreational and residential areas differed significantly between trees and grasses and across seasons (Peters et al. 2011). These functional differences likely result in varying effectiveness of delivering UHI benefits at different temporal and spatial scales, although this has yet to be characterized across MSP. Additionally, past work in MSP has shown that trees can be a major source of nutrient pollutants to stormwater, with higher concentrations and export of stormwater P in watersheds with higher street tree canopy (Janke et al. 2017) (Fig. 12). Trees serve as nutrient "pumps" that take up soil P and deposit litterfall P in streets (Kalinosky et al. 2014) (Fig. 13), where it moves readily into storm drainage networks. Residential streets (essentially urban "headwater streams") have little capacity to retain P or OM (Hobbie et al. 2017) (Fig. 14), especially during spring snowmelt and intense rain events (Bratt et al. 2017) (Fig. 15).

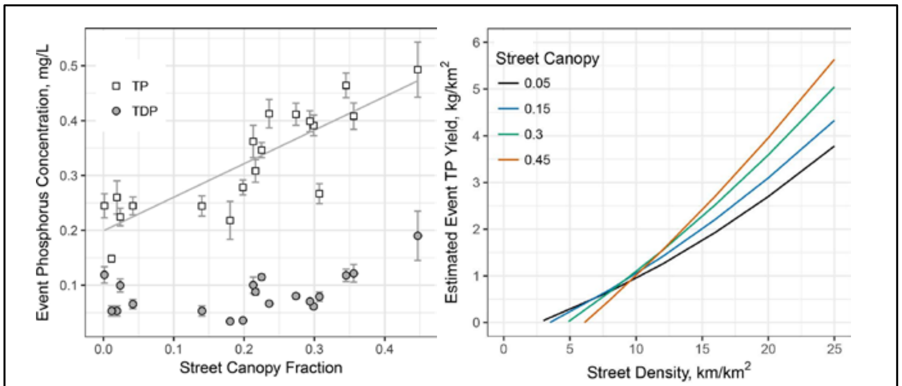


Figure 12. Concentrations and yields of stormwater P in relation to canopy cover and street density in MSP watersheds. Shown (left) are site means ±SE of event total P (TP) and total dissolved P (TDP) concentration in stormwater vs fraction of street covered by tree canopy (n = 19 sites). Trend lines indicate significant relationships. Shown (right) are estimated mean event yields (kg/km^2) of TP as a function of street density (km/km^2) for fixed levels of street canopy cover. Yields were estimated from the product of event mean concentration (mg/L) and event mean water yield (cm) across a gradient of street density with four levels of street canopy that spanned the ranges observed across the watersheds. From Janke et al. 2017.

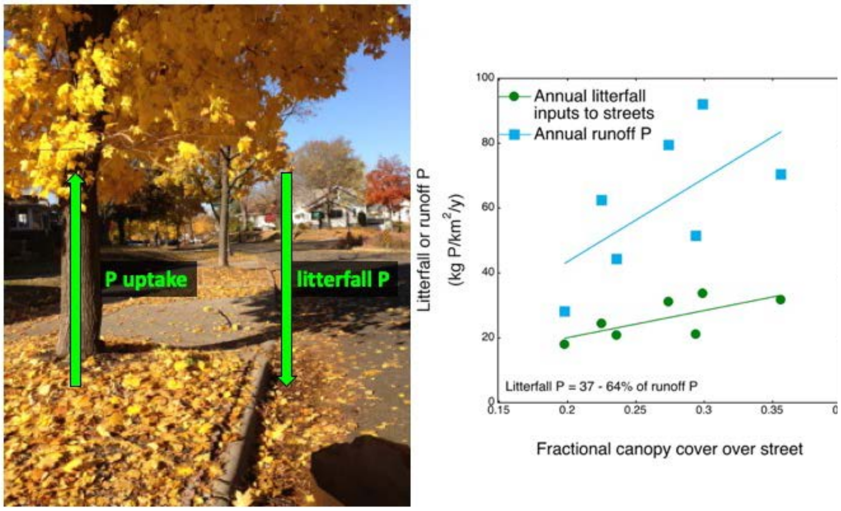


Figure 13. Contribution of tree leaf litter nutrients to stormwater nutrient pollution. Trees adjacent to streets take up nutrients from soils and drop some fraction of those into the street in annual litterfall (left). Unless nutrients are removed during street sweeping operations, they will eventually end up in stormwater. Across seven St. Paul watersheds (right), litterfall contributes 37-64% of total P export in stormwater runoff. Hobbie et al. 2017 and unpublished.

To capture the spatial configuration and flow pathways for water, nutrient, and heat exchange between different land cover components, we conceptualize the watershed as a network (e.g., Newman (2018) of "nodes" (sinks and sources) and "links" (connecting adjacent nodes). This approach mirrors that used to characterize natural stream networks (Rinaldo et al. 2006) and can determine how variation in urban networks influences their hydrologic response, delivery of benefits and burdens, and resilience to climate extremes (Jefferson et al. 2017). Similar network-based approaches in Phoenix, AZ, showed that the evolution of the urban stormwater network resulted in sudden shifts in hydrological functioning induced by urban development and changes in connectivity (Jovanovic et al. 2019). The relative importance of network structure and dynamics in this arid watershed also changed across spatial scales. In the Baltimore Ecosystem Study, riparian zones within the urban stream network also exhibited reduced capacity for nutrient and flow retention (Walsh et al. 2005).

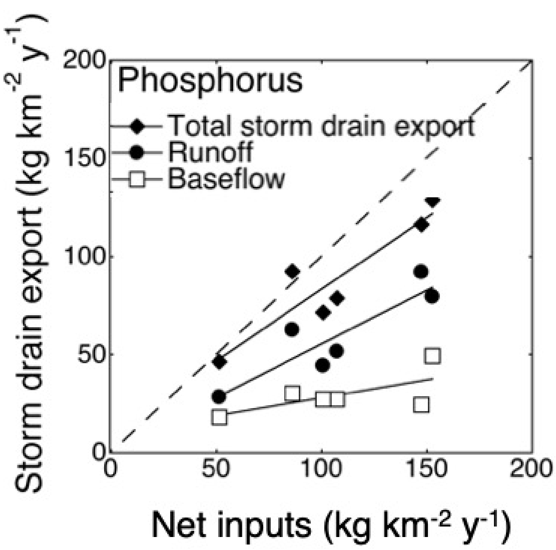


Figure 14. Relationships between net watershed inputs of P and export in storm drains in six St. Paul watersheds (each point represents a watershed). The dashed line indicates a 1:1 line, where net inputs = outputs. P retention averaged 22% across watersheds. From Hobbie et al. 2017.

We aim to fill key gaps in our understanding of the functioning of urban drainage networks in humid, cold, continental watersheds, where perennial lakes and floodplains within low-relief topography may uniquely disperse and retain flow, and where significant flow occurs during snowmelt and increasingly intense rainfall events. We address: (1) how the structure and function (ET) of vegetation influence export of water, nutrients, and OM, and UHI effects, including during snowmelt and rainfall events of varying intensity; (2) how stream and stormwater drainage network configuration and connectivity (characterized by e.g., network clustering, density, and sparseness) influence flows of water, nutrients, and heat (including UHI effects); and (3) how urban vegetation type and management and stormwater management interact with stormwater network connectivity to influence water quality in stormwater and urban streams.

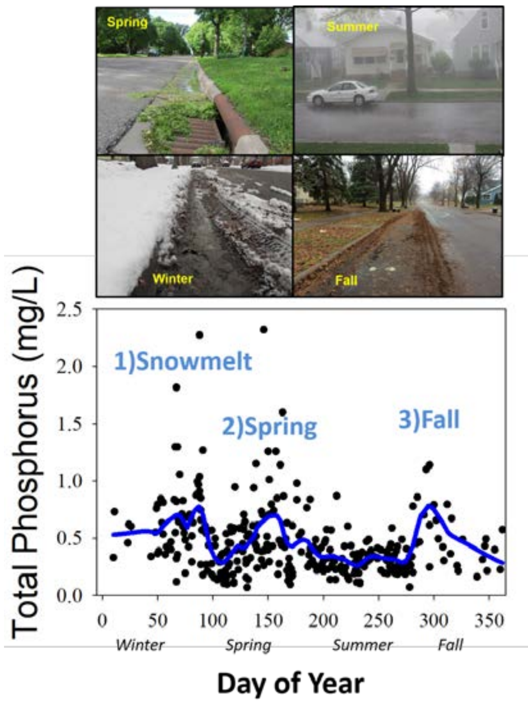


Figure 15. Influence of vegetation phenology and climate on total P concentrations in stormwater runoff for a small, storm-drained residential watershed in St. Paul, monitored by UMN and local partners for 10 years. Extensive boulevard trees line the streets (a), and contribute to peaks in P concentration (b) and load (not shown) at snowmelt, spring leaf out and during fall leaf drop. Runoff water quality is generally highest during summer conditions. Total organic carbon concentrations in stormwater are typically over 30mg/L during spring and fall runoff events, levels typical of wetland watersheds. Finlay et al. unpublished.

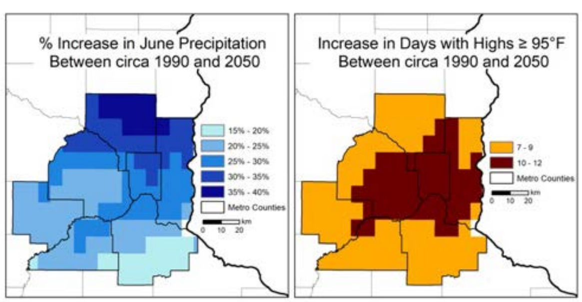


Figure 16. Predicted increases in temperature (left) and precipitation (right) for MSP for 2040-2059 (compared to 1980-1999). From 1980-1999 MSP had ca. 3 days/y with high temperatures >=95°F. Climate change projections indicate an increase of 7-12 days exceeding this threshold by mid-century (2049-2059). While June is already the wettest month in Minnesota, it also had the largest increase in precipitation between the modeled historical and mid-century periods. Projections are an ensemble of seven dynamically downscaled global models: bcc-csm1-1, CCSM4, CNRM-CM5, GFDL-ESM2M, IPSL-CM5A-LR, MRI-CGCM3, and MIROC5. From Noe et al. 2016.

Research Methods. We address our hypotheses with a three-tiered approach. First, at the **habitat patch** scale (Box 2), we will collect new data in 6-8 local sites nested within modeled watersheds (see below) to investigate the effects of land cover types (forests, shrublands, turfgrass) on the mitigation of the UHI effect, stormwater runoff, and OM export. We will coordinate site selection with various watershed management organizations (see letters of collaboration) and **Q1** and **Q3** to (1) span a gradient of urban development within MSP and (2) represent different urban land cover types and connectivity of impervious cover. In places with existing monitoring efforts, new measurements will augment existing data and infrastructure in terms of key missing variables and temporal and spatial coverage (e.g., Bratt et al. 2017, Hobbie et al. 2017, Janke et al. 2017). Specifically, we will collect new snowmelt samples to complement existing measurements of flow rates and rain event samples, and analyze them for solute concentrations, including N, P, and OM. We will also install soil moisture and air temperature sensors at each site, at two different soil depths and under different vegetation covers, to monitor the evolution of plant water use and quantify the contribution of ET to UHI mitigation. In conjunction with nearby weather station data on air temperature, precipitation, solar radiation, wind speed, relative humidity, and water vapor concentrations, these measurements will provide a key linkage between new ET measurements and the high-density air temperature dataset of UHI (Smoliak et al. 2015) to separate the functional cooling effect of ET from the reflective effect of surfaces on UHI mitigation. Remotely sensed datasets of ET and surface temperature will be used to augment local measurements at a larger scale.

At the **drainage network** scale, we will develop network models for 4-6 watersheds to investigate the effects of spatial distributions of urban vegetation on the transport of water, nutrients, and OM, and their contribution to the UHI effect. Within the network model, new patch scale data (described above) will be mapped onto existing land cover data (from RSGAL) to parameterize hydrological responses of each land cover type. Existing data on stormwater infrastructure will be used to construct the dynamics of stormwater pathways over time. We will validate predictions from the network model using water quality and quantity data from urban lakes, streams, and stormwater management structures. We will use machine learning techniques such as random forest regression (e.g., Crompton et al. 2019) to classify and predict the watershed response based on various metrics of network connectivity and other climatic and landscape features and identify the most important climatic and landscape drivers of watershed scale runoff response and UHI mitigation. To extrapolate these outcomes to past and future scenarios, we will apply historical and future climate change (Noe et al. 2019) (Fig. 16) and urbanization trajectories (based on historical maps of stormwater networks available from watershed districts) to the network model to examine the sensitivity of its responses, including any legacy and threshold-like behaviors.

Finally, at the **landscape** scale, we will leverage past and ongoing monitoring of 32 storm drain and ~20 stream sites with diverse watershed features, including social factors, vegetation, and impervious cover characteristics. Various project members have used subsets of these data for past analyses of nutrient dynamics (Janke et al. 2014), tree canopy effects on P loads (Janke et al. 2017), and snowmelt-induced nutrient and Cl dynamics (Bratt et al. 2017). Existing land cover maps, digital elevation models, stormwater management databases, and stormwater network data will be obtained from cities, watershed management organizations, and soil and water conservation districts (see letters of collaboration). We will also quantify the legacy effects of soil nutrient accumulation through existing data on soil N and P concentrations (see **Q1.1**). These datasets will be used to derive different metrics for spatial patterns of urban vegetation type and management, stormwater management, and stormwater network connectivity (such as network clustering, density, and sparseness). These metrics will be analyzed against existing water quality and quantity data from urban streams, storm drains, and stormwater management structures to reveal long-term temporal and spatial correlations.

Expected Results. These activities will elucidate the tradeoffs between the benefits (UHI and runoff mitigation) and burdens (nutrient and OM inputs) of various types of urban nature at the **patch scale** and how these tradeoffs vary over event and seasonal timescales (H2.1.1). The network-based modeling approach in combination with machine learning classifications will identify the most important climatic and landscape metrics of stream and stormwater **drainage network** responses (H2.1.2). Analysis of historical and spatially extensive watershed monitoring sites will determine long-term relationships between climate, urban development, and stormwater management, along with storm and stream water and flux transport, including effects of legacy accumulation of nutrients in residential landscapes (H2.1.3).

Q2.2 How do management activities along urban hydrologic flow paths interact with urban development and climate change to determine the long-term fate and transport of nutrients and OM in urban watersheds and their impacts on urban lakes?

Hypothesis 2.2.1. Urban lake water quality is determined by long-term interactions among watershed structure and management, climate, and lake morphology, with shallow lakes showing greater sensitivity to climate change-driven increases in temperature and runoff in terms of eutrophication, compared to deeper lakes, because of effects on dissolved oxygen regimes and organic matter dynamics.

Background and Past Results. Regional syntheses of factors influencing lake water quality trends show strong signals of both climate change (more intense precipitation and higher temperature) and land use change (Collins et al. 2019). However, in urban areas, some of the most important features affecting lake ecosystems, e.g., configuration of intensive subsurface storm drain networks and legacies of past land use and management, are not easily captured in such syntheses. For example, since the 1970s U.S. cities have made massive investments in water management, which have led to dramatic declines in P pollution from sanitary waste. However, despite large investments in stormwater management, lake water quality outcomes (i.e., water clarity) show diverse trends, with many lakes showing no change and some lakes showing declines in water quality (Fig. 17). Such varied trends in lake water quality likely arise from complex interactions of land use change (e.g., urban expansion) and cover (e.g., canopy cover), increasing runoff caused by more intense rain events, lake-specific management, and in-lake characteristics (Soranno et al. 2015, Collins et al. 2019). Management activities are tailored to individual lakes in response to impairment listings, social drivers, and geopolitical boundaries. In addition, internal factors such as lake morphology and size, which influence water mean residence time, stratification, and mixing, are likely important sources of variability in water quality responses to climate change and other human perturbations (e.g., Jeppesen et al. 2009, Lisi and Hein 2019, Tammeorg et al. 2020).

We aim to address significant gaps in our understanding of trends in lake water quality, especially why some lakes are improving but many others are not. Towards this end, we will: (1) determine how

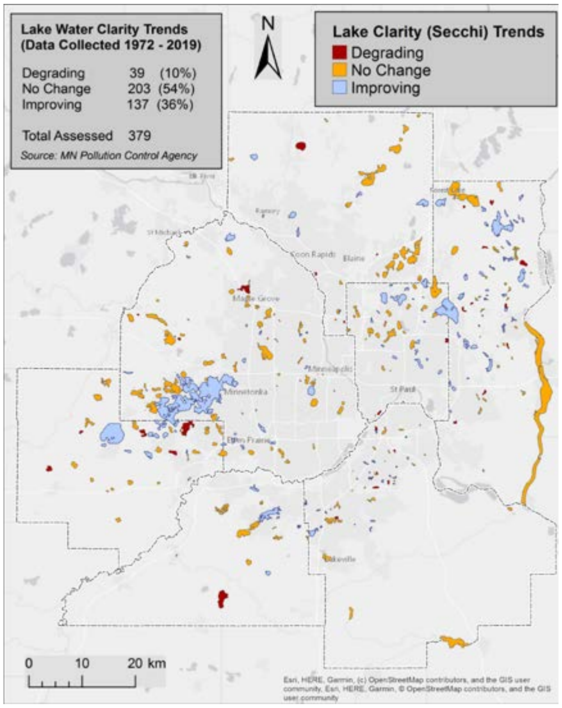


Figure 17. Trends in lake water clarity for MSP lakes. Trends are based on Secchi depth.

management, climate, and biophysical setting (especially urban stormwater drainage networks) interactively affect delivery and fate of water, heat, nutrients, and contaminants to lakes; (2) determine how these inputs in turn interact to influence long-term changes in limnological conditions that affect water quality, ecosystem functioning, and services; and (3) combine detailed, year-round investigations of focal lakes with models and analyses of long-term data for a much larger and diverse set of >150 lakes ranging in size (<2 to 800 ha), maximum depth (<0.5 to 42 m), and water quality (e.g., nutrient-poor to hypereutrophic) to test hypotheses and predictions. Research on lakes in Q2.2 will closely link to measurements and models that examine terrestrial processes, especially nutrients and OM inputs from vegetation (**Q1.2**) and variation in watershed structure and management along flow paths (**Q2.1**).

Research Methods. We will integrate approaches at three scales: (1) analyzing a rich assemblage of lake water quality data at the metropolitan-wide **landscape** scale, providing context for investigations of **Q1**, **Q3**, and **Q4**, (2) developing linked watershed-lake models to assess how climate and management change affect lake water quality, and (3) measuring key processes that determine urban lake responses to interacting stressors in focal lakes. First, at the landscape scale, we leverage existing data and ongoing monitoring to address how the nature and duration of human influences affect urban lake water quality responses. Data for stormwater, streams, and long-term (20-50+ y) records for >150 lakes (Figs. 4, 17) will be analyzed by integrating datasets for land cover, water quality, watershed management, and climate. Understanding lake water quality responses in diverse social and governance settings will drive integration with **Q3** and **Q4** and inform development of models and measurements described below.

Second, we will develop coupled watershed-lake models that are based on detailed analyses of watershed inputs developed in **Q2.1** to understand outcomes of management, structure, and climate change in determining biogeochemical processes and lake water quality. Model development (sensu Motew et al. 2019, Small et al. 2019b) is essential because many urban impacts are strongly interactive and depend on detailed understanding of watershed and lake features (e.g., drain pipe density, lake depth), management (e.g., street sweeping, stormwater management structures, road salt use, alum treatment), and climate. We will develop models to determine the interactive, long-term effects of warming, rising salinity, management, and increasing runoff on water quality, and leverage long-term watershed and lake datasets for model validation. Model predictions will combine physically based models with machine-learning algorithms to improve model accuracy (Hanson et al. 2020). We will focus on assessing sensitivity to future climate and management scenarios, using down-scaled climate and UHI projections (Noe et al. 2019) in conjunction with watershed and lake management scenarios.

Finally, we will develop a long-term measurement program of key biophysical processes to understand urban impacts on lake biogeochemistry. We focus on two intensively managed iconic (i.e., heavily visited) urban lakes that represent contrasting recreational lake types in MSP. Both lakes have intensively storm-drained watersheds and long, well documented management histories, but contrast in their depth, and thus in their presumed susceptibility to climate and management driven change. Specifically, nutrient cycling, salt, and climate impacts are likely to show strong contrasts between shallow, frequently mixed Como Lake and deeper, strongly stratified Lake McCarrons, which may be approaching meromixis due to hypolimnetic salt accumulation. Both have been the focus of intensive efforts over the past 20 years to improve lake water quality, with little success. Our measurements will focus on high-resolution monitoring, including under ice, of water column temperature and conductivity (to measure dynamics of heat and salt inputs from runoff, and determine lake stratification, mixing, and oxygen regimes), nutrient cycling (focusing on N and P), whole-lake metabolism using measurements of oxygen and N_2 and sediment core studies (Small et al. 2013, Small et al. 2016, Bratt et al. 2017, Small et al. 2019b, Taguchi et al. 2020).

Expected Results. These analyses will elucidate relationships between the effects of nutrient and stormwater management strategies along urban hydrologic flow paths and climate-driven changes in precipitation on transport of nutrients and, ultimately, on urban lake quality. By leveraging extensive existing watershed and lake data, the model development described here will allow us to explore novel questions, such as the effects of changing urban tree canopy (e.g., due to invasive pests and pathogens, integrating with **Q1.2**) on lake water quality, how management of public and private land differs in higher- and lower-income communities (integrating with **Q3**), and how these decisions in turn impact environmental benefits and burdens provided by lakes for urban residents (integrating with **Q3-4**).

Q3. How are decisions about urban nature, community wealth, and well-being coupled over space and time to affect social inequities; how can governance institutions be changed to better address equity such that environmental outcomes and human well-being are improved for all urban residents?

Q3.1. How are urban nature decisions coupled to community wealth and human well-being over time and space?

Hypothesis 3.1.1. Greater historical investments in urban nature in white communities have generated wealth in those communities, exacerbating racial disparities between white and BIPOC communities.

Hypothesis 3.1.2. Investments in park creation, maintenance, amenities, and programming have had and will continue to have the largest impact on property values, relative to other types of urban nature, such as trees, lakes, and lawns.

Background and Past Results. On average, Black and Latinx households in the U.S. have less than 10% of the accumulated wealth of a typical white household (Sullivan et al. 2015), where wealth refers to the value of all household assets and is correlated with almost every major indicator of human well-being (e.g., health, education, vulnerability to threats, resilience to disasters). Homeownership significantly contributes to this racial wealth gap, with white homeowners gaining, on average 34% and 54% more wealth through homeownership than Black and Latinx homeowners, respectively (Sullivan et al. 2015). Racial disparities in wealth and income are well-documented, but it is unclear how alternative investments in urban infrastructure, including urban nature, have contributed to these gaps.

Minneapolis's racially exclusionary covenants increased average present-day house values by 15% compared to properties that were not covenanted (Sood et al. In preparation). Our ongoing work suggests that the historic locations of these covenants are correlated with the distribution of parks and park amenities (Keeler et al. unpublished), underscoring the importance of conducting historically grounded research that seeks to understand environmental justice as more than a snapshot of correlations. Our research includes interacting, multi-scalar social, infrastructure, and ecological systems that shape people and nature (Grove et al. 2018). Long-term data collection across social and ecological domains presents an opportunity to advance understanding of how investments in urban nature explain variation in social outcomes, explore and unpack the historical dynamics that gave rise to those outcomes, and develop insights into future policies and programs that can address social and environmental justice.

We aim to address several knowledge gaps in our understanding of the coupling between social and ecological systems in cities: (1) how institutional and policy processes and decisions have contributed to the unequal distribution of environmental benefits and burdens in MSP, (2) which aspects of urban natural capital (e.g., trees, parks, lakes, lawns) best explain variation in social and economic capital, and (3) how future investments in urban nature can address past inequities and promote more just and equitable urban futures.

Research Methods. We use several approaches to understand the mechanistic relationship between historical and future investments in urban nature and the accumulation of household wealth across different demographic groups. First, we will use archival methods and review of public documents to understand how property valuation schemes, tax assessments, and expenditures in park improvements and programming have created wealth in the form of increased property values. We will collaborate with Mapping Prejudice (see letter of collaboration) to relate the distributions of racially restrictive covenants and urban nature benefits for Hennepin and Ramsey Counties, which include the urban core of the 7-

county MSP metropolitan area. Land values in Minneapolis have shifted over time from the 1950s, when the distribution of values was relatively even to 2018, when values are highly segregated into high and low value neighborhoods that map along racial and ethnic lines (Fig. 18). We will explore the role of investments in urban nature in explaining this divergence in values over the past six decades.

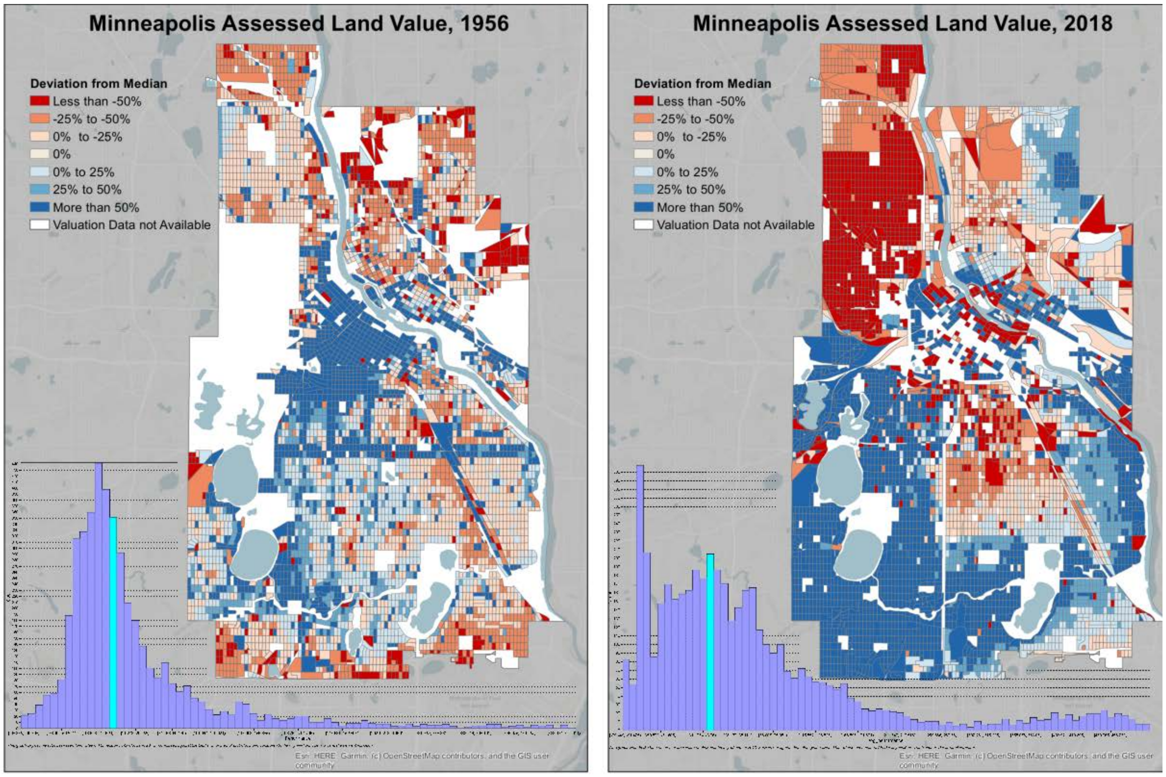


Figure 18. Change in the distribution of land values in Minneapolis from the 1956 to 2018. Total assessed land value of each city block was normalized against the total area of that block to create a standardized value/area metric. Value/area was mapped against the city median block value to determine the percent difference. The 1956 data come from the Minneapolis City Planning Commission and show total assessed land value and total assessed structural value per city block in Minneapolis. The 2018 data come from the Minneapolis city assessor's office and show parcel information (including land and building valuations). Histograms show the distribution of values in each year, with the highlighted bin containing the median value for that year. Comparison with maps of BIPOC communities (Fig. 9) shows decline in wealth in non-white neighborhoods and concentration in wealth in white neighborhoods.

Second, we will use results from **Q1-2** (e.g., canopy cover and diversity, bee diversity and abundance, lake water clarity) to assess the relationship between the quality and distribution of urban nature and property values. Although past studies in MSP (and elsewhere) have linked tree cover (Sander et al. 2010) and proximity to lakes (Sander and Polasky 2009) with increased property values, very few studies have investigated how the quality and spatial configuration of different types of urban nature affect property values *over time and space*. We will use spatially explicit regression analysis to explore how changes in nature types differentially affect property values, accounting for spatial lags and disconnects, and identifying where SETS (social, ecological, technical) factors moderate the relationship between urban nature and human wellbeing. For example, alum treatments in lakes may improve lake water quality even when watershed characteristics decline, or UHI effects may be moderated by the presence of air conditioning in households, which correlates with household wealth.

Finally, we will track investments in trees, parks, lakes, and lawns over time (integrating with **Q1-2**) and how they affect property values; we will also explore policy interventions (integrating with **Q3.2**) that can mitigate the negative effects of increasing property values, such as gentrification and displacement in low-wealth neighborhoods. Preliminary work in Minneapolis suggests that park investments in low-income neighborhoods are correlated with increasing property values in neighborhoods surrounding parks (Keeler et al. unpublished). We will track these relationships over time, creating a first-of-its-kind dataset on the long-term implications of investments in urban nature on property values.

Insights into policy interventions to prevent potential increases in property values are critical as city leaders invest millions in urban parks, green space, and green infrastructure without serious consideration of how these investments interact with existing landscapes of inequality in cities. Research over the past three years in the city of Minneapolis revealed that communities often perceived investments in park quality and access as increasing their risk of displacement and gentrification (Derickson et al. In review). As a result, some low-income residents advocate for solutions that are "just green enough" which may impede implementation of environmental initiatives and exacerbate inequalities (Dooling 2009, Checker 2011, Wolch et al. 2014). An enduring challenge is to improve environmental quality in communities without displacing the very residents these investments are designed to benefit. For example, related work under **Q1.3** will evaluate effectiveness of small and isolated patches, which would likely not contribute to gentrification, for providing pollinator benefits.

Expected Results. The proposed research will improve understanding of the links between housing policy, park financing, and urban nature investments and the accumulation of household wealth. Our work will assess these dynamics across contemporary and historical timeframes, leveraging archival datasets and digitized property values maps unique to MSP. This work will additionally contribute to identifying the contextual factors that affect the impact of alternative investments in different types of urban nature on property values. Finally, the work will contribute to developing a predictive model of green gentrification, building off existing econometric models of urban gentrification (Reades et al. 2018). Only with the systematic collection of long-term data on investments in urban nature and corresponding impacts on property values can we develop the spatial and temporal resolution to create new spatial models of gentrification. Piloted in MSP, this model could be tested in other geographies as we develop an improved understanding of how spatial context and factors that regulate supply and demand for property interact with investments in urban environmental quality.

Q3.2. How does governance of urban nature change over time, and how can it be changed to better address equity, human well-being, and improved environmental outcomes?

Hypothesis 3.2.1. Changes in governance, arising from changes in advocacy, will lead to changes in the provision of urban nature benefits and burdens.

Hypothesis 3.2.2. Changes in local-government policy related to aspects of urban nature that are highly technical (e.g., water quality) will be made by government officials with high levels of expertise, whereas policies related to less complex aspects of urban nature (e.g., different pollinator vegetation) will be made based on citizen input that draws on community values and shared expectations rather than technical knowledge.

Hypothesis 3.2.3. Across municipalities, policy actors who are more networked with actors operating at other levels of governance (e.g., between state and local governments) will be more likely to create changes in both policy and practice.

Hypothesis 3.2.4. Established interest groups are less influential in shaping the urban environment than highly engaged social movements with reflexive learning (critical examination of self in relation to others) and shared expectations (rules and norms).

Background and Past Results. Although there are many theories about changes in governance (Weible and Sabatier 2018), few studies systematically track policy change drivers across government units and across time. The few that have generally do not track relationships between governance and ecosystem processes (Scott 2015) or examine local governance. Further, they focus on testing single theories rather than comparing across them (Baumgartner et al. 2018), and on policy enactment, ignoring the widely recognized ways that implementation shapes policy (Sandfort and Moulton 2015). We will examine how social networks, shared expectations, and reflexive learning during the practice of actors working together contribute to change across diverse urban nature activities and communities.

Policies, rules, norms, and practices shape how people interact with and modify urban nature, and thus affect ecosystem processes and the distribution of urban nature benefits. We aim to understand how advocacy influences urban nature, through practices that shape governance via the following mechanisms: articulated expectations (policies, programs, social norms), social networks (structure and exchange; heterogeneity/homogeneity/centrality), reflexive learning (among actors about each other), strategic interactions (targeted actions, resources, information), and socioeconomic factors. We expect these constructs to positively influence practices, thereby transforming institutions such that stakeholders more effectively collaborate to manage urban nature in ways that promote desired social and environmental benefits.

Past research has shown that environmental policy advocacy is distinct from advocacy in many other policy arenas because of its technical complexity (Ganz and Soule 2019), with experts dominating technically complex policy arenas (Gormley 1986). While experts may dominate technically complex issues like water quality management, for other types of management, such as that of low-input turf and to some extent pollinator habitat, many municipal managers evaluate trade-offs, rather than relying on experts (Barnes et al. 2020b). Working within institutional action situations of power and influence, they implement new practices themselves (Ramer and Nelson 2020). Various studies indicate that linkages across scales (Cudney-Bueno and Basurto 2009, Mwangi and Wardell 2012) and between diverse networks (Granovetter 1973, Berardo 2014) facilitate the spread of information and practices in ways that can both advance and impede environmental governance. However, often in water governance there are spatial and temporal mismatches (York et al. 2019). In MSP, neighbors shared new yard practices to reduce fertilizer and water use (Martini et al. 2014), but discourses and yard stories did not include the yard's broader ecological linkage with neighborhoods and regions (Dahmus and Nelson 2014b, a), limiting practices that address larger environmental issues. However, resident attachment to neighbors and local water bodies through social ties and concerns about stormwater predicts if a resident is "likely to be civically engaged in water resource protection" (Pradhananga and Davenport 2017).

Recent research on political advocacy indicates that successful advocacy for policy change depends on high engagement between leaders and activists (Han 2009, 2014, Crutchfield 2018, McAlevey 2018). It is unclear if these findings apply to environmental problems, where technical experts have long played an important role (Ganz and Soule 2019), or to local politics that are not dominated by social movements, where decades of research point to the importance of established interest group coalitions (Domhoff 2005, Berry and Wilcox 2018, Holyoke 2018). Some argue that advocacy within place-based groups can use engagement, reflexive learning, and shared expectations (rules and norms) for change (Koontz et al. 2015, Crutchfield 2018). For residential yards in MSP, neighborhood social norms are the dominant enforcement mechanism for irrigation and vegetation lawn ordinances (Sisser et al. 2016), and homeowners do identify ecosystem services and disservices, opening the possibility for trade-offs to influence practices (Barnes et al. 2020a). Regarding pollinators, public park users in MSP are supportive of bee lawns for aesthetic and pollinator reasons (Ramer et al. 2019). Linking our place-based research on environmental policy change with broader debates over the drivers of successful policy change will help bring together research on social movements and the governance of ecosystems.

We aim to address key gaps in our understanding of urban nature governance and advocacy, addressing: (1) how long-term changes in governance related to urban nature, especially those arising from advocacy, alter urban nature benefits and harms, (2) whether there are differences in the policy actors who make local-governance policies related to urban nature, depending on the technical complexity of the environmental issue being addressed, (3) whether policy actors who are more networked with actors operating at other levels of governance are more likely to create changes in both policy and practice related to urban nature, and (4) how the influence of established interest groups compares to that of highly engaged social movements in shaping urban nature policy.

Research Methods. To address policy change and governance, we will establish a long-term quasi-experiment to analyze institutional change in the MSP urban ecosystem. We will select 30 municipalities using a stratified random sampling protocol from the MSP federally designated Metropolitan Planning Organization (MPO), which includes 110 municipalities (Engebretson et al. 2020) (note that where possible, studies conducted for **Q1-2** will be co-located in these municipalities). To measure governance, we will analyze policy documents (e.g., city codes, comprehensive plans, program documents) and interview municipal staff and officials, relevant watershed district staff, advocacy organizations (e.g., Friends of the Parks, Metro Blooms, Tree Trust), neighborhood organizations, homeowner associations, landscaping businesses, and community development organizations to understand practices surrounding policy change. We will link to data collected for **Q1.2, Q1.3, Q2.1, Q2.2** to understand better how governance has influenced urban nature and test H3.2.1.

Ordinance and municipal comprehensive plan change. We will evaluate changes in ordinances and comprehensive plans over time, for four-year periods (e.g., 2017, 2021, 2025), to further test H3.2.1 and determine: (1) which values ordinances and plans protect versus restrict, (2) the mechanisms of control and flexibility, (3) how these mechanisms change over time, and (4) how these policies influence human behavior, structural relationships, and urban nature over time. To identify landscape regulations for each municipality, we will collect city ordinances and regulations from legal publishers (Code Publishing, American Legal Publishing Corporation, Municode) and conduct content analysis (Krippendorff 2018). Possible search terms include landscaping, yards, gardens, trees, pollinators, water quality, and lawns. We will use automated change detection analysis to locate instances when city codes have changed since our prior review (Engebretson et al. 2020).

To understand dynamics at the actor level and test H3.2.2-H3.2.4, we will conduct systematic interviews with key stakeholders in each of the 30 cities to understand the drivers of policy change. We will identify stakeholders using a snowball sampling approach, beginning with key elected policymakers (e.g., mayor, city council members) and implementing officials (e.g., city forester) and environmental justice organizations likely to know policy advocates from marginalized communities. We aim to conduct 20-30 interviews in each municipality (600-900 total interviews) and expect this will take 2-3 years to complete. Interview topics will focus on low-input vegetation, bee lawns, trees, and water quality, to integrate with **Q1**, **Q2**, and **Q3.1**. These foci offer a contrast between an established policy area (e.g., cities have been managing urban forests for decades) and one in which policies tend to be new and uncertain (e.g., climate change adaptation). Interviews will focus on understanding three questions: How do individuals and organizations engage in the policy process to seek change or policy stability, i.e., what precisely did they do to influence the policy process, and why? How did they interact with other groups and individuals to make changes? How did these efforts influence the adoption of formal policies and the development of practice on the ground? Analysis of interview data will focus on developing a qualitative understanding of policy change and stability across municipalities and organizations. We will code these data in NVivo (Saldaña 2015) to develop case studies of the process of policy change and implementation and compare these processes across municipalities, allowing the development of causal inference (George and Bennett 2005, Beach and Pedersen 2013).

Expected Results. These analyses will develop and test theories about the relationships between social transformation and urban nature, drawing on a larger number of cases and a longer time frame than has been possible in past analyses. We examine policy experimentation (Huitema et al. 2018) based on the practice of actors and advocacy for policy change across ecological outcomes (**Q1** and **Q2**) and equity (**Q3.1**). Our work will identify the emergent factors that influence policy advocacy and practice related to urban nature. Insights will inform ecological and social theory while also being embedded in practice through reflection with practitioners and communities (see **Q4**).

Q4. How can long-term social-ecological research engage inclusively with diverse urban communities, particularly Black, Indigenous, and People of Color, for equitable and meaningful scientific and community outcomes?

Hypothesis 4.1. Through inclusive, community-engaged research, and continuous evaluation and adaptation, we can advance academic-community partnerships to produce meaningful and equitable scientific and community outcomes (Fig. 19).

Background and Past Results. Many researchers on the MSP Urban LTER team have built relationships with community members, local governments, community-based organizations, and local businesses in ecological and socio-ecological research. A review of the current research programs of MSP Senior Personnel reveals community partnerships with varying structures and at varied stages of development. We have come to realize over decades of community engagement in MSP and elsewhere that community engagement in teaching and research affects not only community partners, but also the teachers, researchers, and the research itself in immutable ways (Pradhananga and Davenport 2017, Pradhananga et al. 2019). Our working conceptual model for Inclusive, Community-Engaged Research (ICER, Fig. 19) builds on the latest theory, practice, and evaluation in the Community-Based Participatory Research (CBPR) field (Ortiz et al. 2020). However, the conceptual model we propose to employ here is innovative in its community-driven approach to model development, application, evaluation, and adaptation.

Our track record indicates success in engaging and sustaining community partners. However, it is clear that many of our existing partners look and think as we do (i.e., are mostly white-dominant), and like us, already have power and privilege in the current systems of ecological science and management. Thus, the knowledge and values that these partners add to the work largely remain centered on white perspectives and colonial practices. By listening to BIPOC communities, we propose to de-center whiteness and de-colonize science in our work to acknowledge and address systemic racism and legacies of discriminatory practices in urban ecosystem science, policy, and management. For example, building strong long-term partnerships with American Indian communities, for whom MSP has deep historical and sacred meaning, is critical for the MSP Urban LTER. Successful partnerships with American Indians recognize Traditional Ecological Knowledge to facilitate cross-cultural collaboration (Whyte 2013) and respect deep ties to place and human relationships to transcend place and connect the broader community (e.g., city to reservation). Other MSP communities from diverse ethnic backgrounds, including Black and Latinx communities, and recent immigrants from Southeast Asia and North Africa, are building ties to place and have created or are creating their narratives of urban nature, which are often overshadowed by the dominant white, wealthy, settler-colonist narratives about social-ecological relationships (Pradhananga et al. 2019).

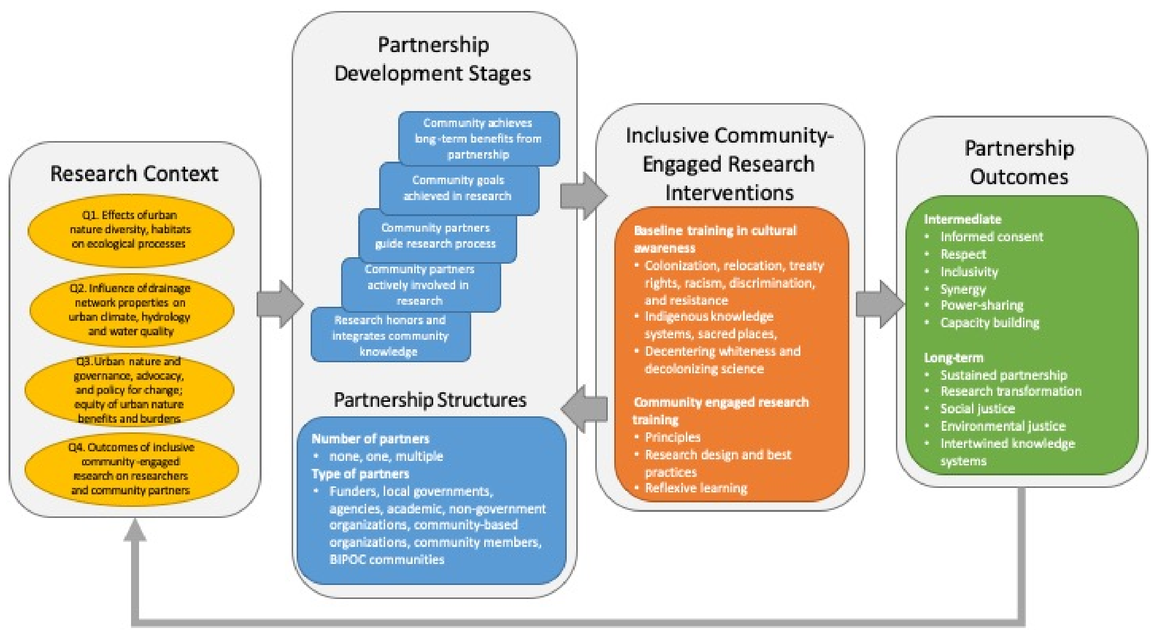


Figure 19. Working Inclusive Community-Engaged Research (ICER) model.

We aim to address significant knowledge gaps related to Inclusive, Community-Engaged Research: (1) how starting with cultural awareness and community engagement training and reflecting with partners about diverse knowledge systems changes the research practice and the researchers themselves over the long term, and (2) whether such research creates more equitable long-term urban nature research outcomes.

Research Methods.

Reflect on past partnerships and develop a new ICER model for long-term ecological research in collaboration with diverse community members. We will conduct an intensive literature review on CBPR with an emphasis on BIPOC scholarly work to assemble the foundation of an ICER conceptual model and typology. We will then examine past and current partnerships between project researchers (and other related researchers) and community partners in MSP. We will gather qualitative and quantitative data through multiple evaluation methods, such as one-on-one interviews, focus groups, and surveys with project personnel (researchers, staff, and students) and community partners. We will reflect on past partnership structures, processes, activities, and outcomes, noting any lack of inclusivity, procedural deficiencies, and harmful outcomes. Guided by this reflexive work, the research team and our community consultant organization Water Bar-Raft (Shanai Matteson, Director; see Facilities, Equipment, and Other Resources) will identify aspirational community partners, especially within historically underserved BIPOC communities, and serve as a bridge between the academic research and MSP communities. Together with Water Bar staff, we will contact "prospective" future partners to discuss opportunities for community-engaged research. Researchers and Water Bar staff will listen to prospective partners' needs, concerns, and expectations for responsible and respectful research partnerships. Discussions will address how race, ethnicity, class, and culture matter in this work. Funds have been budgeted to reimburse community participants for travel and time as research advisors. As community-engaged research initiatives emerge, transparent community and academic goals and critical outcomes will be established. When appropriate, partnerships will be formalized through written agreements.

Apply and Evaluate ICER Model. All project personnel will complete cultural awareness training with support from UMN's Office of Public Engagement, Office of Diversity and Equity, and Institute on the Environment, and Water Bar-Raft. As community-engaged partnerships emerge or existing partnerships further develop, project participants will engage in progressively specialized ICER training with learning outcomes around ICER principles, research design strategies, best engagement practices, and reflexive learning. We will tailor training interventions to suit the development stage of the partnerships (Fig. 19) and, when possible, include community partners, researchers, and students from across UMN. We will assess interactions between community partners and researchers, using multimodal (e.g., small group facilitated discussions, online surveys) and multidirectional evaluation, where researchers and partners reflect on what is and is not working, and how it has changed their approach to their work.

Specifically, together with our consultant community organization Water Bar-Raft, we aim to grow long-term partnerships with American Indian communities and other BIPOC communities to engage in social-ecological research, towards a scholarship that is more inclusive with the communities in which it is conducted. Using culturally relevant methods (Day et al. 1998), our work will include community forums, key informant interviews, focus groups, and development of ecocultural calendars, based not on Gregorian time but on ecological and cultural events in the yearly seasonal cycle of community partners. Consultation will be centered in community institutions and events to build relationships and determine if there is potential for problem-solving collaboration followed by the co-development of research. As one approach and when appropriate, we will use the Urban InVEST suite of decision-making modeling tools developed by the Natural Capital Project (Lonsdorf et al. 2011, Sharp et al. 2020) to help visualize how changes in land use and management affect the distribution of nature benefits and burdens throughout MSP. We will design a survey method cooperatively with community members, and implement it at community centers and events to more inclusively represent all segments of the MSP community. Survey questions will assess environmental beliefs, norms, and behaviors (linking with **Q1-3**). Building diverse narratives of urban nature will improve understanding of urban nature and human well-being and contribute to creating more equitable and sustainable cities and institutions.

Expected Results. We expect our research practice will become more inclusive and equitable as we reflect with partners on our past academic-community relationships, identify community goals, undertake cultural awareness and tailored community-engaged research training for project personnel, and conduct continuous formative evaluation. The MSP LTER affords the opportunity to carefully rethink which communities we, as researchers, engage, and how we do so. We aim to reach across false ecological-social system boundaries inherent in disciplinary science and the academy to work at the intersection of environmental and social justice. This time of great social upheaval and powerful anti-racism movements underscores the critical challenge facing any long-term ecological research program. We have the opportunity to reflect on our past, current, and aspirational community-academy relationships and to recommit to more inclusive and meaningfully engaged community partnerships in research. We will benefit from six years or more to modify, re-envision, or start anew with relationships in BIPOC communities and to collaboratively evaluate community and academic outcomes. Community benefits from this work are essential, and we anticipate that the relationships and partnerships supported through the LTER will translate into shared knowledge, resources, and practices for addressing environmental and social justice issues and public health needs. We anticipate that our ICER model can be adapted and applied beyond MSP to promote more inclusive long-term social-ecological research elsewhere.

V. SYNTHESIS

We will integrate research across project components through (1) shared focus on three aspects of urban nature – insect pollinators, trees, and water – to (a) address basic and applied long-term ecological questions (and in some cases evolutionary questions), (b) understand long-term change in governance of urban nature, and (c) explore social disparities in long-term relationships between urban nature and urban residents; (2) use of empirical studies and models to address social-ecological questions at multiple spatial and temporal scales, including use of models to explore scenarios of change in environmental conditions and human actions; and (3) consideration of social inequalities related to urban nature benefits and burdens across all projects.

For example, work on insect pollinators will explore effects of toxins and habitat characteristics on insect populations and pollination benefits at scales ranging from organisms to species, populations, communities, and habitat patches in fragmented landscapes across socio-demographic gradients, using experiments at nested scales to parameterize landscape-scale models. Research on trees will focus on the patterns and functioning (productivity, resilience, resistance) of diversity from habitat patch to landscape scales, and use empirical results from tree inventories and other ground-based measures to parameterize statistical models and enable remote sensing of diversity and its functioning at landscape scales. These models will facilitate comparison of canopy and diversity patterns and dynamics with those of historical racially discriminatory housing policies and of governance and policy change going forward. Water research will focus on scales ranging from habitat patches to drainage networks to landscapes, and ultimately link to lake water quality because of its importance to urban residents. We will use models to explore consequences of different drainage network configurations, management, and climate change for UHI, ecological, and hydrological processes in watersheds, and ecological and physical processes in lakes. As much as possible, different research components will be co-located to facilitate linking social and ecological questions.

Our work will be further integrated by shared consideration of social disparities related to the relationships between urban nature and urban residents, including LTER researchers. We will examine social disparities both as drivers of ecological patterns and processes and with respect to their role in social-ecological feedbacks. Specifically, we will address how benefits and burdens of urban nature are distributed between white and BIPOC communities, and how change in urban nature and related practices, policy, and governance might address (or reinforce) inequities moving forward. Furthermore, we will explore how adaptive, inclusive, community-engaged research advances academic-community partnerships to change scientific and community outcomes over the long term.

VI. EDUCATION AND OUTREACH ACTIVITIES

Schoolyard LTER. MSP will develop its Schoolyard LTER in partnership with UMN's Bell Museum to support adoption of Minnesota's new science standards, focusing on two activities: (1) school programs that serve seventh-grade learners in the St. Paul and Minneapolis public schools with urban ecology experiences, via field trips on-site at the Bell and/or delivered virtually, and (2) support for the professional development of middle school teachers through workshops and toolkits that help them engage with LTER science and scientists and develop confidence to implement lessons related to natural phenomena, practices of science, and urban nature in their schoolyards and neighborhoods.

The Bell Museum will kick off development of its Schoolyard LTER programs by assembling a *Teacher Advisory Committee*, comprising educators interested in environmental/ecosystem science and representing schools that serve diverse populations. Our Schoolyard LTER efforts will target MSP's most urban school districts, Minneapolis and St. Paul (65 and 79% non-white; 57 and 66% students receiving free/reduced lunch, respectively). In consultation with the *Teacher Advisory Committee* in Years 1-2, Bell staff and LTER researchers (representing the natural and social sciences) will co-design and pilot urban ecology school programs that address LTER-specific research related to urban pollinators, urban trees, water quality, and ecosystem services, in a way that addresses underlying racial and social inequalities. Schoolyard LTER activities will help teachers meet Minnesota's new Earth and life science standards (currently in the rule-making phase, informed by the Next Generation Science Standards and the National Research Council 2012). They will leverage the Bell's resources including world-class exhibits about Minnesota biomes; a planetarium that offers immersive content about water and ecosystem science; a learning landscape that features extensive pollinator habitat; a stormwater pond and rain gardens; and an institutional commitment to racial and social justice, particularly evident through recent exhibits and programs that center the voices of Indigenous people and highlight environmental injustices. We will pilot and assess field trips as both on-site experiences at the Bell and as virtual experiences via distance learning, a format becoming increasingly necessary in response to COVID-19. Programs will be led by Bell Museum educators and feature hands-on experiences and direct connection to LTER research and researchers (with particular focus on graduate students, early career scientists, and those identifying as BIPOC). We anticipate scaling and growing the on-site/virtual programs by Year 3.

Annually (starting in Year 2), the Bell Museum will host middle-school teachers for a daylong professional development workshop with Bell education staff, LTER researchers and collaborators (including community partners), and education professionals. The agenda, co-created with the *Teacher Advisory Committee*, will aim to foster the development of an education community of practice for the MSP LTER and will include presentations from researchers, networking and resource sharing by participating teachers, and opportunities to deepen understanding about phenomena-based instruction, three-dimensional learning, and best practices for exploring and investigating local urban nature with learners in ways that address science standards in socially and culturally relevant ways. We will share materials, resources, and lesson plans that emerge from these annual workshops through the larger LTER Network and its Education Digital Library.

REU Program. MSP will fund two REU positions each summer. REU mentors will rotate among LTER researchers and be selected by the LTER Advisory Committee based on proposals describing potential projects. REU students will be recruited through and integrated into a consortium of UMN-based REU students centered around the NSF-funded REU Site on *Sustainable Land and Water Resources*, directed by Senior Personnel Dalbotten. Students recruited through this consortium are 80% minority and 40% Native. The Consortium provides a support network of organized activities and professional development.

Integration with Undergraduate Curricula. Integration of research activities with undergraduate curricula will be both *broad* and *deep*. At UMN, students will have the opportunity for *deep engagement* with LTER research through two capstone experiences. Students in the Urban Studies major will have the option to work with LTER researchers to design social science research projects to support LTER activities in fulfilment of their capstone requirement. Students in the Environmental Sciences, Policy, and Management major will collaborate with LTER researchers and policy and government stakeholders to conduct original research related to the policy and environmental implications of LTER findings. The close collaboration between LTER researchers in the classroom, supervising undergraduate research and engaging stakeholders, will provide learning opportunities for undergraduates and also create new pathways of data collection and dissemination of findings. Undergraduate researchers will share collected data with LTER researchers and share their findings with community and governmental collaborators. At both UMN and the University of St. Thomas, there will be *broad engagement* across a range of ecology, engineering, environmental science, and other courses. Instructors will work with project investigators to integrate research questions and sites into the curriculum of their courses, an approach that aligns with UMN's and St. Thomas's emphasis on undergraduate research.

Community Partnerships. MSP LTER will foster academic-community partnerships through the activities described under **Q4**. We will support meaningful community-centered dialogue about the Inclusive Community-Engaged Research (ICER) model, LTER research, and community engagement. Our community organizing partner Water Bar-Raft will organize and facilitate sharing and listening sessions through their physical and online community networks. The nature of these partnerships and how we anticipate that they will evolve over time is further described under Q4.

VII. INTELLECTUAL MERIT

The proposed research will achieve **intellectual merit** by illuminating the dynamic and diverse relationships between urban nature and people to improve understanding of social-ecological responses of the urban ecosystem in the face of changes (climatic and social) that are as rapid as any in recent history. By advancing understanding of how pollutants, biodiversity, land cover, habitat fragmentation, and drainage network properties affect urban climate, ecological, and hydrological processes in the face of rapid climate and other changes, we will test whether ecological theories developed in non-urban ecosystems can predict patterns and processes in highly modified and managed urban systems. We aim to shed light on patterns of social disparities – and their underlying mechanisms – in human relationships with urban nature and how such disparities can be addressed through institutional and policy change and greater inclusivity in long-term ecological and social research to improve environmental outcomes for all residents.

VIII. BROADER IMPACTS

MSP LTER will achieve broader impacts by promoting the **full participation of women, persons with disabilities and underrepresented minorities in STEM** through its assembly of a leadership team (five PI/co-PIs) that includes four women and one person of color, two of whom are junior faculty; and a team of Senior Personnel that is 55% women. We will actively recruit undergraduate and graduate students and postdocs from underrepresented groups in STEM through UMN and University of St. Thomas programs described in the Facilities, Equipment, and Other Resources. LTER researchers will **improve STEM education and educator development** through **Schoolyard LTER** activities in partnership with UMN's Bell Museum, aimed at middle school learners and educators, partnerships with undergraduate programs and curricula, and mentorship of graduate students and postdocs. By nurturing numerous existing and cultivating new academic-community partnerships, we aim to **increase public engagement with science and technology** and **increase partnerships between academia and urban nature managers and MSP community members**. Many of these partners are identified in the letters of collaboration and in the Facilities, Equipment and Other Resources document, and we anticipate that new partnerships, especially with BIPOC communities, will form through our efforts related to Q4. We will especially focus on developing new and meaningful partnerships with BIPOC communities. LTER researchers will **improve the well-being of individuals in society** by conducting inclusive, participatory research to understand mechanisms underlying socioeconomic disparities in the distribution of urban nature burdens and benefits related to soil toxins, pollinator diversity and services, tree canopy resilience and resistance, water quality, UHI effects, and others in the context of dynamic human actions and climate. LTER research will directly inform how management, governance, and policy can be improved to better the health and well-being of urban residents. We will promote the **development of a diverse, globally competitive STEM workforce** by funding two **REU** positions each summer and by hiring numerous other undergraduate summer research interns, graduate students, and postdocs to participate in LTER research, education, and public engagement activities. Finally, we will **enhance infrastructure for research and education** in multiple ways: through the creation of curricula and toolkits for middle school educators to teach science standards using outdoor activities in their local schoolyards and neighborhoods; through the development of models of pollinator dynamics, remotely sensed tree diversity, urban drainage networks, and urban lake dynamics; and through creation of a public data portal to synthesize, make accessible, and visualize societally relevant environmental and ecological data towards improving human health and urban nature benefits.

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