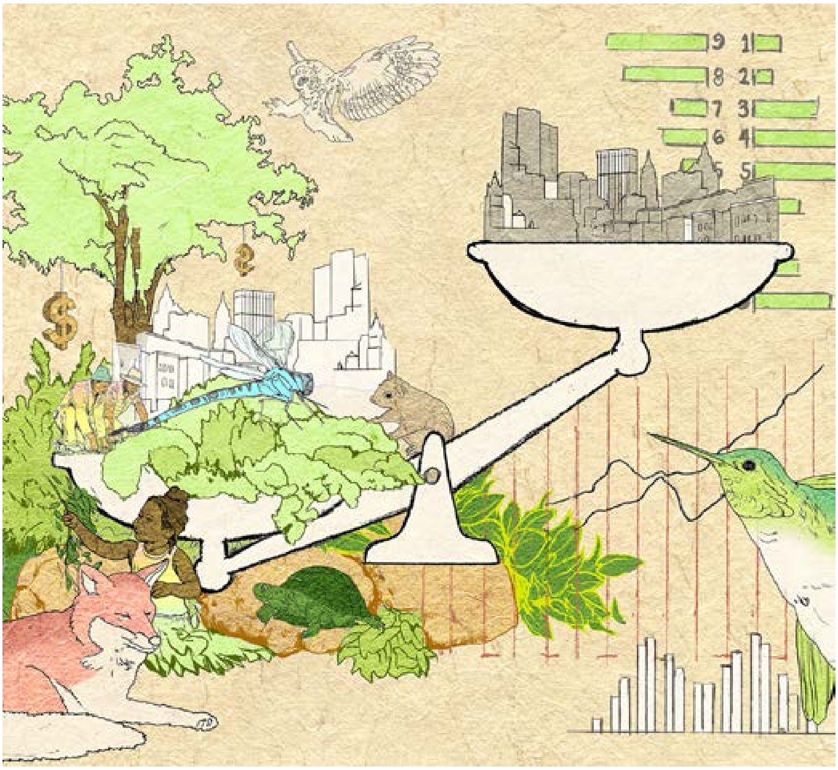

12. Mapping, measuring, and valuing the benefits of nature-based solutions in cities

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INTRODUCTION

Cities must manage multiple, interacting challenges simultaneously—climate change, air and water pollution, flooding, heat waves, affordable housing, public health, and socio-economic and environmental inequities, just to name a few. Solutions to urban problems will necessarily come from a range of interventions that include gray or built infrastructure, green or natural infrastructure, and combinations of the two. Urban development that defaults to gray infrastructure risks inefficient use of resources and lost opportunities for synergies. Nature-based solutions can help cities address many of the challenges they face, breaking down artificial conceptual and policy barriers between urban problems by offering solutions to one problem that can concurrently deliver multiple co-benefits via the services that nature can provide people.

Nature-based solutions can provide a broad range of benefits to people in cities, i.e., ecosystem services—sometimes referred to as nature's contributions to people (Díaz et al., 2018). For example, they can help reduce the risk of flooding, attenuate water, noise, and air pollution, mitigate the urban heat island effect, and provide attractive spaces that promote physical and mental health (Depietri & McPhearson, 2017; Haase et al., 2014; Keeler et al., 2019; van den Bosch & Ode Sang, 2017). Information about how much, where, and for whom investments in natural infrastructure yield benefits can improve urban planning and decision-making and direct limited budgets to projects most likely to provide critical benefits to people (Cortinovis & Geneletti, 2020;

Hamel et al., 2021; Keeler et al., 2019; Lafortezza et al., 2018). Ultimately, understanding the link between urban nature and human wellbeing can guide the design and redesign of more sustainable, livable, equitable cities. In theory, the approach to evaluating the ecosystem services that nature-based solutions can provide should not differ in rural versus urban landscapes. The "ecosystem service cascade" is a conceptual framework that maps the flow of services from ecosystems to people (Haines-Young & Potschin-Young, 2010; Tallis et al., 2012). It integrates two key components: a biophysical model that describes how a landscape or seascape supplies a specific ecosystem service and a valuation function that translates how the service contributes to human wellbeing. This integration allows decision-makers and stakeholders to evaluate how potential changes in land cover (such as a nature-based solution) affect the amount of the service being provided. Areas of greatest importance for nature-based solutions will be places that have (1) a high density of people using services combined with (2) a supply of ecosystem services that are sensitive to changes in land cover. In practice, however, fine-scale biophysical and socio-economic heterogeneity in urban landscapes make mapping and assessing the equitable distribution of services under alternate scenarios a more challenging endeavor than it is in more expansive, "simpler" rural landscapes (Li et al., 2020; Liu et al., 2017; Lonsdorf et al., 2021; Steele & Wolz, 2019).

Here, we introduce key aspects of assessing nature-based solutions in cities—understanding the supply of services, quantifying their value (i.e., how they impact human wellbeing), and exploring how value depends on context. We then use two case studies to put these concepts into practice. The first highlights approaches and tools for mapping and quantifying multiple benefits of urban green infrastructure, and the second focuses on how benefits flow to different beneficiaries. We conclude with future directions exploring how more information about the values of urban nature-based solutions can lead to better decisions for people and nature in cities.

ASSESSING NATURE-BASED SOLUTIONS: FROM SUPPLY TO VALUE

Ecosystem services provided by nature-based solutions include provision of material goods (e.g., food, feed, materials), regulation of ecological processes that provide benefits to people (e.g., regulation of hazards, climate, air quality, and water quality, provision of pollination, and pest control) and non-material (intangible) services (e.g., improvements in physical and mental health, opportunities for recreation, bolstering belonging and sense of place). See Table 12.1 for some commonly assessed urban ecosystem services.

Table 12.1 Common urban ecosystem services and their supply of benefits to people living in cities, along with examples of potential metrics to value those services and methods for quantifying those values

Ecosystem service	Supply metric	Value metric(s)	Valuation modeling approach
Climate change mitigation*	Carbon stored or sequestered	Social cost of carbon	Net present value of change in damages from carbon emissions
		Carbon market price	Change in total revenue from sale of carbon credits
Urban cooling*	Air temperature	Productivity	Loss of workplace productivity as a result of temperature and humidity
		Climate emissions	Increased emissions from cooling (and heating) Cost of carbon (e.g., social cost, market price)
		Private cost of cooling	Cost of cooling (and heating) as a function of temperature
		Mortality or morbidity risk	Relative risk of mortality or morbidity as a function of temperature and region
Stormwater retention*	Stormwater volume and mass of pollutants retained by the landscape	Avoided water pollution	Cost of management practices to remove pollutants to meet water quality standards or regulations
		Groundwater replenishment	Cost of groundwater for irrigation and drinking water
Flood mitigation (coastal and pluvial*)	Flood volume or inundation extent, depth, or duration in extreme storm events	Avoided flood damage	Economic damage of crops and of buildings and other infrastructure estimated via repair cost
		Averting behavior cost	Change in costs associated with changing flood risk at the household or community level
		Mortality/injury	Risk of death/injury; number of people affected; value of a statistical life
Coastal hazard mitigation*	Ranked vulnerability of coastline to erosion and flooding	Role of coastal habitats in reducing vulnerability	Compare the number of people, demographics of people, value of property, type of infrastructure, etc. at increased risk in scenarios with and without habitats
Recreation*	Access (distance to parks), park attributes	Number of visitors to parks	Entry or use fees; willingness to pay; travel cost
		Housing prices	Hedonic pricing

Ecosystem service	Supply metric	Value metric(s)	Valuation modeling approach
Physical activity and health	Access to urban nature (e.g., distance to parks, tree-lined streets, urban gardens, trails, etc.)	Physical activity (e.g., metabolic equivalents)	Relative risk of mortality or morbidity
		Quality of life/years of life	Disability-adjusted life years or quality-adjusted life years
		Avoided cost of treatment	Change in costs associated with treatment to restore original physical health level
Mental health	Access to urban nature (e.g., views of greenery, distance to parks, amount of trees in neighborhood)	Mental health indices (e.g., GHQ12[a], MHI5[b])	Relative risk of mental illness; change in subjective wellbeing
		Avoided cost of treatment	Change in costs associated with treatment to restore original mental health level
Biodiversity	Ensuring continued presence of a species through protection effort	Option value	Bioeconomic modeling of net social welfare as an option value

Notes: * Addressed in the InVEST (Integrated Valuation of Ecosystem Services and Tradeoffs) software suite (Hamel et al., 2021; Natural Capital Project, 2023). Not all models provide the full suite of valuation modeling approaches. While it is not an ecosystem service, we include "biodiversity" in the table because many practitioners are interested in exploring biodiversity alongside ecosystem services to inform decisions.
[a] General Health Questionnaire (12 items).
[b] Mental Health Inventory-5.

On Supply: Translating Landscapes into Ecosystem Services

Biophysical models are used to estimate the supply of ecosystem services arising from the landscape. A biophysical model shows the full potential of ecological functions to provide a given ecosystem service, regardless of whether humans recognize or value that function or service (Tallis et al., 2012). Most of the biophysical models underlying ecosystem services translate land use and land cover inputs into the supply of a service using two steps: the site-specific production of a resource and a spatial process that illustrates how that resource flows through a landscape. Realized ecosystem services flow to people when there is supply and demand for the service generated by the interaction of people with the ecological system (Brauman et al., 2020; Burkhard et al., 2014; Cortinovis & Geneletti, 2019).

On Value: Translating Ecosystem Services to Impact on Human Wellbeing

Value is a complicated word, taking up over a foot of tiny text in its entry in the *Oxford English Dictionary*. Pascual and colleagues (2017) lay out four key meanings important in the context of valuing nature's contributions to people: value is "a *principle* associated with a given worldview or cultural context, a *preference* someone has for a particular state of the world, the *importance* of something for itself or for others, or simply a *measure*." The authors go on to describe the ways in which these meanings are linked, "for example when ethical *principles* lead one to assign *importance* to different aspects of nature's contributions to people, and to have a *preference* for a specific course of action, which in turn can be *measured* by an appropriate valuation tool" (Pascual et al., 2017, p.9, italics in the original).

Realized ecosystem services provide benefits to people; assessing (or measuring) their value is one way to describe the magnitude of their contribution to human wellbeing. There are a number of ways to do so. Economic methods can be used to generate estimates of benefits in monetary metrics, while other methods report estimates of benefits in non-monetary metrics (e.g., impacts on health, livelihoods, or environmental improvements). In sum, a person's or community's values (principles) lead to different assignments of values (importance and preference) and can be valued (measured) in different ways.

In the simplest framing, there are two broad categories used to discuss the values of nature: intrinsic and extrinsic. Intrinsic value is the value that something has *in itself* or *for its own sake* (Zimmerman & Bradley, 2019). Thus, the intrinsic value of nature includes the ways in which it has value irrespective of any relationship to humans. Extrinsic value (also called instrumental value) is the value that something has for the sake of something else to which it is related in some way (Zimmerman & Bradley, 2019) and includes the multiple ways in which nature provides goods and services to people. While Pascual et al. (2017, 2021) argue for "value pluralism" to better incorporate multiple worldviews and values in the identification and implementation of policy, in many cases it is important to recognize the broader context of value pluralism while focusing on individual types of value. Therefore, without denying the importance of intrinsic value, the concepts of ecosystem services and nature's contributions to people focus on extrinsic values of nature in its contribution to human wellbeing (Díaz et al., 2018; Guerry et al., 2015).

In some cases and contexts, monetary valuation of ecosystem services is useful. Monetary values allow aggregation of values into a common metric that can enable comparison of the value of ecosystem services to other goods and services, facilitating benefit–cost analysis of policies or management options. Expression in monetary value can also facilitate application and communication in policy, business, and financial sectors that traditionally use monetary measures as key metrics for decision-making. The provision of nature-based material goods sold in markets as commodities (e.g., agricultural crops, animal products, fish, timber) are relatively easy to value in monetary terms. Statistics on quantities and prices are routinely collected and readily available for many nature-based commodities.

Most ecosystem services, however, are more difficult to value monetarily. Most ecosystem services are not traded as commodities, have no observable price, and data on provision of the services may be sparse or missing altogether. Biophysical models can be used to generate estimates of the flows of ecosystem services and then combined with non-market valuation methods from economics to generate estimates of the monetary value of the flows of these services. Non-market valuation methods have been widely used to value environmental improvement (Freeman et al., 2014) and the provision of ecosystem services (Bateman et al., 2013; Committee of Experts on Environmental-Economic Accounting, 2021; National Research Council, 2005; Ouyang et al., 2020; Van der Ploeg et al., 2010).

There are three main types of non-market valuation applied to ecosystem services: (1) revealed preference methods; (2) stated preference methods; and (3) cost-based methods. Revealed preference methods use data on choices to infer values about non-marketed ecosystem services. For example, the premium price for houses on clean lakes or near nature preserves is evidence of the value that people have for nearby recreational opportunities and scenic beauty. Stated preference methods use survey responses to estimate the value that people hold for various ecosystem services. Cost-based methods use estimates of the costs of replacing ecosystem processes, such as the cost of providing clean drinking water with a water filtration plant instead of naturally provided clean water (Chichilnisky & Heal, 1998; National Research Council, 2000).

Where monetary valuation lacks robustness, feels wrong to key stakeholders, or is not relevant to decisions, it is often preferable to report outcomes of ecosystem service assessments in non-monetary terms. Non-monetary measures include biophysical metrics of environmental quality (e.g., whether lakes and rivers meet water quality standards), along with measures directly related to human wellbeing, such as measures of human health or livelihoods (Díaz et al., 2018; Keeler et al., 2012; Myers et al., 2013; Olander et al., 2018; Ruckelshaus et al., 2015). To be good measures of the value of ecosystem services, these measures should fairly directly connect nature to human wellbeing.

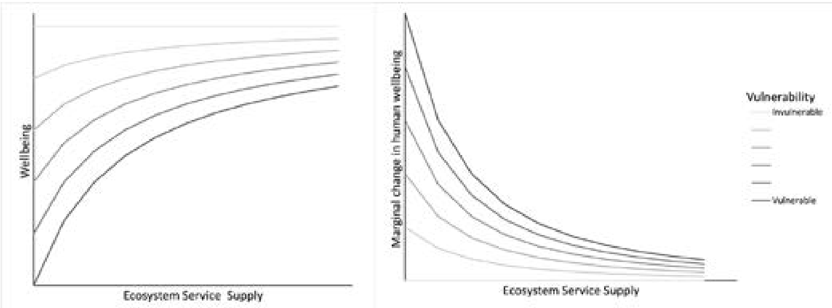
On Context: How Socio-Economic Factors Mediate the Importance of Ecosystem Services

Regardless of how value is measured—in monetary terms, as impacts on human health or livelihoods, or using other metrics—the measure of value will depend on both the ecological and socio-economic context (Nelson et al., 2009; Tallis and Polasky, 2009). For example, the same amount of physical cooling provided by urban vegetation to mitigate the urban heat island effect will be of much greater value to urban residents in regions where hot weather is common compared to those in regions where it is rare. The population density and the size of the "serviceshed," the area in which people benefit from a particular service (Mandle et al., 2015), will determine the number of people affected by a given change in the availability of a service.

Dependence on ecosystem services also varies widely. For example, low-income urban residents who lack air conditioning are more vulnerable to heat stress and more dependent on physical cooling provided by urban vegetation than are high-income residents with access to air conditioning. Careful attention to the vulnerabilities of beneficiaries allows not only for more accurate estimates of the values of services, but also for the addressing of inequities in the flows of services. Characteristics (e.g., age structure, race) and assets (e.g., income) can also affect the vulnerabilities of beneficiaries and vary across groups and with different contexts (see Figure 12.1). As the amount of ecosystem service provided by the urban landscape increases, human wellbeing increases for all people, regardless of vulnerability, but the most dependent would benefit the most and thus receive the greatest value from the increase (Figure 12.1, right). The least dependent on the service receive less value because their wellbeing is already high, regardless of the service supply.

CASE STUDIES

To demonstrate the utility and details of modeling, mapping, and valuing urban ecosystem services we explore case studies from two cities: Guangzhou, China, and Minneapolis, United States (US). These two cases illustrate key elements that make the evaluation of ecosystem services in urban areas uniquely challenging. In Guangzhou, we focus on the impact of fine-scale heterogeneity in land use on the flows of ecosystem services and their values. The Guangzhou case study exemplifies an approach to articulating the ecosystem services provided by a large green space in both biophysical and monetary terms. In Minneapolis, we explore whether the supply of ecosystem services is equitably distributed with respect to socio-economic factors. The Minneapolis case highlights the importance of exploring how different demographic groups benefit from urban ecosystem services, with particular attention on marginalized groups. Together they highlight the linked nature of biophysical processes



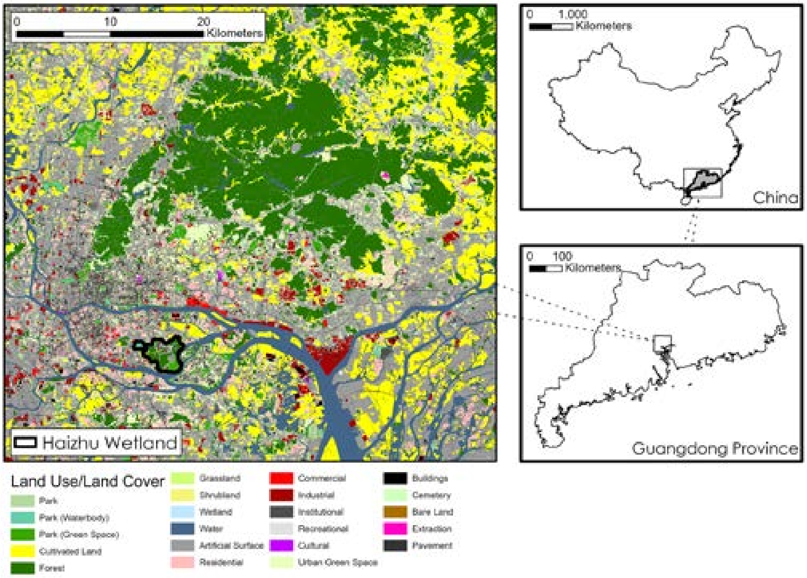
Notes: We expect that increasing supply of an ecosystem service improves human wellbeing but we also recognize that other social factors, e.g., education, access to health care, wealth, and technology or gray infrastructure, can all play mediating roles (left). For example, a neighborhood with high capacity and well-functioning and maintained stormwater infrastructure may be less vulnerable to a heavy rainfall event compared to a neighborhood with poor infrastructure and thus less dependent on the ecosystem service of stormwater retention. The importance of decisions that lead to changes in the supply of the service, i.e., the marginal value, thus depend on the interaction of vulnerability factors and the current supply (right). We expect diminishing returns on investments in ecosystem services for human wellbeing (regardless of vulnerability) as the current supply increases, and that the wellbeing of vulnerable groups continues to be enhanced more than that of less vulnerable groups.
Source: Authors' own.

Figure 12.1 Conceptual illustration of how social vulnerability may mediate the contribution of an ecosystem to human wellbeing

producing benefits, the social dimensions directing their distribution to people, and the data required to quantify them both.

Haizhu Wetland, Guangzhou, China

Guangzhou, China, is part of one of the world's largest metro areas, the Guangdong–Hong Kong–Macao Greater Bay Area, with a population of 72 million at the end of 2019. At 11km^2, the Haizhu wetland in Guangzhou is the largest wetland located in the downtown core of a Chinese megacity (Figure 12.2). Known locally as the "Green Heart" of the city, the wetland is highly accessible from the Central Business District and other densely populated areas, making it a key component of green space access for locals (Figure 12.3). From 2012 to 2020, the wetland received over 60 million visitors. It is also an important area for biodiversity in the city, home to a documented 177 bird and 325 insect species (compared to 72 bird and 66 insect species documented in adjacent urban areas). In 2020, the World Bank partnered with the local planning agency in Guangzhou and our team from the Natural Capital Project to quantify several ecosystem services provided by the wetland—in both biophysical and monetary terms—and to make those benefits explicit to decision-makers to help protect the wetland from future development. We modeled three services provided by the wetland: climate change mitigation (carbon storage and sequestration); urban cooling; and improvements in health (through both mental health and physical health pathways). We then calculated the provision of those same services in a future without the wetland to estimate marginal values.



Source: Land use/land cover is from GlobeLand30 and OpenStreetMaps data (see methods).

Figure 12.2 Location and land use/land cover of the Haizhu wetland region in Guangzhou, China

Land Use and Land Cover in Urban Environments

In urban areas, all ecosystem services are influenced by the interaction of land cover and land use. Land cover describes what the surface is, while land use provides information on how that cover is being used and thus how it might be managed. For example, turf grass is a common land cover type in urban areas. However, differences in management (e.g., mowing frequency and fertilizer application regimes) can vary dramatically with use (e.g., as residential lawn, recreation area, golf course, cemetery, etc.). These differences in land use within the same land cover type can affect biodiversity as well as services

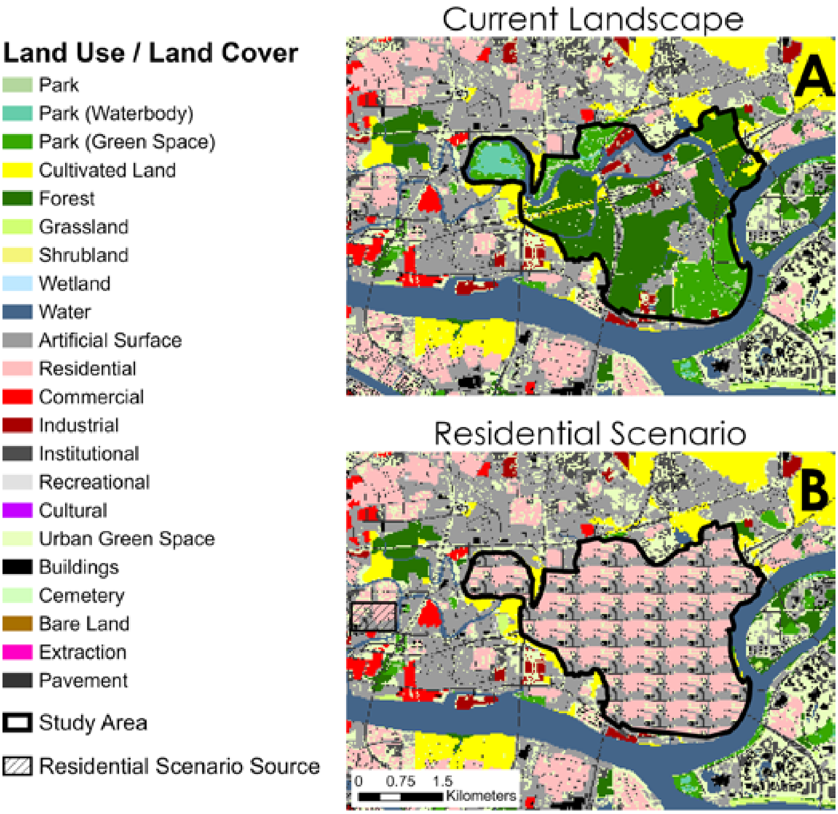
Source: Guangzhou Haizhu District Wetland Protection and Management Office (2020).

Figure 12.3 The Haizhu wetland and nearby Guangzhou Central Business District

such as nutrient runoff and retention, and carbon storage and sequestration. If one only used land cover, these differences would not be captured. Thus, it is critical that assessments of urban ecosystem services include both land cover and land use to accurately assess the impact of nature-based solutions.

Unfortunately, single land use and land cover (LULC) data sets that can account for this degree of heterogeneity are often unavailable, requiring creation by combining information from two or more data sets (Lonsdorf et al., 2021). For Guangzhou, we generated a new LULC dataset for the Haizhu wetland by combining land cover from GlobeLand30 (Chen et al., 2017) with land use from OpenStreetMaps (OpenStreetMap Contributors, 2021) and a Normalized Difference Vegetation Index (NDVI) dataset for 2019, derived from Copernicus Sentinel-2 using Google Earth Engine (World Bank, 2022).

Our partners from the planning department in Guangzhou were interested in comparing benefits provided by the wetland to a "no wetland" scenario—most likely one of residential development, given population growth. To create a residential scenario consistent with local patterns of land use and land cover, we applied the "wallpapering" method (Lonsdorf et al., 2021), which takes a small local LULC sample that best represents the future of interest and replicates it across the selected portion of the study area. We sampled an area



Source: Courtesy of Authors, based on data from World Bank, 2022.

Figure 12.4 *(A) Current land use/land cover in the Haizhu wetland in Guangzhou, China, and (B) alternative residential land use scenario*

of residential housing near the wetland to create a residential scenario (Figure 12.4B), which forms the basis for our marginal value calculations for each of the following ecosystem services. We repeated this process for NDVI using the same sample location as for LULC.

Climate Change Mitigation (Carbon Storage, Sequestration, and Avoided Emissions)

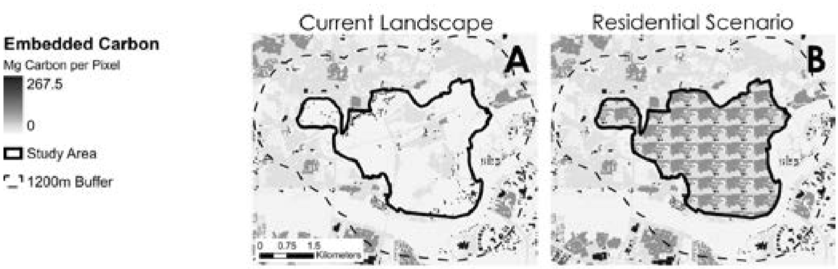
Overview

Climate change mitigation is an important goal for communities and decision-makers in urban areas. Two key mitigation pathways are the reduction of emissions and sequestering of carbon on the landscape through natural lands and green infrastructure. Traditional methods of estimating landscape carbon storage and sequestration often focus on land cover in mostly natural systems and center on four pools of carbon: above-ground biomass, below-ground biomass, soil carbon, and organic matter (Natural Capital Project, 2023). These pools have analogues in the built environment—soil carbon still persists underneath buildings and pavements (Edmondson et al., 2012), urban green spaces have abundant vegetative carbon stocks above and below ground, and we can even account for organic matter stored in the built environment (building materials, furniture, books, etc.) (Churkina et al., 2010). However, a full carbon accounting in urban areas must include emissions: *flux carbon*, or annual emissions from energy use and land management, and *embedded emissions*, the CO_2 generated during the manufacture and construction of built infrastructure (Kuittinen et al., 2016).

Methods

Supply

We reviewed the literature to estimate parameters needed to reclassify the LULC types into each carbon pool (Mg C/ha), flux (Mg C/ha/year), and embedded emissions (Mg C/ha) (World Bank, 2022). We used the parameter table detailed in World Bank (2022) to reclassify the LULC map into each of these categories of carbon (Figure 12.5).



Note: Similar spatial patterns exist for landscape carbon and annual emissions.
Source: Courtesy of Authors, based on data from World Bank, 2022.

Figure 12.5 Carbon in embedded emissions (from the manufacture and construction of built infrastructure) with (A) and without (B) the wetland

Value

We translated the carbon storage and sequestration results into monetary value using the social cost of carbon (Nordhaus, 2017). We report monetary value for the average value of the social cost of carbon with a 5 percent discount rate currently in use by the US government, as a conservative estimate of value (Interagency Working Group on Social Cost of Greenhouse Gasses, 2021).

Results

Replacing the wetland with the residential scenario adds 3,700 Mg (3.2 Mg/ha) of carbon to landscape pools, primarily from carbon stored in wood and other building materials. This is equivalent to US$52,500 (US$45 per hectare) in sequestration value. However, the residential development scenario generated significant embedded emissions from manufacturing concrete, steel, and other components of the built environment, increasing embedded emissions by 763,000 Mg (659 Mg/ha), at a societal cost of US$10.7 million (US$9,200 per hectare). Annual emissions similarly increased with transition to the residential scenario by 213 kMg CO_2-e/yr (184 Mg/ha/yr) at a societal cost of US$3.0 million per year (US$2,600 per hectare per year). Using a net present value approach with a discount rate of 5 percent and a 30-year time frame to estimate overall damages of annual emissions, we found the combined climate impacts from landscape carbon and annual and embedded emissions under the residential scenario amount to US$89.7 million of damages.

Urban Cooling

Overview

The urban heat island (Deilami et al., 2018; Oke, 1973; Rizwan et al., 2008) arises in cities due to a combination of heat capture and radiation by the built environment. Buildings and pavements capture solar radiation as excess heat, releasing that stored heat slowly and, if arranged in a dense enough urban fabric, raising the city's baseline ambient air temperature. This process can exacerbate extreme heat waves and increase the risk of mortality and morbidity among vulnerable populations, a pattern likely to worsen under human-induced climate change (Santamouris, 2020).

Methods

Supply

We used the InVEST Urban Cooling model to calculate the effect of the residential land use scenario on the local urban heat island (Natural Capital Project, 2023). In addition to LULC maps, this model requires data for reference evaporation (Trabucco & Zomer, 2019), reference air temperature for each month

(Kenji & Willmott, 2018), the maximum urban heat island magnitude (2.07°C, from https://yceo.users.earthengine.app/view/uhimap) (Chakraborty & Lee, 2019), the air temperature blending distance (600 m) (Lonsdorf et al., 2021; Oke, 2006; Schatz & Kucharik, 2014), and the maximum distance from which large contiguous green areas (> 2 ha) contribute additional cooling (100 m, as per InVEST recommendations). The model also relies on a parameter table linking each LULC category with five primary drivers of the urban heat island: shade, evapotranspiration potential, albedo, green area inclusion, and building intensity. To inform the selection of values, we reviewed the global literature on each of these parameters as they related to each of our LULC categories (see more detail in World Bank, 2022).

Value

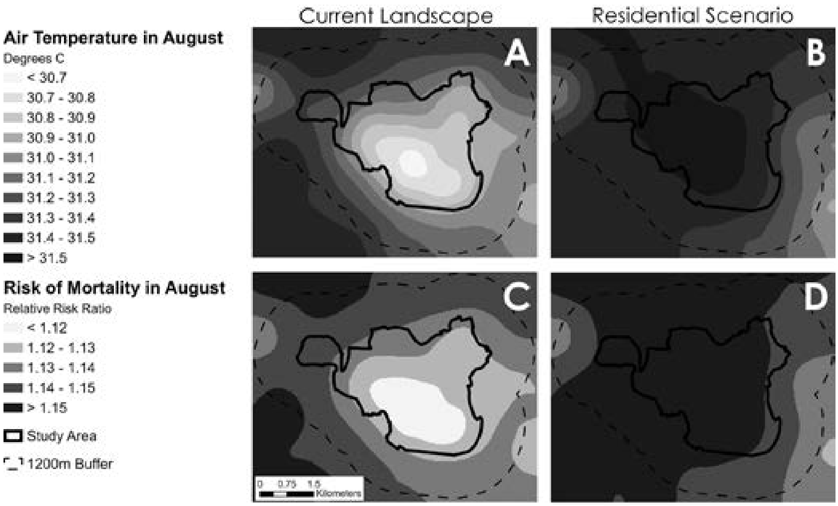
We assessed the monetary value of the wetland using a marginal value approach, calculating the projected loss of workplace productivity and increased energy cost of heating and cooling buildings in the surrounding areas. We used the Wet Bulb Global Index to inform changes in workplace productivity (Kjellstrom et al., 2009), using an average relative humidity of 71.1 percent (Ou et al., 2014). As data on workplace location and intensity of work (e.g., outdoor labor versus indoor office work) were unavailable for the study area, we assumed that commercial and institutional areas were "light work" and industrial areas were "heavy work" as per the InVEST guidelines (Natural Capital Project, 2023).

We translated the increased air temperature from the residential scenario into the increased energy consumption necessary to cool residential and commercial buildings using heating and cooling degree days (Roxon et al., 2020), assessing monetary value using the typical costs of energy per building type in Guangzhou (World Bank, 2022).

In addition to monetary value, we also estimate the avoided mortality provided by the wetland. Epidemiological literature reports that relative risk of heat-induced mortality increases above a "minimum-mortality" threshold temperature, which varies regionally due to the acclimatization of local populations but generally hovers around the 75th percentile of a region's temperature range (Guo et al., 2014). While this relationship between relative risk and temperature is non-linear, changes in risk are only reported at certain thresholds in the literature (i.e., 90th and 99th temperature percentiles) so we assumed a linear relationship to convert temperature maps into risk maps.

Results

Average August air temperatures surrounding the wetland vary between 30.7 and 31.5°C (Figure 12.6A). Under extensive residential development, the surrounding 600 m buffer would experience an average 0.25°C increase in air temperature on a typical day, a figure that increases to more than 1°C within the wetland area itself (Figure 12.6B). This represents the typical summer urban heat island effect—during an extreme heat wave, we can expect the loss of the wetland to further exacerbate temperature rise.



Note: Under the residential scenario, temperatures increased by an average of 0.25°C within the 600 m buffer surrounding the wetland (dotted line), corresponding with a 1.23 percentage increase in mortality risk.
Source: Courtesy of Authors, based on data from World Bank, 2022.

Figure 12.6 Modeled air temperature in August under (A) the current landscape and (B) residential scenario and the associated relative risks of mortality (C, D)

Without the wetland, workplace productivity in nearby heavy work environments fell during May and October (2.5 and 16.1 percent, respectively), while light work environments saw no change (World Bank, 2022).

The increase in air temperature under the residential scenario increased cooling energy demand during the summer months but decreased demand for heating energy during the winter months. As Guangzhou sits in a generally tropical climate with an average annual temperature of 22.4°C, the cooling demand outstrips heating demand over the course of a year: annual energy consumption by buildings within 600 m of the wetland increased by 1.1 million kWh at an annual cost of US$119,800. Using a net present value approach with a discount rate of 5 percent and a 30-year time frame, this represents US$1.9 million.

For Guangzhou, the 75th, 90th, and 99th temperature percentiles are 28, 30.1, and 32°C, respectively; the relative risk of mortality at each of those thresholds is 1, 1.08, and 1.18 (Guo et al., 2014). Linear interpolation between these points allows us to convert temperature to relative risk, which we use to compute the difference in relative risk of mortality between our two scenarios (Figure 12.6C, D). The surrounding 600 m buffer will experience between 1.23 and 1.27 percent increase in mortality risk each month between June and September, a pattern likely to worsen during extreme heat events (World Bank, 2022).

Improvements in Physical Health Through Increased Access to Urban Green Space and Resulting Increases in Physical Activity

Overview

Access to urban nature and green spaces can affect people's physical health by increasing physical activity, which leads to multiple positive health outcomes (Remme et al., 2021; Warburton, 2006). We adapted an approach from Vivid Economics (2017), where the value of green space for physical health is a function of the "catchment area" (or area of influence of a park on people's physical activity), the contribution of green space to physical activity provision, population, and the costs of physical inactivity.

Methods

Supply

Liu et al. (2020) determined that the catchment area was 2,230 m from a park boundary for Guangzhou. To incorporate the effect of other parks beyond the Haizhu wetland catchment on the population in the catchment, we doubled this buffer size and included any parks larger than 2 ha in that area. Given a lack of local data on the relative importance of green space to physical activity, we assumed that green space could contribute up to 11 percent of an individual's total physical activity, a conservative estimate from a study in Seattle, US (Stewart et al., 2018). We used a population dataset created by Liu et al. (2020) for central Guangzhou to scale the benefit of activity by each person.

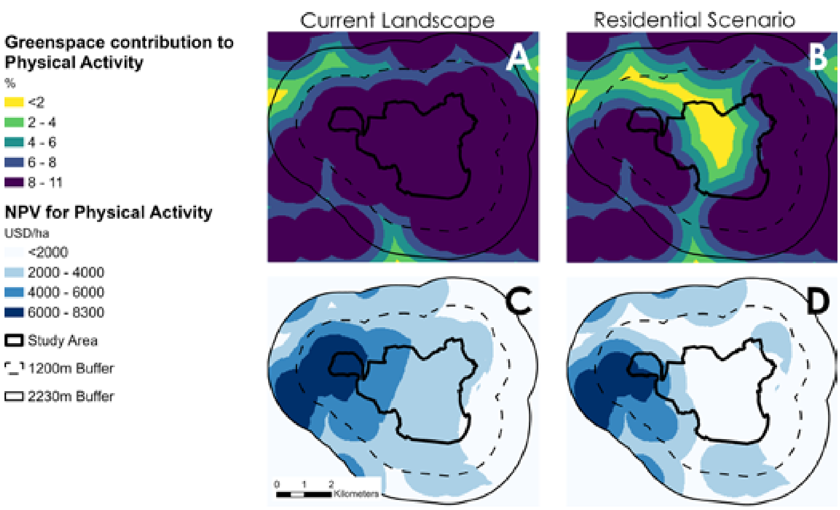
We assumed that the contribution of green space to total physical activity declines with increasing distance from green space throughout the catchment: for the first 300 m that contribution remains 11 percent (Labib et al., 2020), followed by a linear decay towards 0 percent at 2,230 m from a park.

Value

To estimate the avoided health-care costs due to physical activity in green spaces, we calculated the costs of physical inactivity, based on the valuation done by Zhang and Chaaban (2013) for China: US$44.2 billion in 2007. We extrapolated these costs to 2017 based on health expenditure figures for China (World Bank, 2021) and corrected those to 2020 US dollars: US$215.6 billion. With a population of 1.386 billion in 2017 this results in physical inactivity costs of US$23.58 per capita. We applied a net present value approach with a 30-year time period and a discount rate of 5 percent to calculate the difference in value between the current and residential scenarios.

Results

If the Haizhu wetland were developed into a residential area the average contribution of green space to physical activity would drop from 9.3 to 7.0 percent in the 2,230 m buffer zone (Figure 12.7), affecting about 230,000 people. The Haizhu wetland is of particular importance for the population living in and to the northwest of the wetland where there are few alternative sizable urban green spaces that could sufficiently support physical activity (Figure 12.7). In monetary terms, the development of the wetland could cause a loss of at least US$3.0 million in net present value over 30 years from declines in physical activity.



Note: In the 2,230 m buffer zone, the net present value loss is US$3.0 million under the residential scenario compared to the current situation.
Source: Courtesy of Authors, based on data from World Bank, 2022.

Figure 12.7 The percentage contribution of green space to physical activity for (A) the current landscape and (B) the residential scenario, and the corresponding net present value (NPV) of green space for physical activity in the Haizhu wetland and surroundings (C, D)

Improvements in Mental Health Through Access to Urban Green Space

Overview

Mental health has been linked to access to green space in urban areas (Bratman et al., 2019; Gascon et al., 2015; Houlden et al., 2018). Here we use a dose–response relationship where mental health outcomes and changes in expenditures on those outcomes at the population level are derived as a function of natural area within a given distance from urban populations.

Methods

Supply

We linked the WHO-5 index value, a commonly used survey-based index measuring psychological wellbeing (Topp et al., 2015), to natural areas in Guangzhou based on Liu et al. (2019). Liu et al. used multiple regression methods to relate WHO-5 scores in Guangzhou to a variety of neighborhood characteristics, including demographic variables and importantly an indicator of green space—the mean NDVI within a 1 km buffer around the neighborhood. The mean observed WHO-5 score (out of 25) was 12.081 in Guangzhou, and the WHO-5 goes up by one point for every 0.136 increase in mean neighborhood NDVI (baseline 0.097), all else being equal, i.e.

$$W = 12.081 + ((NDVI - 0.097)/0.136) \tag{12.1}$$

Unfortunately, we did not have the required data to apply this as a functional value transfer approach, where we adjust neighborhood estimates based on variation in other covariates besides NDVI, so we use this unit value approach. While unit value transfer approaches generally perform poorly compared to function transfer (Kaul et al., 2013), in this case the value estimates were derived in the Guangzhou case study area and are more likely to be representative of the population than if they were transferred from elsewhere.

Value

We linked changes in population-level expenditures on mental health in the Haizhu wetland area in Guangzhou to natural areas using the following equation adapted from Vivid Economics (2017):

$$\text{Change in expenditures}_i = \text{Pop}_i * \text{Exp}_i * \left((-1) * \left(W_i^R - W_i^C\right) / W_i^C\right) \tag{12.2}$$

where

> Pop_i = population in raster cell ("neighborhood") *i*
> Exp_i = per capita mental health expenditures in neighborhood *i*
> W_i^j = score (index value) on the World Health Organization five-question wellbeing survey in neighborhood *i* for scenario *j*, where *j* *{Current, Residential}*.

WHO-5 scores range from 0 to 25, with larger values indicating greater quality of life. The (−1) establishes an inverse relationship between expenditures and WHO-5 score.

The linkage between change in WHO-5 survey score and change in per capita mental health expenditures is assumed to be 1:1. The actual relationship is likely to be more nuanced than this ratio suggests (Buckley et al., 2019), though there is insufficient data to parameterize it in Guangzhou.

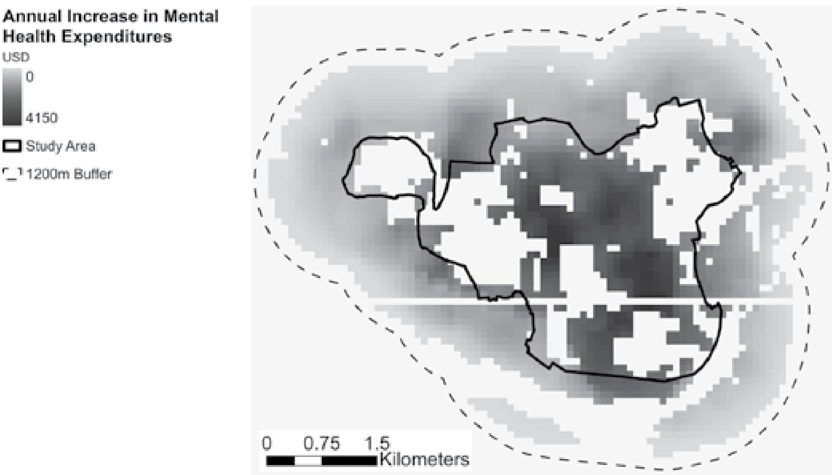
We treated per capita neighborhood expenditures on mental health as constant across neighborhoods. We derived this value from Xu et al. (2016), who estimated an annual burden in China (total social expense) of US$88.1 billion (2013) for those that elect for treatment, and US$484.1 billion if all who suffered mental health issues were treated. This latter figure is more appropriate as a social welfare metric. The population of China was 1.357 billion in 2013, so this comes out to US$356.74 per person per year, or US$389.54 in 2020 US dollars.

We calculated the expected change in expenditures for a given neighborhood between scenarios *Current* and *Residential* by substituting equation (12.2) into equation (12.1) for WHO-5 values calculated at *{Current, Residential}*. Total change in expenditure is equal to the sum of the change in neighborhood expenditures, for neighborhoods within 1 km of Haizhu wetland. Neighborhoods for this analysis are defined as population cells (~ 90 m^2) from the 2020 WorldPop global population map.

We reflected the difference between the *Current* and *Residential* scenarios through a change in NDVI within the Haizhu wetland boundaries, holding all else equal. We extracted the mean NDVI within a 1 km buffer for all neighborhoods from the baseline NDVI map, derived from Copernicus Sentinel 2/ Google Earth Engine at 10 m resolution. For the *Current* scenario, we calculated a neighborhood's mean NDVI and then used a NDVI "wallpapering" approach consistent with the land cover wallpapering (Figure 12.3) to calculate the NDVI for the *Residential* scenario. The analysis does not account for the wellbeing of any new residents that accompany a developed Haizhu wetland.

Results

Residential development leads to decreases in the value of the Haizhu wetland for mental health (Figure 12.8). Aggregate population in neighborhoods inside or within 1 km of the Haizhu wetland equals 238,000 people. Mean NDVI across neighborhoods in the *Current* scenario is 0.21; in the *Residential* scenario it is 0.15. This loss in natural areas leads to an annual increase in mental health expenditures of US$2.90 million, with a net present value of US$44.6 million over 30 years at a 5 percent discount rate.



Source: Authors' own.

Figure 12.8 Annual increase in mental health treatment expenditures expected without the wetland

Synthesis of the Haizhu Wetland Assessment

Analyzing the supply and consequent value of ecosystem services allows decision-makers to fully understand the externalities of development decisions. To examine the change in the total value of the four services examined here, we excluded the value of physical health. This is a conservative approach, recognizing that the value of physical health is theoretically incorporated within the mental health valuation methods. When viewed in aggregate, the marginal value of the explored ecosystem services provided by the Haizhu wetland totals US$146.8 million over the next 30 years (Table 12.2), in addition to reduced mortality risk and increased workplace productivity in

Table 12.2 The marginal value of four ecosystem services generated by the Haizhu wetland when compared to a residential development scenario

Ecosystem service	**Value metric(s)**	**Marginal value of the Haizhu wetland (net present value using 5 percent discount rate unless otherwise noted)**
Climate change mitigation	Social cost of carbon	US$100.3 million over 30 years
Urban cooling	Productivity	2.5 to 16.1 percent increased workplace productivity within 600 m (May and October)
	Private cost of cooling	US$1.9 million over 30 years
	Mortality risk	1.23 to 1.27 percent decreased risk of monthly mortality within 600 m (June through September)
Physical health	Avoided cost of treatment	US$3.0 million over 30 years
Mental health	Avoided cost of treatment	US$44.6 million over 30 years

Note: The value metrics and ecosystem services presented are not exhaustive, and thus this is an underestimate of the marginal value of the ecosystem.
Source: Courtesy of Authors, based on data from World Bank, 2022.

the surrounding landscape. Crucially, this is an explicit underestimate of value because wetlands, in addition to mitigating climate change, cooling the urban fabric, and bolstering mental and physical health, also improve water quality, enhance biodiversity, and mitigate flood risk, among other services (Maltby & Acreman, 2011).

While this case study succeeds in articulating the ecosystem service supply and value of an urban green space, it does not explicitly identify those who benefit most from these services. For instance, the urban cooling benefit of the Haizhu wetland will be of greatest benefit to households lacking air conditioning or individuals at greater risk of complications due to excessive heat exposure. Socio-economic status can intersect with ecosystem services to ameliorate—or exacerbate—existing vulnerabilities (Keeler et al., 2019). We must, therefore, expand our effective definition of value to include not only the services rendered, but the relative needs of the recipients as well. In short: Who benefits the most from nature?

Multiple studies have demonstrated dramatic inequities in the distribution of green space and other types of green infrastructure throughout individual cities (Gerrish & Watkins, 2018; Landry & Chakraborty, 2009; Nesbitt et al., 2019). This affects the delivery of multiple different urban ecosystem services, including urban cooling. For example, canopy cover, percentage of impervious surface, and poverty level were all strong correlates of extreme heat in Richmond, Virginia, US (Saverino et al., 2021), and the Urban Heat Risk Index and proximity to green space are associated with many indicators of social vulnerability in Delhi, India (Mitchell et al., 2021). In the second case study, we build on the themes of mapping and valuing urban ecosystem services explored in the Guangzhou case to sharpen the focus on beneficiaries by exploring the flow of benefits to different groups in Minneapolis, Minnesota, US.

Minneapolis, Minnesota, United States

Minneapolis is a city of nearly 430,00 people (United States Census Bureau, 2020b) in the Great Plains region of the US. Previous work has been done to detail methods for applying urban ecosystem service models and valuation techniques to the city (Hamel et al., 2021; Lonsdorf et al., 2021), similar to the Guangzhou case study. We present this case study here to demonstrate how an assessment of the benefits provided by urban green space and nature-based solutions can and should go beyond valuation to take into account existing socio-economic disparities and vulnerabilities, which can temper or enhance the benefits urban nature provides (see Figure 12.1).

Rather than analyzing the benefits of any one urban green space in Minneapolis, we focused instead on the beneficiaries of urban ecosystem services across the city—mapping one service (urban cooling) and exploring how its benefits flow differently to different groups, with particular attention on marginalized groups. The first step in understanding disparities in the distribution of benefits from nature-based solutions is identifying who is marginalized and why marginalization occurs in a given local context. Schemata or mechanisms of historic and/or ongoing marginalization in a particular area can include processes as broad as colonization, settlement, land seizure, racism, and classism—or they can be narrow and place-specific, such as racially restrictive housing covenants placed on properties for sale. Minneapolis has a history of seizure of Indigenous lands, racist housing and land tenure policies through "redlining" programs like the Home Owners Loan Corporation, and housing covenants disallowing sales to non-white prospective owners (Delegard & Ehrman-Solberg, 2017). Given high disparities along racial lines in Minneapolis, we focused on race and poverty as mechanisms of marginalization to analyze disparities in urban nature's contributions to human wellbeing.

Methods

To reveal the impacts of structural inequities, we analyzed whether the distribution of impoverished or Black, Indigenous, and People of Color (BIPOC) residents in Minneapolis is related to the distribution of ecosystem services. We analyzed the distributional impacts of Minneapolis' urban heat island using the InVEST Urban Cooling model (Natural Capital Project, 2023) following prior work in the study area (Hamel et al., 2021). Assessing whether the risks of urban heat island exposure correspond to the locations of vulnerable populations is of paramount importance for municipal decision-makers interested in addressing inequities in urban green infrastructure (e.g., Hoffman et al., 2020; McDonald et al., 2021). While techniques exist to analyze and detect spatial patterns of inequality for different socio-economic groups (Roberto, 2016), relatively few studies have examined patterns of distributional inequity in urban ecosystem services (but see Liotta et al., 2020; McDonald et al., 2021; Nesbitt et al., 2019).

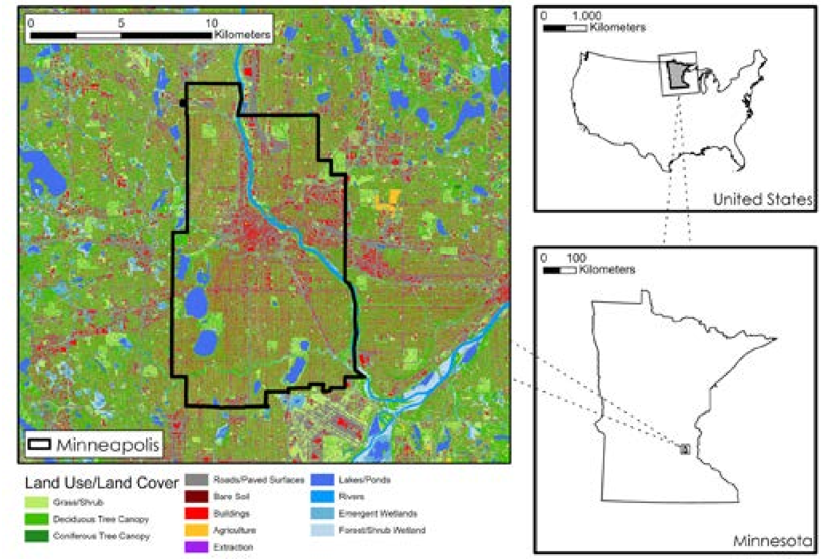
Similar to Nesbitt et al. (2019), we use a simple measure of correlation between two variables summarized at the US Census block group level—air temperature in degrees Celsius and either the percentage of the population that is BIPOC or the percentage of the population deemed impoverished in the 2018 American Community Survey (United States Census Bureau, 2020a). We mapped out how each block group contributed to the overall correlation to identify areas that are either (1) relatively cool and socio-economically privileged (white or not impoverished) or (2) relatively hot and socio-economically vulnerable (BIPOC or impoverished). This technique maps potentially uneven distributions of ecosystem services or vulnerability to environmental hazards and thus can help to highlight areas with a greater need for nature-based solutions.

Results

Areas of the city with higher poverty rates are hotter than average. High-poverty neighborhoods do not benefit as much from nature-based urban cooling (Figure 12.10A–C). The correlation between the poverty rate and air temperature was 0.57 ($r_s = 0.57$, $p < 0.01$; Figure 12.10D). A similar but less stark relationship exists for areas of the city with predominantly BIPOC residents ($r_s = 0.17$, $p < 0.01$).

Synthesis of the Minneapolis Inequalities Assessment

Urban cooling is distributed unequally in the city, particularly with respect to poverty. Revealing inequities like this can help encourage city officials to prioritize investments in poorer neighborhoods. This is especially important when the value of the services is higher for people with lower incomes. For example, more economically vulnerable people could lack air conditioning or be more dependent on publicly provided benefits as compared to privately provided (e.g., Figure 12.1). We suggest caution, however, in only using this kind of analysis to guide action. Distributional inequity often results from deeper, structural inequities and actions to improve services provided through nature-based solutions without addressing these could contribute to gentrification and displacement (Amorim Maia et al., 2020; Zhao et al., 2018). Overall, these types of distributional equity maps of ecosystem services add needed



Source: Courtesy of Authors, based on data from Host et al., 2016.

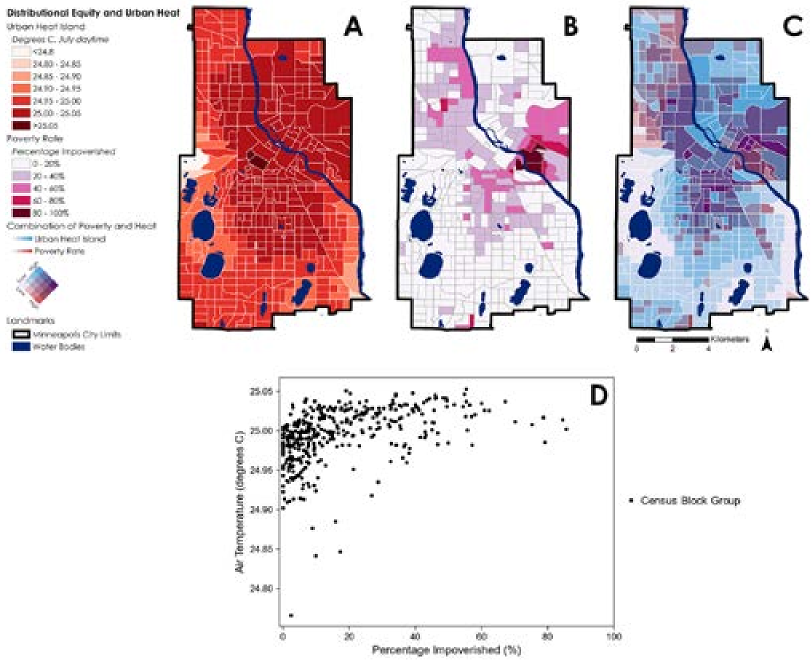
Figure 12.9 Location and land use/land cover of Minneapolis

context for decision-makers who may need to determine whether policies are improving equity and locating the most inequitable areas.

FUTURE DIRECTIONS AND CONCLUSIONS

Using Ecosystem Services to Inform Urban Decisions at Different Scales

Here, we explored two case studies in which information about urban ecosystem services can inform specific land use decisions within a city. In the first, we explored how multiple ecosystem services might change in a future scenario with a significant loss of urban green space. In the second, we examined how attention to the beneficiaries of a single ecosystem service can expose inequities and ultimately inform plans and policies that address them. Both of these examples are from roughly the same scale—exploring how LULC within a city affects the flows of benefits to people. However, decision-makers interested in incorporating the benefits provided to people from nature in the city ask different types of questions at different scales and varying levels of specificity.



Source: Authors' own.

Figure 12.10 *Maps of (A) modeled air temperature in degrees Celsius and (B) the percentage of the population per census block group below the federal poverty line in 2018, alongside (C) a bivariate map highlighting areas with high levels of both heat and poverty and (D) a scatterplot showing the city-wide relationship between heat and poverty (Spearman's $r = 0.57$, $p < 0.01$)*

At the finest scale, such as our examples from Guangzhou and Minneapolis, municipal governments, private landowners, community associations, development agencies, and public–private partnerships ask questions about how nature-based solutions can inform particular land use decisions within a city. Asking and answering these questions and those like them take place daily and cumulatively shape the future of cities. These are questions such as: What benefits do urban residents get from this natural area within the city and does it make sense to maintain it as it is? What use of this parcel will best satisfy diverse objectives? How can we best serve neighborhoods with poor access to parks for physical activity and mental health: through increasing the number and size of local parks, improving transportation to more distant parks, or programming that targets those with greatest need? As with the city-scale questions, these parcel-scale questions also require the exploration of alternative future scenarios and their likely impacts on the delivery of ecosystem services to people.

At a larger scale, questions that focus on a city as a whole can be used to prioritize investment in nature-based solutions and to create zoning plans within the city. These questions help explore opportunities and challenges in the city, such as stormwater management or resilience to intensifying temperature extremes. Illustrative questions include: Where does urban nature provide the most benefits to people in this city? Where are more trees, wetlands, or other nature-based solutions needed in the core urban area to moderate temperature? Where do parks and other forms of open space most benefit the health of urban residents? How can we improve equity in the delivery of nature's benefits to residents? These questions are being asked by city governments, planners, utilities, non-governmental organizations, environmental justice groups, and public–private partnerships. Answering these questions often requires the exploration of alternative development scenarios and their likely impacts on the delivery of ecosystem services to people.

Similarly, municipal decision-makers in a single city can examine their city as a whole, comparing it to peer cities or working to meet urban, national, or international targets for urban nature and biodiversity. In this type of enterprise, urban leaders can use peer cities to spark inspiration for the development trajectory of their city. Questions such as these are often asked principally by city governments and planners, in consultation with national governments and international organizations to guide urban design at the highest levels. They can help lenders such as multilateral development banks and non-governmental organizations prioritize investments in particular cities—or help those cities argue for such investments. Some example questions include: Is our city a good candidate for a water fund (e.g., Tellman et al., 2018)? Which cities have pioneered approaches for incorporating nature-based solutions into urban planning that my city could adopt? Here, as with questions at the global scale, understanding broad patterns of biodiversity and ecosystem service provision helps to provide context. Assessments of biodiversity and ecosystem services of particular regions or cities can help to answer these questions (e.g., Heris et al., 2021; PBL Netherlands Environmental Assessment Agency, 2021).

At the broadest scale, international organizations, national governments, and non-governmental organizations are interested in identifying cities of particular significance in achieving stated goals. Questions at this scale can help target partnerships and investments in particular municipalities. Example questions include: Which cities have globally significant levels of biodiversity either within the city or in the hinterlands that might be at risk from urban growth (e.g., McDonald et al., 2018)? What cities are good candidates for upstream or upwind investments in ecosystem restoration to improve water or air quality (e.g., Tellman et al., 2018)? Understanding global patterns of biodiversity and ecosystem service provision is a critical piece of information that can help answer these and other broad-scale questions (Brauman et al., 2020; Chaplin-Kramer et al., 2019; Díaz et al., 2018). No matter the scale of question asked, approaches to mapping, measuring, and valuing the benefits provided by nature to people can inform the strategic use of nature-based solutions in cities in ways that improve the livability, sustainability, and equity of the cities of today and tomorrow.

Improving Assessments of Beneficiaries

An important aspect of nature-based solutions that has not received as much attention to date as it deserves is the distribution of benefits across different groups in society. Disadvantaged and marginalized groups often are more dependent on their local environments but have less say over the policies that influence their environment. The Minneapolis case study examined the distribution of urban cooling benefits across both income and BIPOC groups. We found that neighborhoods with low income and minority groups that had less green space were exposed to higher temperatures than high-income and predominantly white neighborhoods.

Addressing the imbalance in benefits, however, is not so straightforward. Investing in nature-based solutions in a neighborhood increases the relative attractiveness and value of the neighborhood, an effect often referred to as "green gentrification." Approaches like ours do not (yet) predict how people may move in response to changes in ecosystem services. Community engagement around fears of gentrification and green gentrification (Ehrman-Solberg et al., 2020) suggests that adding nature-based solutions to areas of the city that house historically marginalized communities can inadvertently lead to displacement and therefore may not end up helping the people that the policy intended to benefit. Improving outcomes for marginalized communities may require addressing deeper structural inequities that go beyond investing in nature-based solutions in currently disadvantaged neighborhoods. A model that could identify when projects may lead to green gentrification due to a large influx of capital or green infrastructure investment in vulnerable communities would help decision-makers. The ability to quantify equity and the distribution of ecosystem services is critical to the equitable use of nature-based solutions in urban planning. We hope to engage the research community—and urban communities—in the creation of tools that help integrate ecosystem services and equity into urban planning.

Improving and Disseminating Models of Supply and Value

As we have shown in the two case studies, we have useful models for mapping and valuing ecosystem services in urban areas. However, the urban models in InVEST are very new (Hamel et al., 2021) and will, over time, benefit from improvement as they are applied in real decision contexts in cities around the world. Over time, we expect that there will be improvements in the breadth of coverage of benefits (and costs) from investing in nature-based solutions, as well as the accuracy and reliability of our ability to estimate the supply of ecosystem services and their value. In many cases, general models of the supply of ecosystem services are easier to create in one context and apply in others, lending themselves to inclusion in flexible, global tools such as InVEST. General models for the value of ecosystem services tend to be more context-specific and thus require more detailed site-specific information that goes beyond what can be taken from an off-the-shelf globally applicable model. Models that address issues of distribution of benefits also tend to be context-specific, requiring more detailed information about different groups within the urban area and how dependent they are on ecosystem services. Further innovation connecting global and local data sources to valuation approaches such as those outlined in Table 12.1 will allow for broader inclusion of valuation in accessible tools such as InVEST.

Urban planners use a wide range of approaches to connect today's cities with visions of the future. Lowering barriers to including information about the multiple benefits of green infrastructure in cities can help integrate knowledge of ecosystem services into urban planning, but only if tools are actionable and easy to use. Tools such as InVEST are currently accessible only to those with GIS skills and basic modeling experience. Gathering the relevant local data can also be a barrier to using these approaches in urban planning. Lowering barriers by including relevant data sources with models and enhancing the user-friendliness of the software will increase the uptake of these approaches and tools.

Ultimately, a deeper understanding of the multiple benefits provided to people by urban nature can inform smarter targeting of investments in urban nature-based solutions. Creative solutions for more sustainable and equitable cities of the future can emerge from better, more accessible models of the supply of those services coupled with better, more nuanced understanding of their values to people.

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