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Title: Community Values of Golf Courses: From the Minneapolis-St. Paul Region to US Cities

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Objectives:
1. Adapting newly developed models of urban ecosystem services to multiple cities in the United States
2. Applying these new models towards evaluating the benefits nature provides people around golf courses in urban areas
3. Engaging community with our findings

Start Date: January 1, 2020
Project Duration: 1.5 Years
Total Funding: $75,307

Summary Points:
- Golf courses are important urban greenspaces that can provide services to the community irrespective of whether an individual participates in golf.
- There is growing recognition that nature based solutions are critical to urban planning and golf courses are in the crosshairs of urban planning issues so this work is timely.
- If golf courses were developed into residential or industrial uses, we estimate that it could cost society $2-20M per course in just climate and stormwater-based ecosystem services.
- We will continue to build off this platform by adding additional ecosystem services and working to integrate our work into existing urban planning processes.
- We have developed an approach that would allow anyone in the US to evaluate the ecological value of a golf course compared to other scenarios, but it is critical to include social and economic considerations in urban planning and the ongoing discussion about golf courses and housing availability.

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Introduction

During the time we have been supported by the US Golf Association, we have worked to develop science and tools to inform the debate around golf courses and with focus on the environmental benefits they provide. Throughout the evolution of this project, an additional value regarding housing availability has emerged that expanded the potential impact of this work. We have maintained our efforts to evaluate the specific natural benefits that golf courses provide people, but we have been pulled into discussions in our communities around the availability of low-income housing and the role golf courses could play in providing it.

Rather than view this urban planning challenge as an either-or decision (golf vs. housing), a view that emphasizes sustainability and resilience would promote integrating these values into a broader framing of how much and where green space, golf courses and housing are needed to best support society from an environmental, economic and social perspective. However, without providing a clear and transparent assessment of the full benefits green spaces like golf courses could provide to the community, they are likely to be valued for a limited number of benefits. Golf courses, for example, can provide economic benefits through jobs and revenue and this is the typical argument for keeping a course - it generates revenue. However, a perception that golf courses provide only economic benefits with negative environmental and social benefits will not make them a likely sustainable land use choice in the long-term.

What has become clear is these ecosystem services evaluations cannot be done in isolation from the broader urban planning context that these courses are a part of and so the work has also connected us, surprisingly, with other social issues around equity, food insecurity, and community engagement opportunities around these broader sustainability issues. The overall objectives of this USGA funded work were to:

1) Quantify the ecosystem service benefits that a golf course provides in any US City
2) Illustrate how both the science and tools we have developed can be used to evaluate social issues such as distributional equity of those same services and inform management of rising food insecurity that emerged during the pandemic
3) Engage with several communities around sustainability-driven urban planning
4) Develop an interactive tool that allows others to view the results of our efforts in a broader sustainability context

Here we provide an overview of progress made on each of these efforts and conclude the report by describing what our next steps will be and where we think more work is needed.

Ecosystem Services Provided by Golf Courses

In our first two years on the project we developed a general approach to analyzing the ES provided by golf courses in the Minneapolis-St. Paul metropolitan area.The goal was to frame our approach so that it would ultimately be applicable to any golf course in any city in the US. We recently published this work in Landscape and Urban Planning. We have appended the publication to this report and use part of the introduction here.

There is growing awareness of the vital importance of the benefits humans receive from ecosystems (Daily, 1997; Guerry et al., 2015). Nature's contributions to people, i.e. ecosystem services, support human systems around the world from agriculture to coastal resilience (Pascual et al., 2017) and with more than half the world's population living in urban areas (Gómez-Baggethun & Barton, 2013), most of people's potential to receive ecosystem service benefits occurs in cities. Indeed, one of UN's 19 Sustainable Development Goals for 2030 (SDGs) is to create sustainable cities and communities for the increasing number of people in urban areas (ECOSOC, 2019; Elmqvist et al., 2019). However, despite this high-level attention, urban planning decisions often overlook the value of natural capital and the services that flow to urban residents (Cortinovis & Geneletti, 2018; Tobias, 2013). So while urban populations may exert development pressure to convert green spaces into residential or commercial districts (Nor, Corstanje, Harris, & Brewer, 2017), planners are not able to make fully informed decisions about these conversions without understanding how different types of green space benefit urban residents through ecosystem services (Derkzen, van Teeffelen, & Verburg, 2015; Keeler et al., 2019).

Due to the complexity of urban landscapes, mapping and measuring the unique ecosystem services in urban areas requires modifying existing models and developing new models. Ecosystem services such as carbon sequestration, stormwater nutrient retention, and pollination have been modeled and assessed in larger spatial planning contexts in rural and agricultural landscapes (Nelson et al., 2009; Polasky et al., 2008), but urban landscapes have much finer-scale heterogeneity stemming from buildings, pavement, and particular management of open spaces. For example, grasses are a common "natural" landscape within cities in the United States but the contribution of this vegetation type to ecosystem services and biodiversity is likely to vary significantly with how it is managed. An indication of the different functions of green space comes from an assessment of mammal diversity in Chicago which found that city parks, golf courses, cemeteries, and natural areas each supported different combinations of raccoons, coyotes, and deer, with various levels of richness (Gallo, Fidino, Lehrer, & Magle, 2017). As urbanization increases development pressure on green spaces across cities, it is increasingly important to have tools and processes that help identify which green spaces are most appropriate to develop.

Given golf courses are common in urban areas and the potential pressures to develop them, golf courses provide an important case study to evaluate urban ecosystem services in a changing environment. **The goal of our urban ecosystem services work is to apply newly developed models to address the question: how do golf courses support nature's benefits to the surrounding community in urban areas?** And how do these benefits, relative to other land uses, vary across different cities in the United States? Our objective was to develop an approach that allows one to answer this question in any urban area in the United States and communicate this information to scientists and potential stakeholders.

To answer these questions, the Natural Capital Project's Sustainable, Livable Cities worked to develop a new suite of ecosystem service models for urban locations called the Urban Integrated Valuation of Ecosystem Services and Tradeoffs (Urban InVEST) software package (Hamel et al. 2021) to account for management practice variation in urban areas. We used these tools with our own novel land-use land-change model to allow us to represent urban typologies (land cover and land use change) to better model how the supply of these services may change with the hypothetical redevelopment of golf.

Methodology

We applied our analysis of ecosystem services provided by individual golf courses to include seven cities across the United States: Atlanta, GA; Dallas, TX; Detroit, MI; Philadelphia, PA; Phoenix, AZ; San Francisco, CA; and the Twin Cities of St Paul and Minneapolis, MN. These cities represent a variety of ecoregions across the United States. We chose these cities as references for multiple eco-regions so that we can account for spatial variation in factors that might affect the provision of ecosystem services provided by golf courses. By parameterizing the models for each city, we hope that these approaches could be applied quickly to other cities within those same ecoregions.

Spatial Data

City extent. For each city, we used the Metropolitan Statistical Area boundary files from the US Census Bureau to delimit the study area surrounding each city in our analysis (CITE). All subsequent data was extracted from and/or clipped to these extents.

Golf courses. Unfortunately, there is no publicly-available national repository of golf course parcel boundaries. We used OpenStreetMaps to extract shapefile boundaries of golf courses within each city (OpenStreetMap contributors 2021). For quality control purposes, we did our best to hand-edit courses to exclude sections of residential housing and minimize other geospatial errors that come with crowd-sourced data such as OpenStreetMaps.

Land cover. We used the 2016 National Land Cover Database for land cover data (Dewitz 2021).

Parameterization

Since our initial study of the Twin Cities, the tools available to perform urban ecosystem service assessments using InVEST software suite have expanded (Hamel et al. 2021). Here we use the InVEST v3.8.0 Pollination, Urban Cooling, Carbon and Urban Stormwater Retention models to evaluate ecosystem services provided by US golf courses. Additionally we have applied the CADDIES flooding model to several courses in the Twin Cities to model fine-scale hydrological assessments of the mitigation potential provided by golf courses during heavy rainfall events. Environmental parameters such as annual rainfall, evapotranspiration and soils can vary across the United States, so we parameterized our models for each city. We used current literature and expert judgment to develop these biophysical parameter tables.

It is important for us to note that the current multi-city assessment is based on nationally-available land cover data, specifically the NLCD data layer. At the time of our assessment, an up-to-date land use layer suitable for golf course assessments is not available. The National Land Cover Data typically classifies golf courses as “developed open-space” in our parameter tables and the changes in management from different land use of golf courses compared to other turf land use, such as housing, cemetery or park use is not captured.

Urban Cooling

Urban heat mitigation is a priority for many cities that have undergone heat waves in recent years. Vegetation can help reduce the urban heat island (UHI) effect by providing shade, modifying thermal properties of the urban fabric, and increasing cooling through evapotranspiration. This has consequences for the health and wellbeing of citizens through reduced mortality and morbidity, increased comfort and productivity, and the reduced need for air conditioning (A/C). The InVEST urban cooling model calculates an index of heat mitigation based on shade, evapotranspiration, and albedo, as well as distance from cooling islands (e.g. parks). The index is used to estimate a temperature reduction by vegetation. Finally, the model estimates the value of the heat mitigation service using two (optional) valuation methods: energy consumption and work productivity. The full description of the model can be found on the InVEST software user’s guide (http://releases.naturalcapitalproject.org/invest-userguide/latest/index.html; Sharp et al. 2020)

We performed an initial parameterization of the Urban Cooling model for the Twin Cities, assigning measures of shade, evaporation (Kc), albedo, green area, and building intensity to each NLCD based on a literature review (Hamel et al. 2021). During this review, we found a paucity of city or even ecoregional-specific data on such parameters; rather than adapting the parameter table (Table 1) to each city, we applied different baseline air temperatures (NOAA 2021) and urban heat island magnitudes (Chakraborty and Lee 2019) based on the city's location (Table 2).

Table 1. Biophysical parameters used to apply the InVEST Urban Cooling model to each city.

NLCD Classification	NLCD Code	Shade	Kc	Albedo	Green Area	Building Intensity
Background	0	0	0	0	0	0
Open Water	11	0	1	0.056	0	0
Developed, Open Space	21	0	0.516	0.161	0	0
Developed, Low Intensity	22	0	0.430	0.228	0	0.33
Developed, Medium Intensity	23	0	0.328	0.208	0	0.66
Developed, High Intensity	24	0	0.179	0.162	0	1
Barren Land	31	0	0.613	0.232	0	0
Deciduous Forest	41	1	1.004	0.142	1	0
Evergreen Forest	42	1	1.004	0.140	1	0
Mixed Forest	43	1	1.004	0.141	1	0
Shrub/Scrub	52	0	0.968	0.189	1	0
Herbaceous Grassland	71	0	0.932	0.193	1	0
Hay/Pasture	81	0	0.932	0.171	1	0
Cultivated Crops	82	0	0.717	0.161	0	0
Woody Wetlands	90	1	1.1	0.161	1	0
Emergent Herbaceous Wetlands	95	0	1.1	0.142	1	0

*Valuing urban cooling services***:** The value of urban heat island mitigation is specific to the local and regional context, and we do not assess it here in terms of economic or health impacts. Different cities have different rates of air conditioning and their populaces are acclimated to different heat extremes (Guo et al. 2014). The value of urban heat island mitigation is specific to the local and regional context, and we do not assess it here in terms of economic or health impacts. Different cities have different rates of air conditioning and their populaces are acclimated to different heat extremes (Guo et al. 2014). Golf courses in highly developed areas with little other green space are of greater cooling value than courses surrounded by natural or park landscapes, both from a strictly ecological perspective (more green space leads to more cooling) and from a beneficiaries perspective (highly developed areas have more people that can benefit from said cooling).

Table 2. Baseline air temperature and Urban Heat Island magnitude by city.

City	Baseline Air Temperature (July, degrees C)	Urban Heat Island Magnitude (Summer Daytime)
Atlanta, GA	26.9	1.42
Dallas, TX	29.7	1.77
Detroit, MI	23.0	1.23
Philadelphia, PA	25.6	1.99
Phoenix, AZ	34.9	0.55*
San Francisco, CA	16.3	3.06
Twin Cities, MN	23.2	2.05

* We used summer nighttime magnitude for Phoenix as the daytime magnitude was negative.

Our preliminary work, however, has suggested that courses provide no more than a few thousand dollars per month per course so we have not pursued this. We translated increases in temperature into the energy required to cool a house and then used available data on energy costs to translate this into value.

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

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 the sequestering of carbon on the landscape, via natural lands and green infrastructure. Traditional methods of estimating landscape carbon storage and sequestration often focus on land cover and center on four pools of carbon: aboveground biomass, belowground biomass, soil carbon, and organic matter (Sharp et al. 2021). These pools have analogues in the built environment—soil carbon still persists underneath buildings and pavement (Edmondson et al. 2012), urban green spaces have abundant vegetative carbon stocks above and belowground, and we can even account for organic matter stored in the built environment (e.g. building materials, furniture, books) (Churkina et al. 2010). However, carbon accounting in urban areas must be expanded to include human impacts on the carbon cycle: *flux carbon*, in the form of 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). Embedded emissions are an acknowledgement of the carbon cost of development, as producing building materials and constructing the built environment generates carbon emissions that are unaccounted for in either landscape carbon or annual emissions. Increases in embedded emissions therefore represent increases in the landscape's climate impact.

Climate change mitigation supply: We reviewed the relevant literature linking our classifications to landscape **carbon stocks** (Nowak 1993, Jo 2002, Nowak and Crane 2002, Kaye et al. 2005, Golubiewski 2006, Pouyat et al. 2006, Chaparro and Terradas 2010, Churkina et al. 2010, Escobedo et al. 2010, Hutyra et al. 2011, Davies et al. 2011, Strohbach and Haase 2012, Raciti et al. 2012b, 2012a, Edmondson et al. 2012, Kellett et al. 2013, Nowak et al. 2013, McPherson et al. 2013, Luo et al. 2014, Bae and Ryu 2015, Vodyanitskii 2015, Tang et al. 2016, Yoon et al. 2016, Nero et al. 2017, Ziter and Turner 2018), **embedded carbon emissions** (Boyle and Lavkulich 1997, Norman et al. 2006, Churkina et al. 2010, Kuittinen et al. 2016, Arıoğlu Akan et al. 2017), **and annual carbon emissions** (Norman et al. 2006, Golubiewski 2006, Fissore et al. 2011, Kellett et al. 2013, Kuittinen et al. 2016, Tidåker et al. 2017, Goldstein et al. 2020) to distill a parameter table that reclassifies LULC into estimates of carbon pools and fluxes (Table 3). Using carbon storage and emissions estimates for equivalent classifications from the literature, we performed a weighted average calculation to condense that literature into the individual values presented below. We then reclassified land cover into each carbon stock pool (Mg C/ha), flux emissions (Mg C/ha/year), and embedded emissions (Mg C/ha) under each scenario.

Valuing climate change mitigation: We use the social cost of carbon for the United States, $51 per CO2e. As there is 3.47 CO2 to 1 Mg C, each Mg C is valued at $177. To calculate total costs or benefits of each scenario in each city, we multiply the average change in carbon by its cost.

Stormwater and Nutrient Retention

The runoff retention model partitions annual rainfall into the volume of surface runoff (export) and the volume of retention (abstraction, evaporation, and infiltration), along with the associated mass of nutrients (e.g., nitrogen and phosphorus). The volume of water percolating past the root zone into deeper groundwater is also estimated, though this value should be considered an upper bound on the actual groundwater recharge. Nutrient transport associated with this volume is not considered by the model. Primary spatial input datasets include land cover, soil hydrologic group, and road centerlines. A full description of the methods and Twin Cities (Minneapolis-St. Paul, MN) parameterization of the Stormwater Runoff Retention model can be found in the Supplementary Information of Hamel et al. (2021)[1], but a brief overview is included here along with a description of parameterization of the model for the other U.S. cities in our analysis. The stormwater model will be included in the next release of InVEST (likely late 2021).

[1]https://static-content.springer.com/esm/art%3A10.1038%2Fs42949-021-00027-9/MediaObjects/42949_2021_27_MOESM1_ESM.pdf

Table 3. Biophysical parameters used to apply the InVEST Carbon model to each city.

NLCD Classification	NLCD Code	Aboveground (Mg/ha)	Belowground (Mg/ha)	Soil (Mg/ha)	Litter (Mg/ha)	Embedded Emissions (Mg/ha)	Annual Emissions (Mg/ha/yr)
Background	0	0	0	0	0	0	0
Open Water	12	0	0	0	0	0	0
Developed, Open Space	21	52.9	3.4	101	4.4	0	0.3
Developed, Low Intensity	22	187.5	2.9	78	17.5	707.7	122.6
Developed, Medium Intensity	23	171.9	2.9	77.2	17.3	636.6	480.5
Developed, High Intensity	24	142.5	2.9	78.9	17.3	1627.9	860.1
Barren Land	31	6.8	0	25.5	0	0	0
Deciduous Forest	41	85.9	0	96.1	8.8	0	0
Evergreen Forest	42	105.7	0	96.1	14.4	0	0
Mixed Forest	43	91.8	0	119.2	8.8	0	0
Shrub/Scrub	52	47.9	0	68.2	0	0	0
Herbaceous	71	10.1	8	98.8	0	0	0
Hay/Pasture	81	10.1	8	98.8	0	0	0
Cultivated Crops	82	4.8	0	65.8	0	0	0
Woody Wetlands	90	33.5	0	716.9	0	0	0
Emergent Herbaceous Wetlands	95	33.5	0	716.9	0	0	0

Calculations for all volume and pollutant mass components are made for each pixel in the land cover raster. Parameters used in these calculations (listed below) are assigned to each pixel based on land cover type and on soil hydrologic group (HSG), with the latter determined from an overlay with a soil map. Major parameters used by the Stormwater Runoff Retention Model include the following (Table 4):

- *Runoff Coefficient (RC)*: fraction of annual rainfall that becomes surface runoff; is a function of land cover and soil infiltration capacity (HSG), and adjusted for surfaces with high drain connectivity (i.e., pixels in proximity to roads and high-impervious surfaces);
- *Infiltration Coefficient (IC)*: fraction of annual rainfall that is infiltrated, and potentially percolates past the rooting zone of plants and trees to recharge groundwater;
- *Event Mean Concentration (EMC)*: concentration of nitrogen or phosphorus in surface runoff, characteristic of land cover type.

Event mean concentrations of nitrogen and phosphorus were determined from published values in several U.S.-based studies where results could be attributed to the various land cover types in the land cover dataset, which ranged from dense urban to un-developed parks, and included agriculture and golf courses (Line et al., 2002; Lin, 2004; U.S. National Stormwater Quality Database, bmpdatabase.org/nsqd.html; Maestre and Pitt 2005; Tetra Tech, 2010; King and Balogh, 2011). Note that the EMC's for golf courses were derived from SWMM model runs of the Les Bolstad golf course (Falcon Heights, MN) in Phase 1 of this project (see Horgan et al. 2018).

Runoff coefficients and infiltration coefficients can also be found in the literature, but we developed an approach using EPA's Stormwater Management Model (SWMM; Rossman and Huber, 2016) to estimate these coefficients as a function of weather data to make the model more easily applied to cities with different climates from the Twin Cities. The simple SWMM model consisted of 20 synthetic watersheds with combinations of uniform land cover (n = 5) and HSG (n = 4), with land cover including bare (unvegetated), pervious (vegetated) with and without tree canopy, and impervious surface with and without tree canopy. The model was run using 10 years of local climate data (2008 – 2017) taken from the major airport in each of the seven case study cities, retrieved from Midwest Regional Climate Center[2] (see Table 4 below). RC and IC values for each of these prototype cover types were then combined based on the nominal or assumed impervious levels in each NLCD land cover category, with the further assumption that all surfaces were half covered by trees. See SI of Hamel et al. (2021) for details of this computation. The prototype RC/IC tables for each city are shown in Table 5 below.

[2] https://mrcc.illinois.edu/CLIMATE/

Table 4. Runoff coefficients (RC) and event mean concentrations (EMC) of nitrogen (N) and of phosphorus (P) assigned to the NLCD land use classes used to run the Stormwater Runoff Retention model for the Twin Cities case study.

Cover Class	Runoff Coefficient				EMC (mg/L)		References
	HSG A	HSG B	HSG C	HSG D	P	N	
High Intensity Developed	0.758	0.766	0.773	0.787	0.753	2.33	NSQD (site median EMC as function of TIA); n = 34 (N), n= 50 (P)
Med. Intensity Developed	0.548	0.576	0.601	0.649	0.544	2.53	NSQD (site median EMC as function of TIA); n = 66 (N), n = 77 (P)
Low Intensity Developed	0.296	0.348	0.394	0.484	0.294	2.34	NSQD (site median EMC as function of TIA); n = 76 (N), n = 91 (P)
Open Space Developed	0.086	0.158	0.222	0.346	0.085	2.85	NSQD (site median EMC as function of TIA); n = 22 (N), n = 23 (P)
Golf Course	0.044	0.123	0.192	0.324	0.52	2.68	King and Balogh 2011; Horgan et al. 2018
Cultivated Land	0.002	0.087	0.161	0.304	3.44	3.44	King and Balogh 2011; Lin 2004
Pasture / Hay	0.002	0.087	0.161	0.304	0.53	1.25	Lin 2004
Grassland / Herbaceous	0.002	0.087	0.161	0.304	0.53	1.25	Lin 2004
Deciduous Forest	0.001	0.078	0.146	0.277	0.11	1.23	King and Balogh 2011; Tetra Tech 2010; Line et al 2002; Maestre and Pitt 2005
Evergreen Forest	0.001	0.078	0.146	0.277	0.11	1.23	King and Balogh 2011; Tetra Tech 2010; Line et al 2002; Maestre and Pitt 2005
Mixed Forest	0.001	0.078	0.146	0.277	0.11	1.23	King and Balogh 2011; Tetra Tech 2010; Line et al 2002; Maestre and Pitt 2005
Shrub/Scrub	0.002	0.087	0.161	0.304	0.053	1.25	Lin 2004
Barren Land	0.002	0.101	0.189	0.349	0.13	1.63	Lin 2004
Open Water	0.000	0.000	0.000	0.000	0.0	0.0	NA
Emergent Herbaceous Wetlands	0.000	0.000	0.000	0.000	0.0	0.0	NA
Woody Wetlands	0.000	0.000	0.000	0.000	0.0	0.0	NA

Table 5. Runoff coefficients and infiltration coefficients determined from a 10-year (2008–2017) run of the simple SWMM model in each case study city, with climate data taken from the nearest major airport. (a)

Atlanta (Hartsfield-Jackson Atlanta International Airport), (b) Dallas (Dallas-Love Field Airport), (c) Detroit (Coleman A. Young International Airport), (d) Philadelphia (Philadelphia International Airport), (e) Phoenix (Phoenix – Sky Harbor International Airport), (f) San Francisco (San Francisco International Airport), (g) Twin Cities (Minneapolis-St. Paul International Airport).

City	Cover Type	Runoff Coefficient				Infiltration Coefficient			
		HSG A	HSG B	HSG C	HSG D	HSG A	HSG B	HSG C	HSG D
(a) Atlanta	Impervious	0.901	0.901	0.901	0.901	0.000	0.000	0.000	0.000
	Impervious w/ Tree Cover	0.864	0.864	0.864	0.864	0.000	0.000	0.000	0.000
	Pervious	0.004	0.117	0.23	0.418	0.136	0.044	0.006	0.005
	Pervious w/ Tree Cover	0.004	0.105	0.211	0.39	0.136	0.050	0.006	0.005
	Bare Land	0.005	0.137	0.265	0.468	0.135	0.033	0.005	0.005
(b) Dallas	Impervious	0.901	0.901	0.901	0.901	0.000	0.000	0.000	0.000
	Impervious w/ Tree Cover	0.866	0.866	0.866	0.866	0.000	0.000	0.000	0.000
	Pervious	0.007	0.126	0.247	0.438	0.012	0.011	0.009	0.007
	Pervious w/ Tree Cover	0.006	0.114	0.228	0.412	0.012	0.011	0.009	0.007
	Bare Land	0.008	0.148	0.281	0.483	0.012	0.011	0.009	0.006
(c) Detroit	Impervious	0.847	0.847	0.847	0.847	0.000	0.000	0.000	0.000
	Impervious w/ Tree Cover	0.793	0.793	0.793	0.793	0.000	0.000	0.000	0.000
	Pervious	0.002	0.048	0.098	0.225	0.029	0.019	0.019	0.015
	Pervious w/ Tree Cover	0.001	0.043	0.086	0.199	0.029	0.021	0.019	0.015
	Bare Land	0.002	0.058	0.12	0.271	0.029	0.019	0.018	0.014
(d) Philadelphia	Impervious	0.89	0.89	0.89	0.89	0.000	0.000	0.000	0.000
	Impervious w/ Tree Cover	0.852	0.852	0.852	0.852	0.000	0.000	0.000	0.000
	Pervious	0.009	0.118	0.206	0.368	0.188	0.083	0.005	0.004
	Pervious w/ Tree Cover	0.008	0.11	0.191	0.342	0.189	0.091	0.011	0.004
	Bare Land	0.011	0.134	0.232	0.414	0.187	0.069	0.005	0.005
(e) Phoenix	Impervious	0.81	0.81	0.81	0.81	0.000	0.000	0.000	0.000
	Impervious w/ Tree Cover	0.743	0.743	0.743	0.743	0.000	0.000	0.000	0.000
	Pervious	0	0.047	0.095	0.203	0.011	0.011	0.011	0.011
	Pervious w/ Tree Cover	0	0.042	0.086	0.181	0.011	0.011	0.011	0.011
	Bare Land	0	0.056	0.115	0.241	0.011	0.011	0.011	0.011
(f) San Francisco	Impervious	0.881	0.881	0.881	0.881	0.000	0.000	0.000	0.000
	Impervious w/ Tree Cover	0.846	0.846	0.846	0.846	0.000	0.000	0.000	0.000
	Pervious	0	0.009	0.071	0.248	0.002	0.002	0.001	0.001
	Pervious w/ Tree Cover	0	0.006	0.06	0.222	0.002	0.002	0.002	0.001
	Bare Land	0	0.015	0.095	0.297	0.002	0.002	0.001	0.001
(g) Twin Cities	Impervious	0.87	0.87	0.87	0.87	0.000	0.000	0.000	0.000
	Impervious w/ Tree Cover	0.82	0.82	0.82	0.82	0.000	0.000	0.000	0.000
	Pervious	0	0.09	0.16	0.3	0.084	0.036	0.017	0.006
	Pervious w/ Tree Cover	0	0.08	0.15	0.28	0.084	0.038	0.020	0.006
	Bare Land	0	0.1	0.19	0.35	0.084	0.032	0.010	0.005

Valuing stormwater retention:

To estimate the value provided by golf courses through nutrient retention, we used an avoided cost approach (Gómez-Baggethun and Barton 2013). We used estimates of nitrogen and phosphorus removal from Price et al. (2019) that the median annual cost of removing nitrogen and phosphorus with stormwater BMP is $2,380 and $8,440 per kilogram, respectively. We determined the change in nitrogen and phosphorus exported for each scenario, and simply multiplied this value by the export and report the average values for each scenario for each scenario. Since these services are provided each year, we estimated the total net present value using a discount rate of 7%.

Pollination

We applied the InVEST pollination model (v3.3.0) to evaluate the consequences of urban land use change for pollinators using previously validated parameter estimates (Davis et al., 2017; Koh et al., 2016; Lonsdorf et al., 2011). The model interprets land cover into floral and nesting resources for bees and provides an index of habitat quality (0 to 1), based on the spatial relationship between nesting and the foraging landscape. The NLCD data only reports a single agricultural category; past assessments provide an estimate for many different crop types grown in the US. Because each city has a different mix of crops in production, we calculated the crop-weighted average of nectar and floral resources for all agricultural land in the city. Other land cover classes in the NLCD remain the same across all cities (Table 6).

Table 6. Biophysical parameters used to apply the InVEST Pollination model to each city.

NLCD Classification	NLCD Code	Nesting Availability	Floral Resources
Background	0	0	0
Open Water	11	0	0
Developed, Open Space	21	0.324	0.489
Developed, Low Intensity	22	0.291	0.537
Developed, Medium Intensity	23	0.172	0.440
Developed, High Intensity	24	0.092	0.343
Barren Land	31	0.213	0.253
Deciduous Forest	41	0.552	0.530
Evergreen Forest	42	0.439	0.415
Mixed Forest	43	0.677	0.482
Shrub/Scrub	52	0.720	0.560
Herbaceous	71	0.383	0.450
Hay/Pasture	81	0.383	0.450
Cultivated Crops – Atlanta, GA	82	0.345	0.402
Cultivated Crops – Dallas, TX	82	0.273	0.285
Cultivated Crops – Detroit, MI	82	0.236	0.264
Cultivated Crops – Philadelphia, PA	82	0.211	0.265
Cultivated Crops – Phoenix, AZ	82	0.262	0.292
Cultivated Crops – San Francisco, CA	82	0.233	0.278
Cultivated Crops – Twin Cities, MN	82	0.175	0.226
Woody Wetlands	90	0.221	0.514
Emergent Herbaceuous Wetlands	95	0.156	0.474

Results

Urban Cooling

In general, all cities exhibit similar urban heat islands in response to land use change (Figure 1). Supplanting golf courses with more developed landscapes (high-density residential, industrial, and to an extent low-density residential) increases nighttime temperatures due to an influx of grey infrastructure. Replacing golf courses with more natural landscapes (natural areas, city parks) reduces daytime temperatures via increased evapotranspiration and shade from additional green space. Notably, golf courses in Philadelphia and the Twin Cities provide similar daytime cooling benefits and increased nighttime cooling compared to city parks. The magnitude of temperature change is driven by each city's unique urban heat island; Phoenix in particular has a very small maximum urban heat island effect (0.55 degrees C) despite being the hottest city in our study.

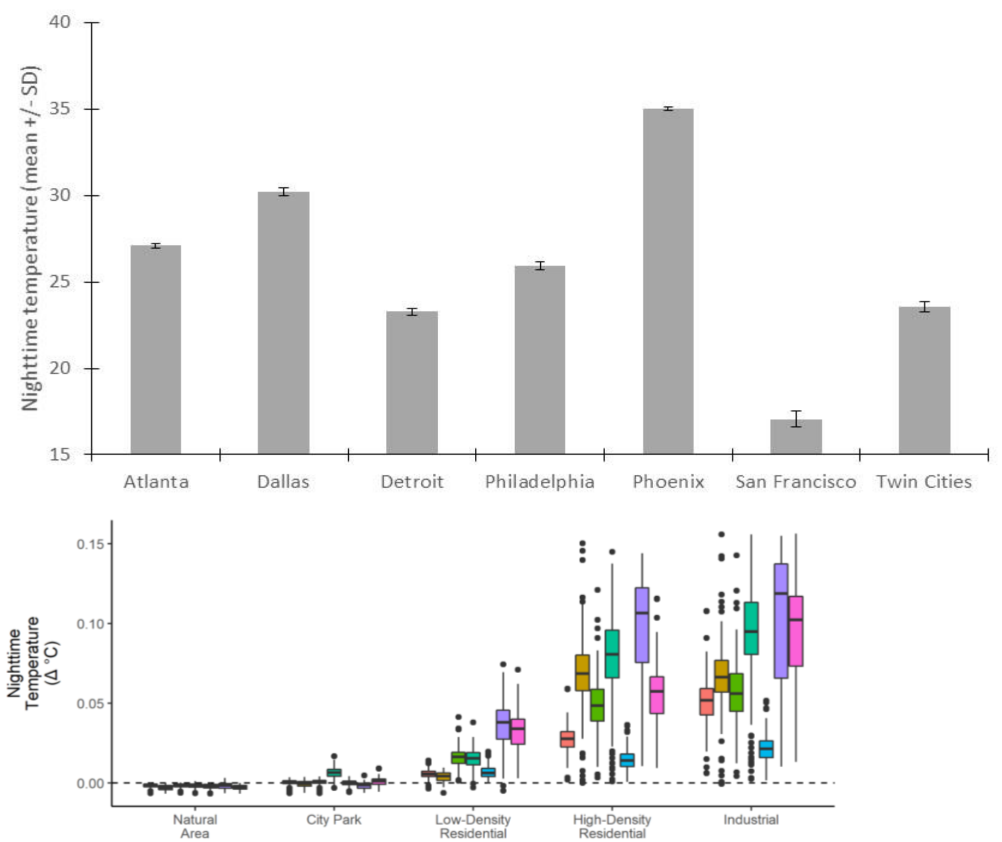


Figure 1. Expected nighttime temperatures on original golf course (A) and change in temperature (B) due to land use change. Box plots reflect interquartile range and outliers from each scenario within each city.

Climate Change Mitigation

As building intensity increases, contribution to climate change mitigation decreases (Figure 2). This decline is due mainly to increases in embedded emissions from development activities. Stored carbon in the built environment is actually higher on average than golf courses, primarily driven by carbon stored in building materials. For the most part, carbon is not affected by the city location although there is some variation among cities in carbon storage from natural areas with drier areas that are less likely to have trees, Dallas and Phoenix, showing slight decreases in stored carbon.

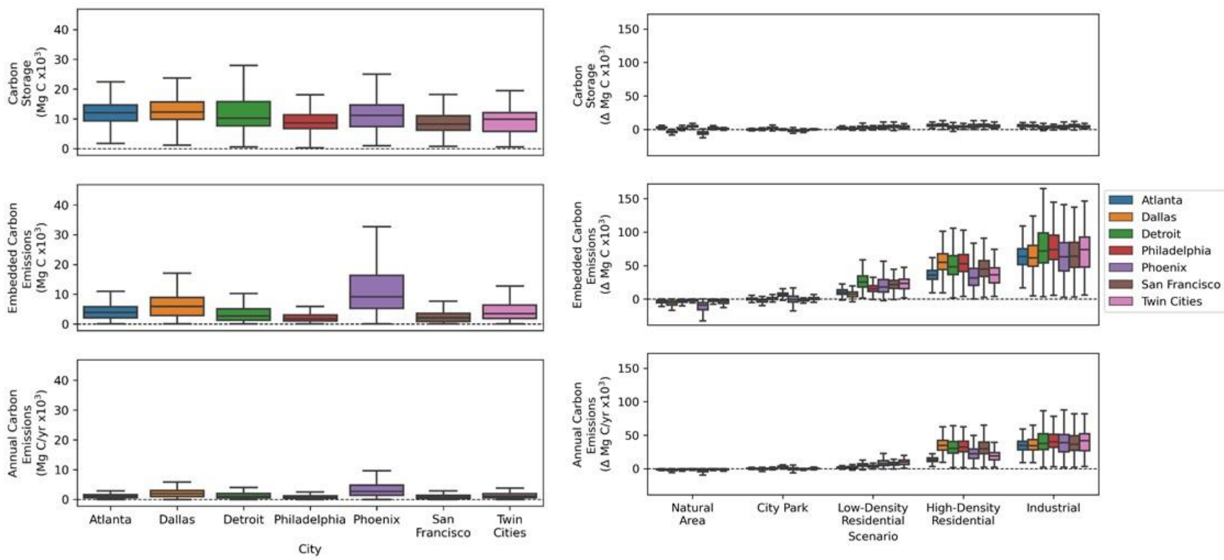


Figure 2. Modeled carbon stores and emissions (embedded and annualized) from baseline assessments in each city (left panels) and due to land use change on golf courses (right panels). Box plots reflect interquartile range and outliers from each scenario within each city.

Carbon emissions (both embedded and annualized) increase with development intensity across all cities. Natural areas see slight reductions in both emissions types compared to golf due to a lack of any built infrastructure. All developed landscapes (low- and high-density residential, industrial) increase both types of emissions. City parks act similarly to golf courses, although this depends on the city: in Philadelphia city parks slightly increase emissions relative to golf, while in Dallas and San Francisco they slightly reduce emissions.

Stormwater and Nutrient Retention

Baseline: Eco-regions and their asosciated cities differ widely in the baseline stormwater and nutrient export (Figure 3). Cities with higher precipitation tend to have higher export per golf course. Dallas has much lower infiltration rates so that is likely to have led higher expected export.

Change: As impervious surfaces increase, runoff and nutrient exports tend to increase within each city (Figure 4). Precipitation amounts magnify the effect of land use change—a city with a low annual rainfall such as Phoenix (167mm) will have lower runoff and nutrient export as there is less water to carry nutrients off the course. The opposite occurs in cities like Philadelphia (1150mm) and Atlanta (1296mm). Golf courses export similar amounts of runoff and nutrients to city parks, although the exact relationship depends on the city in question. City parks in Philadelphia export more nutrients and stormwater compared to golf courses while those in Dallas export less.

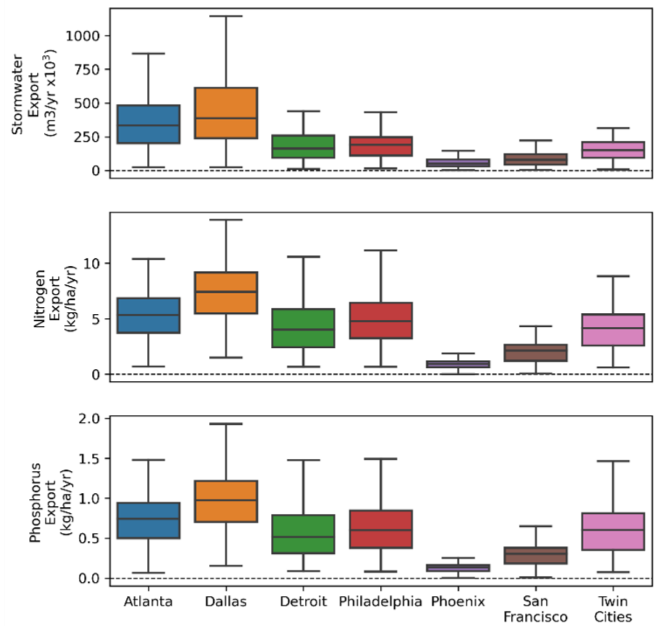


Figure 3. Expected stormwater and nutrient (nitrogen and phosphorus) export on each golf course in each city.

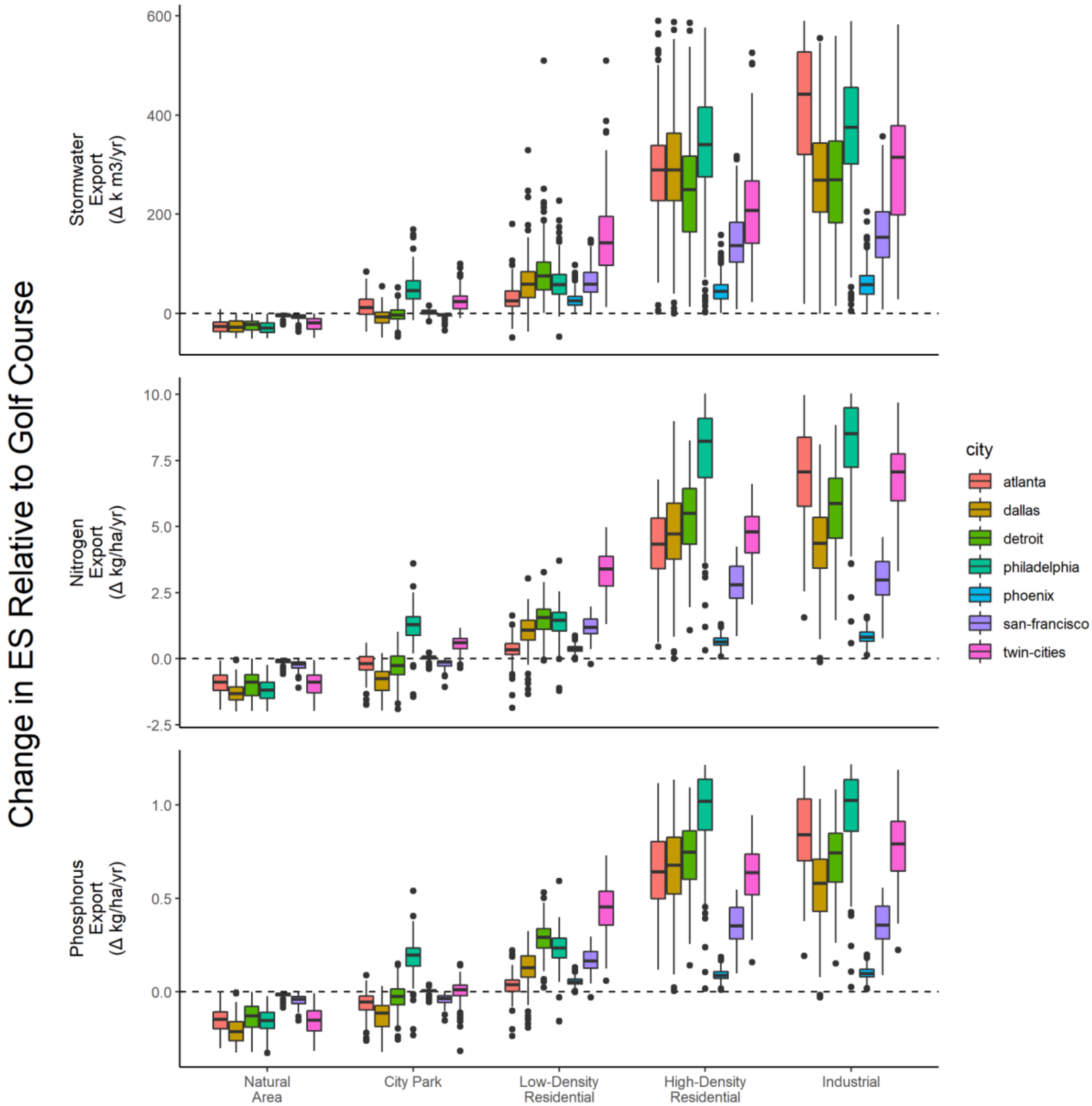
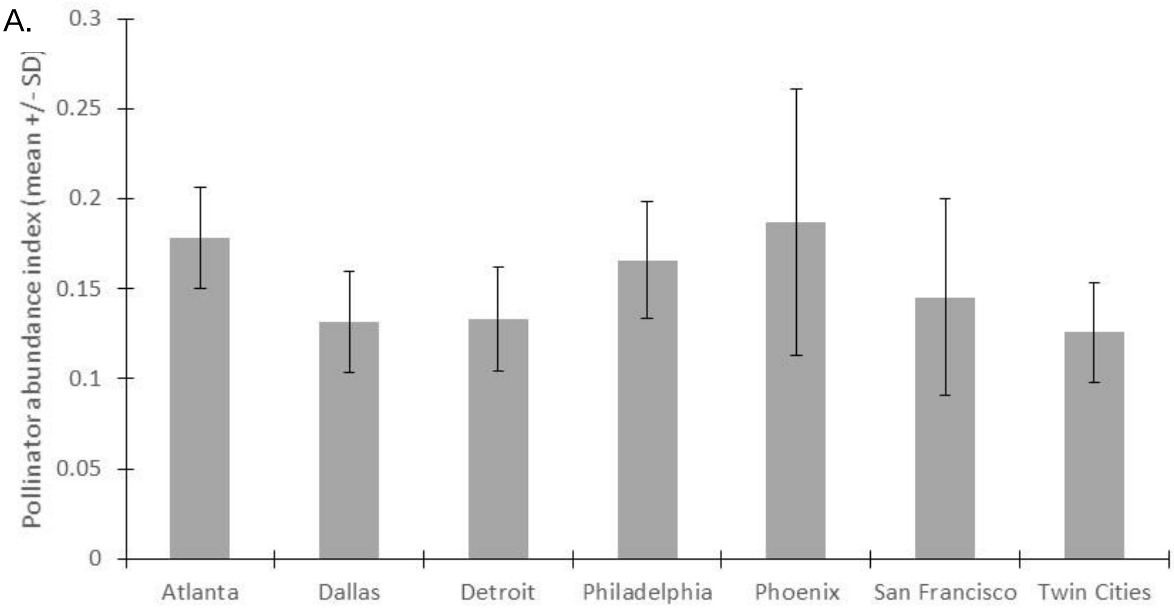


Figure 4. Change in modeled stormwater and nutrient (nitrogen and phosphorus) export due to land use change on each golf course. Box plots reflect interquartile range and outliers from each scenario within each city.

Pollination

For all cities, pollinator abundance typically falls under more intensive development (Figure 5). Highly developed scenarios (high-density residential, industrial) limit pollinator populations in the surrounding landscape while natural areas magnify it. Golf's position relative to city parks and low-density residential areas is especially city-dependent. In Dallas, Detroit, and San Francisco, pollinator abundance increases in city parks relative to golf courses, while in Atlanta it increases in city parks and low-density residential development. However in Philadelphia, city parks reduce pollinator abundance. This is likely a function of the types of green space found in parks in each city, as some natural landscapes are more suited for pollinators than others (e.g. prairie vs forest).

A.



B.

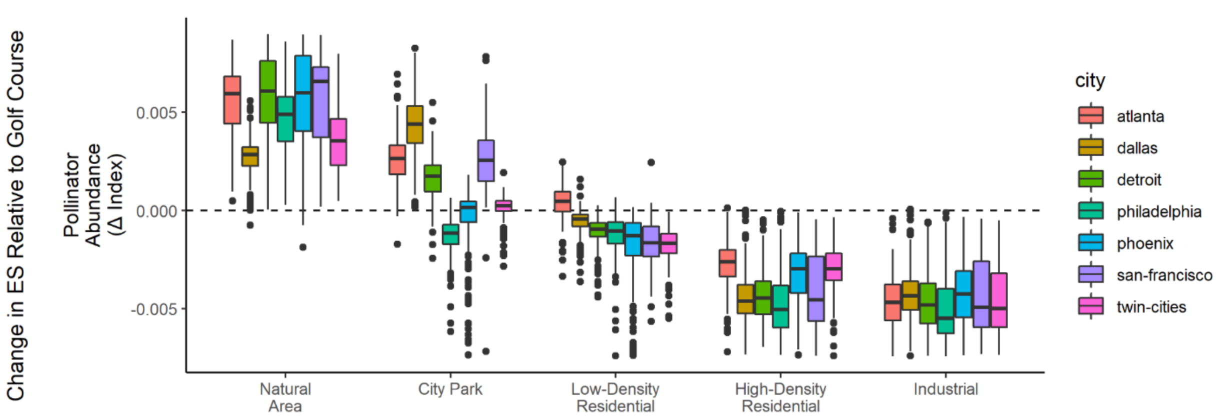


Figure 5. Pollinator abundance index. Average abundance index +/- standard deviation on original golf courses (A) and changes in index from land use change scenarios (B). Box plots reflect interquartile range and outliers from each scenario within each city.

Ecosystem Service Valuation

Golf courses can provide valuable stormwater management and climate change mitigation services when compared to alternative more intensive land uses. The differences in value between cities and scenarios simply rescale the services provided by stormwater nutrients and carbon respectively (Figure 6). A change to industrial would result in a one-time loss of $10M to $12M based on the social cost of carbon through contributions to climate change, regardless of location. Since phosphorous and nitrogen would be removed together the value estimates are similar so we provide only one service. Stormwater nutrient impacts are most valuable in Atlanta, where the estimated net present cost to remove the additional phosphorus is over $10M. As a city's precipitation decreases, the potential to remove stormwater nutrients and the value provided also decline. For high-density residential areas, the cost to society of lost carbon and increased nutrients approaches $15M per course and up to $6.5M per course for low-density residential. If every course in the Twin Cities converted to low-density residential, it would cost society an estimated net present value of nearly $1B.

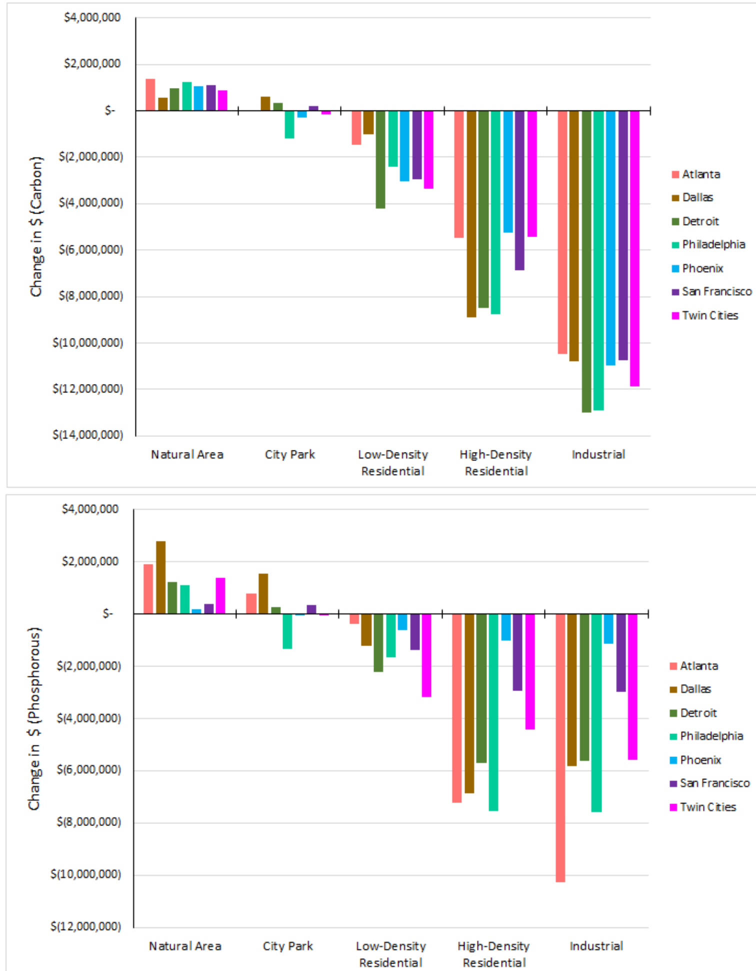


Figure 6. Change in economic indicators from phosphorus mitigation and climate change due to land use change, displayed by scenario and city.

Discussion

We have made methodological advancements in urban ecosystem service analysis, applied those advancements to study the role golf courses play in providing ecosystem services within urban environments across multiple cities in the United States. To evaluate their contribution, we (1) parameterized the inputs of urban InVEST by accounting for the effects of land cover and eco-regional differences, (2) applied a replicable framework (the “wallpapering” approach) to create and assess changes in urban ecosystem services that could inform land planning decisions and (3) began estimating the economic value to society that a few of these services provide.

In general, we found that golf courses as green infrastructure provided an intermediate amount of services compared to other five land use options (Figures 1-4) for each modeled service. Our valuation suggests that the median cost to society of converting a single golf course to residential is $2M- 20M depending on the city and land use change (which would be equivalent to $12K to $120K per acre). When compared to more intensively-developed land uses, golf courses provide increased ecosystem services but, they provide reduced ecosystem services relative to land uses with more green spaces. The magnitude of these effects differ across cities.

Across cities the benefits or cost of golf course for the supply of ecosystem services is determined by the potential to provide the service. For example, Phoenix is a city with little rainfall and a mild heat island during the night so land cover doesn’t influence stormwater services or cooling very much. On the other hand, Atlanta and the Twin Cities have higher precipitation and more extreme urban heat islands so there is simply greater potential for changes in land cover to alter stormwater and cooling services. Climate mitigation is a global service and ironically, eco-regional differences in climate don’t really influence differences among land use scenarios. As intensity of development increases, the ability to mitigate climate change decreases.

Our results suggest that golf courses are more supportive of pollinators than residential and industrial areas (Figure 4). While much of the green space within golf courses does not provide suitable habitat for pollinators (rough, fairways and greens) there are often unplayable, natural areas within courses that can provide good habitat (Colding & Folke, 2009; Threlfall et al., 2015). This is similar to suburban residential areas which have unsuitable habitat mixed with habitat that provides good nesting and floral quality (Davis et al., 2017). With more pavement and buildings, which provide no nesting or foraging habitat, our analysis suggests that urban residential developments and industrial areas reduce pollinator habitat and abundance accordingly when compared to golf courses.

Nutrient retention results with respect to industrial land use reveal a few limitations of our work. Counter-intuitively, industrial areas (despite high proportions of paved surface) produced lower nitrogen and similar phosphorus export as golf courses, due primarily to the much lower nutrient inputs to industrial land use (essentially deposition and weathering only; Table S4). However, if runoff volume retention were considered as a service, it would have been substantially higher for golf courses relative to the industrial land use due to lower impervious cover – an important consideration in general, as a primary goal of stormwater management is to prevent street flooding and protect downstream aquatic resources from washout. Similarly, irrigation, a potentially significant hydrologic input to golf courses and residential lawns that has implications for soluble nutrient transport as well as a city's overall water use, was not included in the model.. Additionally, we do not consider other potential waterborne pollutants such as pesticides and herbicides, which could pose significant environmental risk and are commonly applied on both golf course and residential turfgrass (Haith & Duffany, 2007; Weston, Holmes, & Lydy, 2009; Wittmer et al., 2010).

Incorporating Ecosystem Services into City Planning

Our approach clearly provides opportunities for the golf industry, public agencies and urban planners to assess the effects of policies on public issues of equity and sustainability. Results of models such as those we used here could be integrated into studies on environmental justice or distributive equity, whereby spatial distribution of ecosystem services is examined in the context of, e.g., socio-economic resources (Maantay, 2002). Similar approaches have been taken to assess the equitable distribution of vegetation in cities (Nesbitt, Meitner, Girling, Sheppard, & Lu, 2019)—transitioning to ecosystem service evaluation could help illuminate additional tradeoffs between sources and users of ecosystem services (Baró et al., 2016).

We note here that some of our results are biophysical outputs rather than societal or economic values, so each ecosystem service model would benefit from value-focused approach (Keeney, 1992) that translates the biophysical output into human value (Merrick, Parnell, Barnett, & Garcia, 2005). The societal value of an ecosystem service results from the interaction of social and technological factors with the supply of the service itself (Keeler et al., 2019). Engaging stakeholders early in a planning process while recognizing principles of procedural and contextual equity is essential in determining the appropriate shape of this value function (Geneletti et al., 2020; McDermott et al., 2013; Merrick et al., 2005). This marginal value approach would then be applied to human values (rather than biophysical) and would thus be one part of a more integrated process where ecosystem services are one component of city planning. With a specific decision, sensitivity analysis would be used to determine how potential uncertainty in model parameters affect the outputs of the value-function (as opposed to the biophysical supply). We suggest that while assessing urban ecosystem service supply could be standardized, the social valuation of services depends solely on local context and a proper engagement process.

Our efforts provide an approach and insights for urban planners interested in exploring public consequences of land cover and land use changes for environmental services in cities. As a portfolio of expected ecosystem services from different urban land uses, our results can allow both public and private entities to better weigh the costs and benefits of different urban development schemes by illuminating previously unquantified environmental externalities. First, the marginal value approach provides improved ability to explore how land cover and land use changes affect a common good (i.e. the provision of ecosystem services) in urban areas and to explore public consequences of private land cover decisions. By uncovering the external benefits or costs provided by any parcel of urban land, developers can work to integrate ecosystem services into impact assessments of land use planning. As a hypothetical: city planners intent on reducing nutrient export into stormwater could develop scenarios in which residential areas adopt green roofs and rain gardens and use a wallpaper approach to explore where in a city changes like this would have the largest marginal value. In this scenario, the land use categories would be expanded to include homes with green roofs and rain gardens, while the land cover patterns would not change. The approach would also allow for the impact on pollinators and nighttime summer temperatures to be modeled.

Limitations and planned improvements

In our past work, we stated that combining land cover with land use zoning data to parameterize existing ecosystem service models for urban use was an important advancement but we were not able to do that here. We relied only on land cover. We recognize that in order to apply the ecological production functions to other urban areas, it is critical to account for the ways human management alters the base land cover characteristics and these could vary depending on the location. Our estimates of carbon are limited because we cannot differentiate different types of turfgrass management. This is a limitation of our work and the results should be taken with a grain of salt. Developing a national land use layer that allows more refinement of services and better evaluation is a critical next step and we report on those advancements in our report.

In addition to improving the inputs, there are additional services that could be added. To our multi-city assessment such as flood mitigation, biodiversity beyond pollinators, home value and physical and mental health. Our broader Natural Capital Project team is actively working on each of these services and hope to have them integrated into a broader assessment within a year. Our current analyses focused on long-term average benefits of ecosystem services rather than “events”, like a heat wave or a flood. We will be working to determine if we can develop better event-models and in our next section, we report on our efforts to use CADDIES to quantify the benefits provided by some golf courses for flood mitigation.

Flood Modeling

Introduction

An important consideration in any major land use development or redevelopment is *flood risk mitigation* or minimizing impacts of runoff from large (extreme) storm events. InVEST currently includes a Flood Risk Mitigation model, which does not include the capability to simulate extent or severity of flooding (i.e., it does not simulate routing of surface runoff); rather, it provides a measure of the flood water retention (essentially infiltration of rainfall) provided by the landscape for a given size storm, typically an extreme event with a long return period, such as 100 years. In the context of this project, in which we were interested in understanding the impacts to areas around golf courses in response to land use change on the golf parcels, such a model was not applicable as it would only estimate changes in infiltration on the golf parcel itself without any impacts to surrounding areas. However, we would expect changes to flood extent or depth in areas around golf course parcels for more extreme land use changes – such as leveling out of topography, removing ponds, and adding impervious surface – that might be associated with more intense types of developments.

Therefore we turned to a standalone (non-InVEST) model to estimate potential impacts to flood extent and depth in areas around golf courses that stem from changes in land use on the golf parcels. For this task, we used a model called CADDIES[3], a two-dimensional flood model developed by the University of Exeter (UK) that uses a machine learning approach to rapidly simulate surface flooding for large storms. The advantage to such an approach is relatively fast computation times, at a small cost in accuracy, compared to deterministic 2D or 3D models that solve the full set of momentum and mass conservation equations for surface runoff (Ghimire et al. 2013; Webber et al. 2018). The CADDIES model produces a map of peak flood depths and velocities over the entire simulated domain, and time series maps of flood depth could be used to compute duration of flooding as well. Inputs to the model are straightforward: (1) elevation map (raster), (2) rainfall, (3) infiltration rates, and (4) surface roughness (Manning *n*, a parameter for a widely used equation for surface runoff). The elevation map is typically a DEM (digital elevation map) and serves to define the simulation domain, and rainfall can be spatially and temporally variable. The roughness and infiltration parameters are land characteristics, and similar to InVEST, these are mapped onto the domain (DEM) pixels through the use of an overlayed land cover map (which has the same resolution as the DEM), whose cover types correspond to unique infiltration and roughness values in a biophysical mapping table (see Table 7).

[3] http://emps.exeter.ac.uk/engineering/research/cws/resources/caddies/

Methodology

Model Setup

Inputs to the CADDIES model in our application to the TCMA were as follows. First, as was done for the InVEST Stormwater Retention Model, we developed a map of land cover x hydrologic soil group (A/B/C/D), using a 1m resolution map of the TCMA for land cover [4] and SSURGO[5] to define HSG (hydrologic soil group), with infiltration rates per NRCS (NRCS 2009). Second, we defined surface roughness (Manning *n*) for the cover types in the land cover map using values provided for SWMM (Rossman and Huber 2016). The elevation data were taken from MNTopo[6], and we used the 3m resolution version derived from LiDAR. For this reason, we had to re-sample the land cover map from 1m to 3m so that the infiltration and roughness parameters were mapped 1-to1 on each grid cell in the DEM. The DEM was clipped to a 1000m buffer around each golf course, a distance assumed to incorporate most of the flooding impacts from land use change on the golf parcels. We used a spatially-constant rainfall input to the model; rainfall rates for 1-hour duration over several return periods (1, 5, 10, 50, 100, and 500 years) were taken from NOAA Atlas-14 for the Minneapolis-St. Paul Airport[7]. A summary of parameter values used in the model are shown in Table 7 below.

The CADDIES model setup was modified further per recommendations of Dr. James Webber, a researcher at the University of Exeter (UK) with extensive experience in working with the model, who also provided input and initial guidance in the model application. These modifications included: (1) overlaying a building footprint layer[8] with the DEM and raising the DEM in these locations by 15 cm to prevent routing of shallow water over the tops of buildings, until a critical depth was reached (15 cm) when the buildings were assumed to be inundated; (2) increasing surface roughness on these building footprints to slow the movement of water through inundated buildings (Manning $n = 0.3$); and (3) overlaying a road layer[9] with the land cover map to serve as a surrogate for storm sewers, and giving these pixels an increased infiltration rate (15mm/hr) to approximate the capacity of storm sewer conveyances. These modifications had all been used in previous work applying CADDIES to urban areas (Ghimire et al. 2013; Webber et al. 2018).

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[4] https://gisdata.mn.gov/dataset/base-landcover-twincities

[5] https://catalog.data.gov/dataset/gridded-soil-survey-geographic-database-gssurgo-minnesota

[6] https://www.dnr.state.mn.us/maps/mntopo/index.html

[7] https://hdsc.nws.noaa.gov/hdsc/pfds/pfds_map_cont.html?bkmrk=mn

[8] https://github.com/Microsoft/USBuildingFootprints

[9] https://gisdata.mn.gov/dataset/trans-roads-centerlines

Table 7. Surface roughness and infiltration rates used in CADDIES simulations in the TCMA, defined by combination of 1m Land Cover Class and Hydrologic Soil Group. Note that the "road" class was defined from an overlay with a separate road layer and is not part of the original land cover classification.

1m Land Cover Data		**Roughness**	**Infiltration Rate (mm/h) per HSG Type**			
Name	**ID**	**Manning n**	**Inf_A**	**Inf_B**	**Inf_C**	**Inf_D**
Roads	none	0.065	15.0	15.0	15.0	15.0
Grass	1	0.24	36.0	19.8	3.6	1.4
Bare Soil	2, 12	0.11	36.0	19.8	3.6	1.4
Rooftops	3	0.30	0.36	0.36	0.36	0.36
Impervious	4	0.065	0	0	0	0
Lakes/Ponds	5	0.065	0	0	0	0
Deciduous	6	0.40	36.0	19.8	3.6	1.4
Evergreen	7	0.40	36.0	19.8	3.6	1.4
Ag	8	0.110	36.0	19.8	3.6	1.4
Water/Wetlands	9, 10, 11	0.065	0	0	0	0

Application to the TCMA

Our approach to applying the CADDIES flood model to the TCMA was intended to approximate the marginal value approach we used for the InVEST models (pollination, urban heat island). In this case, flood simulations were run for the baseline (golf course) scenario as well as one land use change scenario, with the differences in flooding *around* the golf course being of primary interest. As we were most interested in the maximum benefit being provided by the golf courses, we chose a worst-case as the land use change scenario: industrial land use (mostly impervious) with a flattened DEM to simulate the earth-moving and leveling that might occur in order to construct parking lots and large buildings. We used ArcGIS to provide the DEM modifications (filling holes and leveling hills by linear interpolating the DEM from the border of the parcel so that it would still match up with the elevation of the adjacent land). We acknowledge that this is in most cases an unrealistic scenario, as considerable stormwater infrastructure and ponds are often found on golf course parcels, but this was intended to illustrate the maximum possible flood benefits of greenspace.

Even with the machine learning approach, the computational time required for a single CADDIES simulation was not trivial (30-60 minutes). Combined with the need to manually download DEM data for each course, it was not feasible to apply the model to all 130+ TCMA golf courses, and instead we chose a random subset of 27 golf courses. These are shown in Figure 7. Six simulations were run for each land use scenario, corresponding to the six storm event return periods (1-hour duration each, with depths of 1.16, 1.77, 2.14, 3.17, 3.68, and 5.02 inches for the 1-, 5-, 10-, 50-, 100-, 500-year return periods). Thus, a total of 12 simulations were run for each golf course: six simulations for the baseline (golf course) case and six simulations for the worst-case scenario (flattened topography + industrial land use).

A final step in the analysis is to assess the extent and depth of flooding around the golf course parcel for each scenario. The original intent was to take the individual flood maps for each event-land use scenario and overlay this with building footprints around the courses to understand how many and to what depth buildings were inundated. Having a range of event sizes would allow the construction of a “curve” showing how flooding extent increases for an increase in storm size. There would be two curves per golf course: one for the baseline and one for the “worst-case” scenario; these could be further aggregated across golf courses with each point on the curve plotted as mean and variance across all courses for the given scenario and return period. Furthermore, these depth maps could be combined with depth-damage curves to estimate potential property loss in $USD, again as a function of storm size (e.g., Pistrika et al. 2014; Webber et al. 2019). Due to the time required to setup, troubleshoot, and complete the full set of simulations, this valuation step is still being carried out by collaborators at the University of Exeter (UK) and potentially at Nanyang Technical University (Singapore), but is unavailable for this report and will be published in peer-reviewed literature in the future. For the purposes of this report, we will show an example application and a depth-return period curve for a single golf course as a proof of concept.

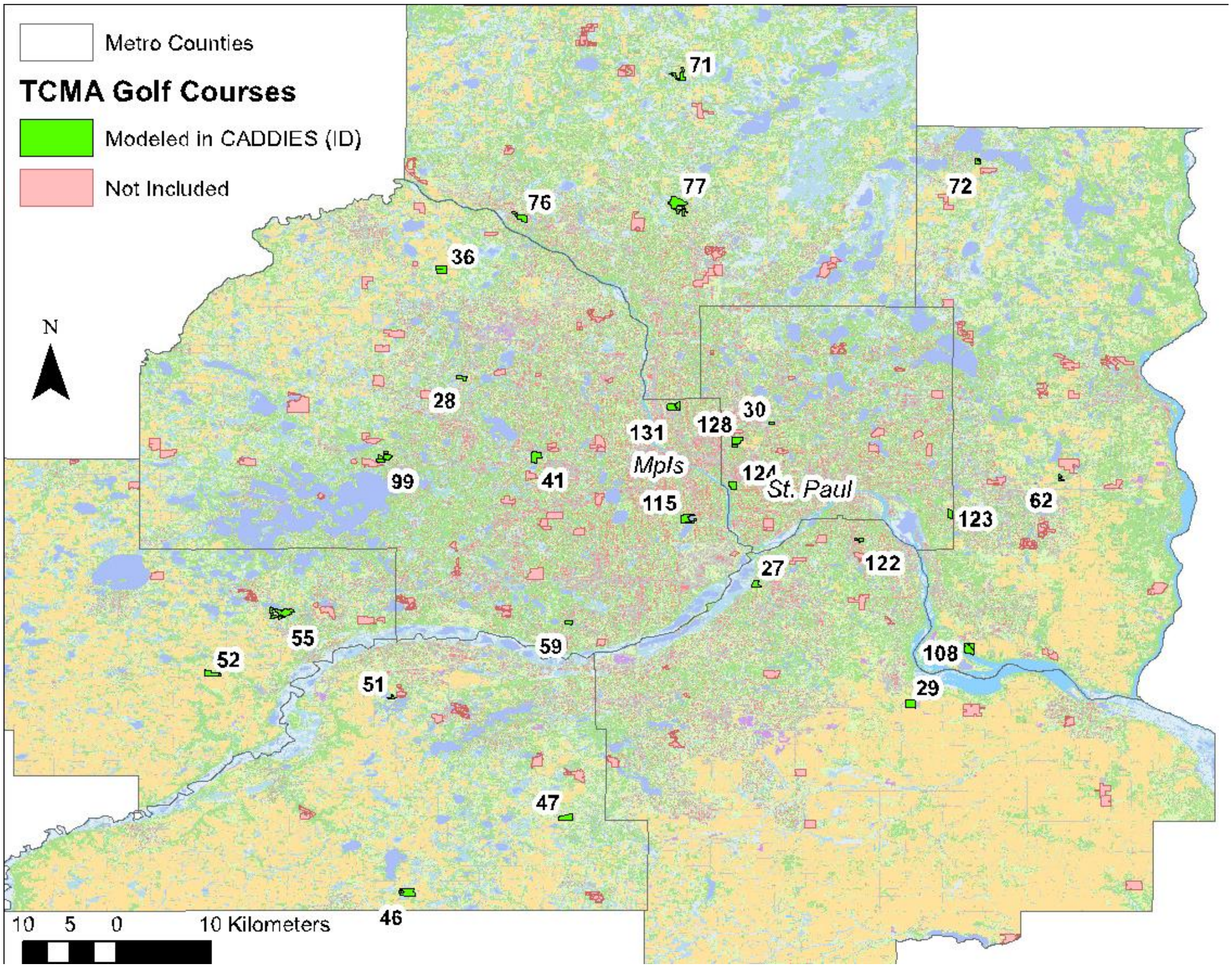

Figure 7. Location of randomly selected golf courses in the TCMA for which we conducted the CADDIES flood model analysis (n = 27). Courses located in a range of land cover contexts.

Results

Example #1:

A sample from the CADDIES application to the TCMA is shown in Figure 8 below for the industrial land use change on a single golf course (Ponds at Battle Creek; see also case study in next section), for a one-hour duration event with a 100-year return period (3.68 inches). The flood difference map (Figure 8d) is computed as the difference between the baseline flood depth map and the industrial scenario flood depth map, with red indicating areas that have greater flooding under the industrial scenario. As this simulation shows, some flooding of streets and potentially of basements would occur in this extreme scenario of land use change on the golf course (see Figure 8e, showing overlay of flood depth map with building footprints). The suggestion is that areas near golf courses may become more flood prone when golf courses are converted to land uses with more impervious surface (such as residential and industrial) -- *in the absence of any stormwater control measures, which are typically required in new development*.

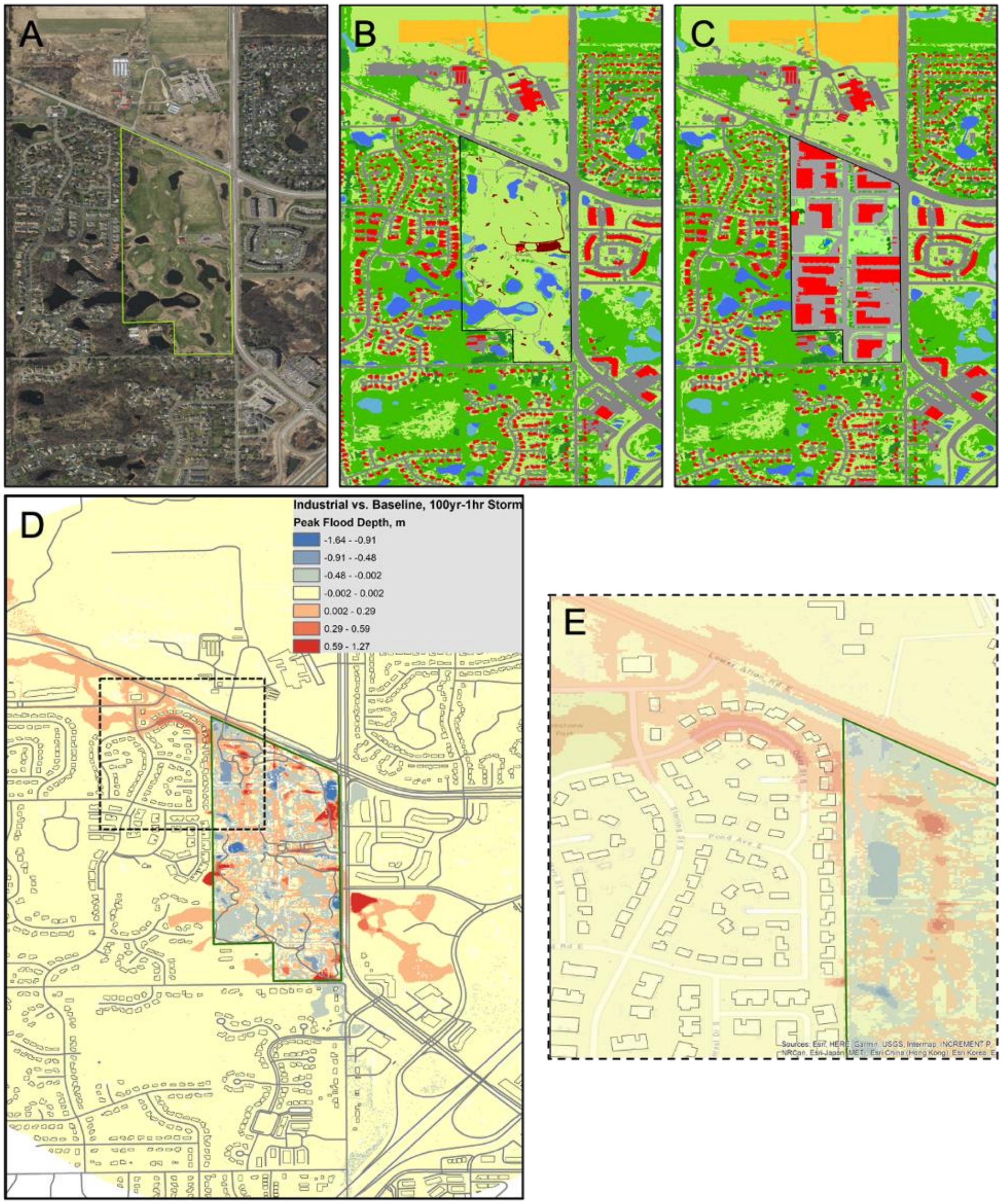

Figure 8. Application of the CADDIES flood model to a golf course for a 1-hour 100-year storm (3.68 in rainfall depth). Aerial photo shown in (a), with existing land cover shown in (b). Modified land cover (c) is for an industrial-type land use with flattened elevation (i.e. ponds filled in). Resulting change in peak flood depth for a scenario of industrial land use on the course is shown in (d), and detailed inset at right (e) to illustrate street flooding, with red shading indicating areas of greater flooding for this scenario.

Example #2:

We present here the summary of flood results for a single golf course (Parcel 131; Figure 9), using the number of inundated buildings as the output metric. In this analysis, we examined all building footprints in the 1000-m buffer around the golf course parcel and determined how many of these had at least some inundation (defined as a flood depth of ≥ 15 cm anywhere on the footprint). This process was repeated for all events (1-, 5-, 10-, 50-, 100-, 500-year return periods) and land use scenarios (baseline and industrial).

In Figure 9, we show the flood difference map (industrial – baseline) for the 500-year return period, with the red color indicating areas where there is greater flooding in the industrial land use scenario. As this map shows, most of the flood depth differences are located within and adjacent to the golf course, with potential impacts to buildings in the reddest areas to the northwest and east. The buildings with darkest shading (n = 21) are those that were not inundated in the baseline scenario but that become inundated (water depth > 15 cm) in the worst-case industrial scenario; these are the buildings that are receiving potential flood mitigation "benefit" from the golf course. **It is important to note that this is a *potential* flood benefit, derived by comparison to an unrealistic but worst-case scenario. Our analysis cannot precisely quantify the actual benefit being provided to the buildings.** Also, of the 4,607 buildings in the 1000-m buffer, roughly 105 experienced an increase in flooding of any kind (flood difference > 0 cm) for the development scenario vs. the baseline scenario in the 500-year event, suggesting that a 1000-m buffer was much larger than needed for this assessment. Finally, we note that 419 buildings suffer inundation (depth > 15 cm) in both the baseline and industrial scenarios (shown in light orange shading in Figure 9) for the 500-year event, indicating that for this combination of extremely large storm and a buffer size, roughly 80% of the total inundated buildings are *not* affected by the golf course land use.

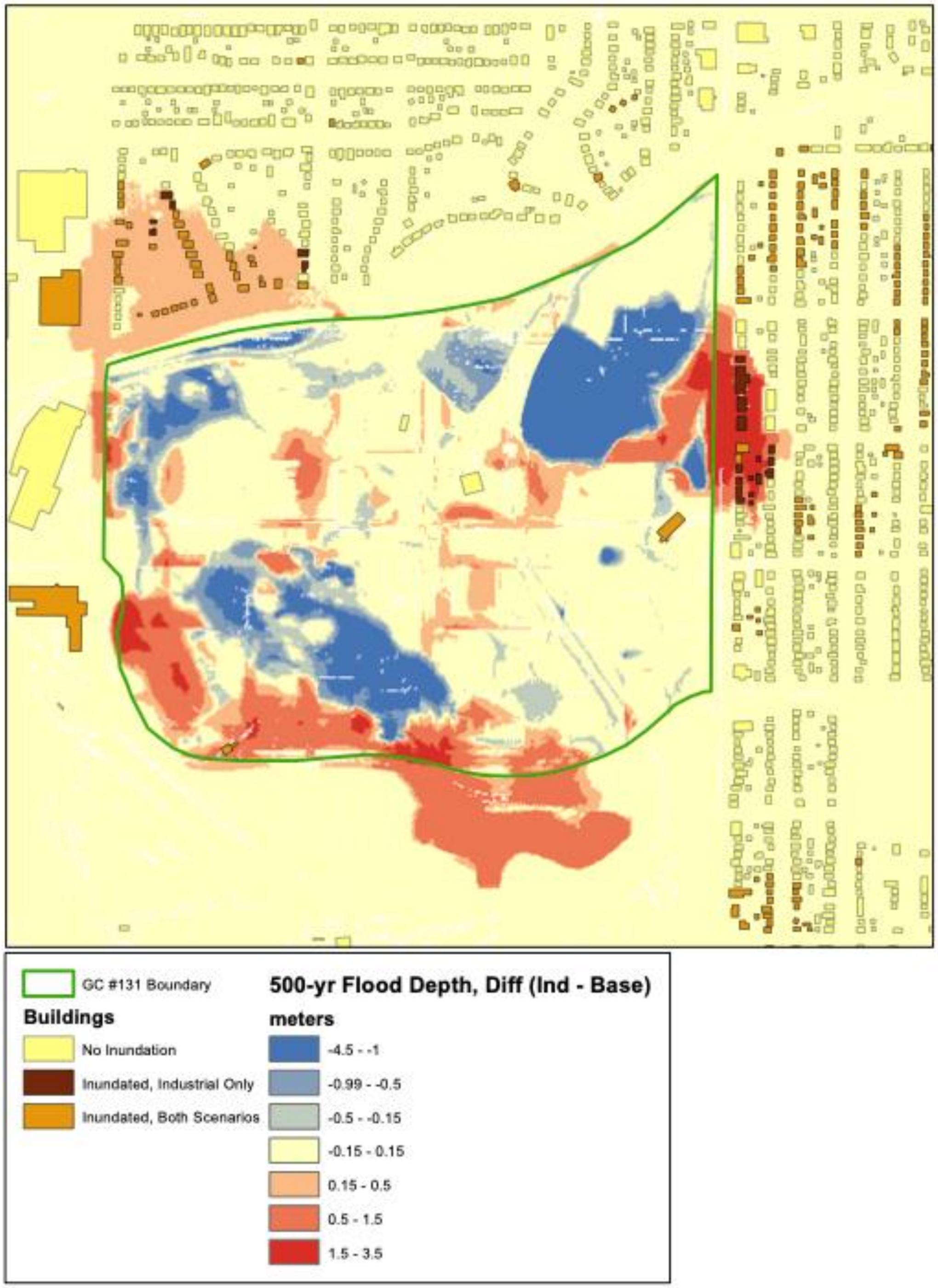

Figure 9. Flood difference map for golf course parcel #131 for a 500-year storm event (5.02 inches in one hour), with red indicating areas of greater flooding in the industrial land use scenario and blue indicating areas of less flooding. Building footprints: darkest shading indicating buildings that are inundated at > 15 cm water depth in only the industrial scenario, while the light orange are flooded in both scenarios. Note that the entire 1000-m buffer is not shown, and some buildings inundated in both scenarios are outside map extent.

In Figure 10, we show a summary of flooding impact to buildings around this particular golf course for a range of storm sizes under the two land use scenarios. Smaller extreme storms (1 – 10 year return period) produce very little difference in flooded buildings between the two land use scenarios. At larger storms, we start to see larger differences, suggesting a greater benefit from the pervious land cover and surface depressions on the golf course. In extremely large storms, the rate of rain infiltration into the ground is often greatly exceeded by rainfall rates, and therefore topography (and to an extent, the amount and connectedness of impervious surfaces that increase runoff rates) tend to dictate where and how much flooding occurs. Unsurprisingly, the industrial land use, which was nearly 100% impervious cover and very flat, produced flooding on and around the golf course (Figure 9), which started to impact surrounding buildings at higher rainfall rates. The curve in Figure 10 suggests roughly 20 more buildings are flooded in the industrial vs. baseline case at 100-year return period, and 21 more (industrial vs. baseline) at 500-year return period.

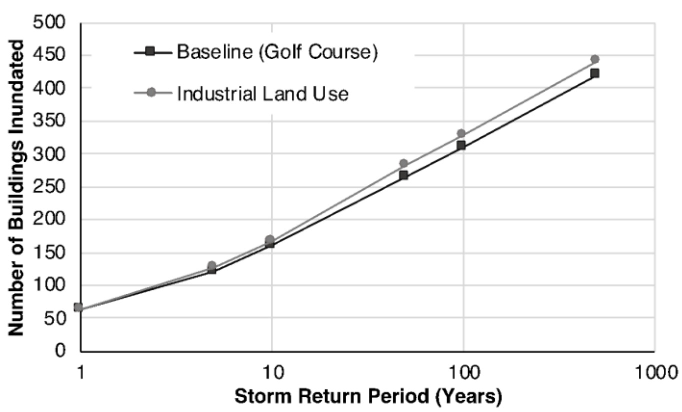

Figure 10. Number of buildings inundated at greater than 15 cm depth in both the baseline scenario and worst-case (industrial) scenario on golf course parcel #131 for a range of storm return periods (1-year to 500-year). Note log scale for x-axis.

Ponds at Battle Creek Application

A flooding analysis was not explicitly included in the Lonsdorf et al. (2021) paper that provided the basis for several scorecard metrics related to ecosystem services of (for example) pollinator habitat and urban heat island mitigation. So, we provide here an illustration of the flood modeling approach (using the CADDIES model) to make estimates of qualitative scorecard ratings (low, medium, high) from more quantitative models.

As this modeling was carried out prior to development of the scenario plans, we simulated uniform land use scenario changes on the golf course (e.g., conversion from golf course to 100% single-family residential land use), identical to the approach in Lonsdorf et al. (2021). We simulated flooding for a 1-hour duration storm with a 100-year return period (3.67 inches; NOAA Atlas-14) on four total land use scenarios: 1) golf course (baseline), 2) single-family residential, 3) industrial, and 4) city park (Figure 11). These three land use change scenarios were assumed to bracket the range of possible development on the golf course.

For the three land use change scenarios (non-golf land use), the extent of flooding on and around the golf course was compared to flooding in the baseline (golf course) scenario. These maps are shown in Figure 12 below. **Note***: no attempt was made to incorporate the benefits of existing stormwater infrastructure (ponds, pumping stations, and storm drains) into these simulations, and therefore they represent a sort of worst-case scenario of no infrastructure present. In other words, the flood maps can be thought of as showing the amount of flooding that would need to be mitigated in a given scenario in the absence of any stormwater infrastructure.* We are aware that considerable infrastructure exists on the golf course, including a pumping station. **Therefore, these results are intended to be compared against each other rather than considered in isolation**.

As in the primary CADDIES application, the industrial land use, which was nearly 100% impervious cover and very flat, produced the most flooding on and around the golf course. The residential land use produced a moderate amount of flooding, and the golf course and city park land uses produced much less flooding. Therefore, we assigned a flood mitigation rating of “low” to the industrial land use, or any scenario plans with higher amounts of paved surfaces; a rating of “medium” to residential land use and any scenario plans with higher amounts of housing; and a rating of “high” to park and golf course land use, and any scenario plans with high amounts of undeveloped or vegetated land use.

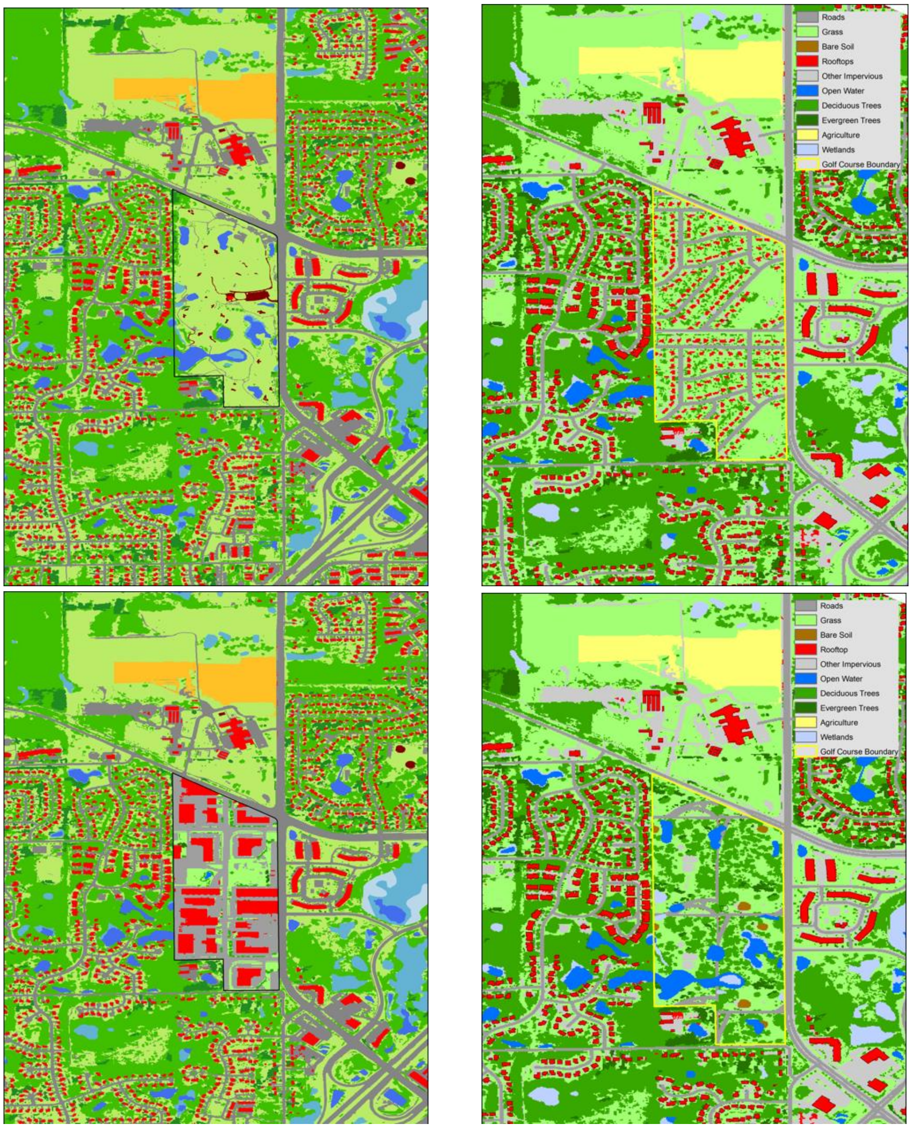

Figure 11. Illustration of land cover on the four land use scenarios considered in the initial flood modeling analysis. Top left: baseline (golf course), top right: single-family residential, bottom left: industrial, and bottom right: city park.

Figure 12. Flooding extent and depth for a 1-hour, 100-year storm (3.85 inches) based on NOAA Atlas-14. The plots show the flooding for the given land use scenario (top left: single-family residential, top right: city park, bottom: industrial) relative to flooding on the golf course as it is currently. Note difference in depth scale among maps. Red areas are locations where more flooding occurs vs. baseline scenario, blue areas are location where less flooding occurs vs. baseline scenario (primarily from less runoff into the ponds).

National Land Use Dataset

One key finding of our pilot study in the Twin Cities was that patterns of land use (the human-driven management practices associated with a parcel of land, e.g. fertilizer use) moderated the provisioning of urban ecosystem services beyond what can be modeled simply by land cover (the actual things present on the land, e.g. grass). When we analyzed the TCMA, we leveraged zoning data in the Twin Cities to identify key land uses that would affect our model parameters: for example, we increased the expected levels of nutrient loading on any grass cover within residential housing (Lonsdorf et al. 2021). However, a standardized land use map like the one we used for the TCMA publication does not exist for the United States. We used land cover only.

We feel that a national land use dataset that complements a national land cover data set is a prerequisite for future work so think it will be critical moving forward to include effects of land use and management in addition to land cover in the future. Fortunately, Theobald (2014) created a National Land Use Dataset (NLUD) by integrating many other nationally available data based on job sector data, census information, protected areas and more. Theobold's layer was based on data from 2010 so we have worked to replicate his work with updated nationally available datasets. For this project, we follow the general methods from Theobald (2014) but with several changes to adapt to modern data availability and organization (see Figure 13 for preliminary results). We are currently writing up these methods for publication and will be updating our parameter tables so that we can rerun our analyses accordingly. The land use classes are described to the right (taken directly from Theobald's 2014 paper).

Level 1	Level II	Level III
1 Water	Natural - area	Lake
		Swamp/marsh
		Playa
	Human - area	Reservoir
	Natural - linear	River
		Wash
	Estuary	Estuary & complex channels
	Human - linear	Canal/ditch
	Wetlands	Wetlands
	Ocean	Open ocean
		Bay inlet
2 Built-up	Residential	Dense urban (>0.1 ac)
		Urban (0.1–1)
		Suburban (1–2.5 ac)
		Exurban (2.5–10 ac)
		Rural (10–40 ac)
	Commercial	Office
		Retail/shopping centers
		Entertainment (stadiums, amusement, etc.)
		Lodge
	Industrial	Factory, plant
		Landfill (land fill, waste trt.)
		Confined animal feeding
		Utilities (power, sewage)
	Institutional	Schools (dev)
		Schools (undeveloped)
		Medical (hospitals, nursing home, etc.)
		Government/public
		Military/DOD/DOE (dev)
		Military/DOD (training)
		Fire & police stations
		Church/religious
		Prison/penitentiary
	Transportation	Airports (developed)
		Highways, railways
		Other transportation
		Port, train station
		Undeveloped
	Miscellaneous	Rural buildings, cemetery
3 Production	General	General agricultural
	Cropland	Cropland/row crops
		Pastureland
		Orchards
		Sod & switch grass
		Aquaculture
	Rangeland	Grazed
		Stock tank
	Mining	Mining strip mines, quarries, gravel pits
		Mine shafts
	Timber	Timber harvest
		Timber plantations
	Extractive/barren land	Oil/Gas wells
		Misc. barren
4 Recreation	Undifferentiated park	General park
	Developed park	Urban park
		Golf course
		Motorized
		OHV staging area/trailhead
		Resort/ski area
		Marina
		Campground/ranger station
		Picnic/trailhead
		Boat/fishing access
	Natural park	Natural park
		Designated recreation area
		Designated scenic area
Conservation	Public	Wildlife habitat (hunting & fishing)
		Conservation area (BLM)
		Nature reserve
		Wilderness
		Areas of Critical Env. Concern, Research Natur
		Fish & Wildlife Service refuge
		Wilderness study area
		Archaeology, historic site, scenic area
		Wild & Scenic river
	Public-limited access	Municipal watershed
		Corps of Engineers dam
		Marine Protected Area
	Private easement	Wildlife conservation
		Agricultural conservation

doi:10.1371/journal.pone.0094628.t001

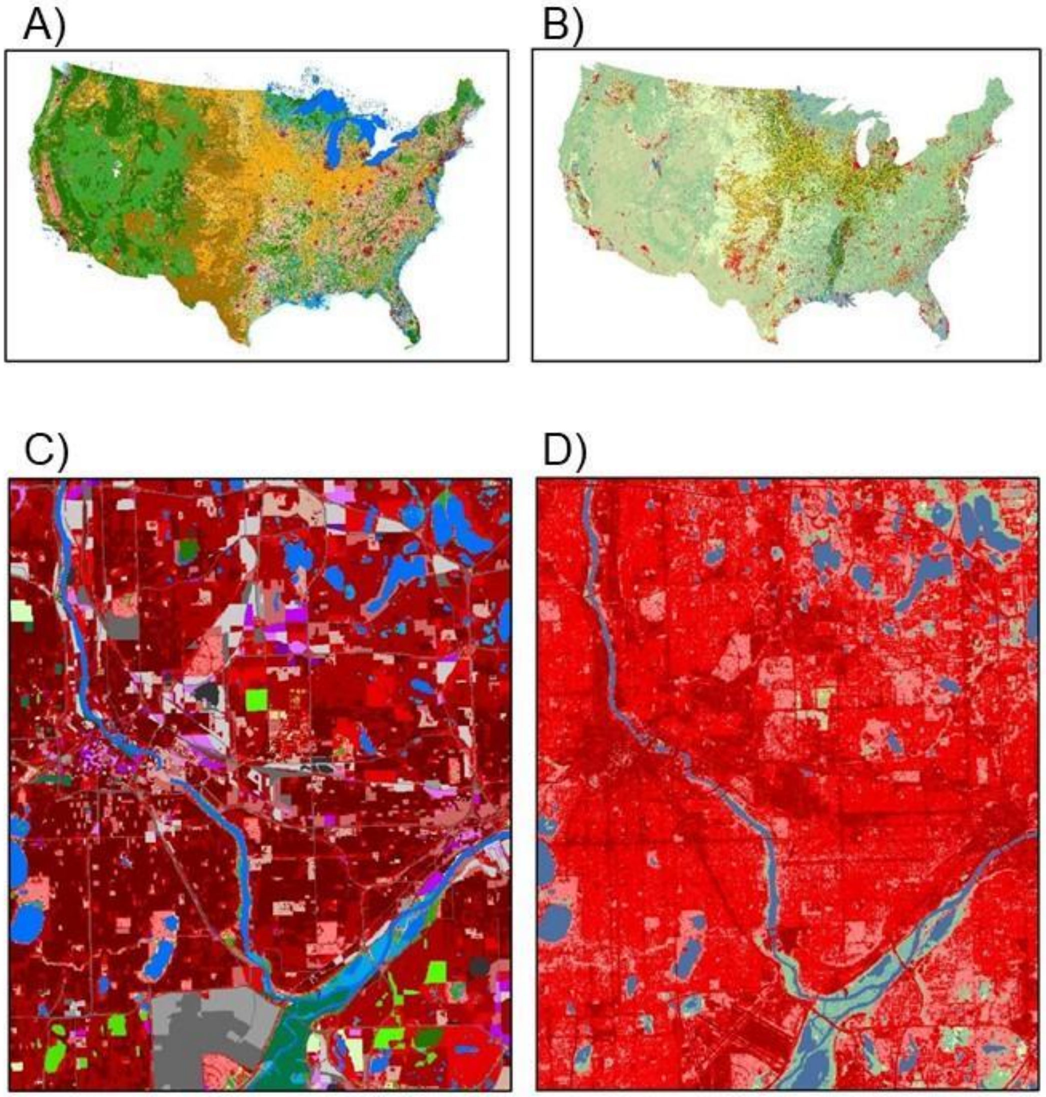

Figure 13. National land use and land cover maps. We have developed an updated national land use dataset (A) which can be used with an existing national land cover dataset (B) to improve our ability to model urban ecosystem services. Panels C and D show the Twin Cities and how land use data (C) would provide additional heterogeneity in additional to the land cover (D). For reference, the Les Bolstad golf course is the bright green square toward the center part of panel C.

Distributional Equity

As our work progressed and we engaged more with stakeholders, it became clear that decisions about green infrastructure and nature's benefits on golf courses in US Cities must also include equity. 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 from nature?

In Minneapolis, we focused on the beneficiaries of urban ecosystem services—mapping one service and exploring how its benefits flow differently to different groups, with particular attention to 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 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 and 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.

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 (Sharp 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 (Nesbitt et al. 2019, Liotta et al. 2020, McDonald et al. 2021).

Methods: 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 percent of the population that is BIPOC or the percent of the population deemed impoverished in the 2018 American Community Survey (United States Census Bureau 2020). We mapped out how each block group contributed to the overall correlation to identify areas that are either (a) relatively cool and socioeconomically privileged (white or not impoverished) or (b) relatively hot and socioeconomically 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: In Minneapolis, we found that 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 14 a-c). The correlation between the poverty rate and air temperature was 0.57 (r_s=0.57, p<0.01; Figure 14d). A similar but less stark relationship exists for areas of the city with predominantly BIPOC residents (r_s=0.17, p<0.01).

Discussion: The results show that ecosystem services are 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., Fig. 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 (Zhao et al. 2018, Amorim Maia et al. 2020). Overall, these types of distributional equity maps of ecosystem services add needed context for decision-makers who may need to determine whether policies are improving equity and locating the most inequitable areas.

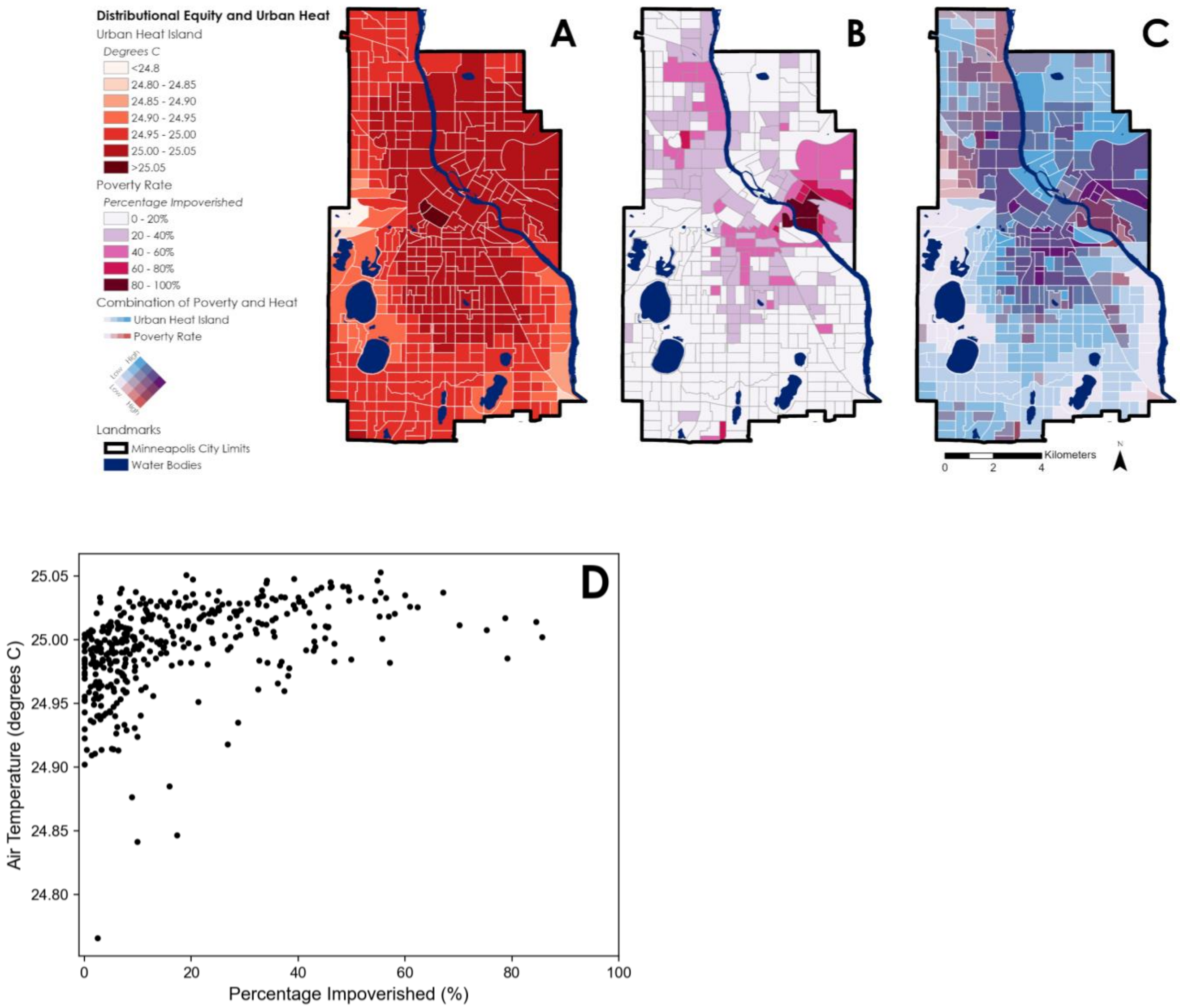

Figure 14. Maps of (A) modelled air temperature in degrees C and (B) the percentage of the population per census block group below the federal poverty line in 2018, alongside (C) a bivariate map highlighting areas with high levels of both heat and poverty and (D) a scatterplot showing the city-wide relationship between heat and poverty (Spearman's $r = 0.57$, $p < 0.01$).

Engagements

As we were working on the project, we made several connections with other engagements around urban planning, some directly related to the project around the sale and redevelopment of golf courses and some less related to the project on the general spatial distribution of benefits to people that our growing resources and knowledge could address. These engagements have proven to be extremely helpful in shaping the direction of our work and hope to provide insight to the USGA as to the challenges and opportunities ahead for golf. The first example of engagement we describe was not golf related but it speaks to changing priorities in urban areas that have related from the death of George Floyd and the inequitable impacts of the pandemic.

The second engagement we describe involves the proposed sale of a golf course in the city of Maplewood, MN a suburban city of the Twin Cities. Ramsey County owns the land while Maplewood controls its zoning. The third engagement involves town planning in Warren, MN, a small town in northern Minnesota. We are working with the University of Minnesota College of Design to integrate our tools into a community planning process designed to envision potential sustainable futures.

Food Insecurity in Minneapolis, MN

On May 25, 2020, George Floyd was murdered by police in Minneapolis. The civil unrest that followed resulted in the destruction of many buildings, including several grocery stores. Because of the economic impacts of the pandemic, food insecurity was already increasing in the TCMA but it seemed unclear how the destruction of grocery stores might affect this or whether certain socio-economic groups might be affected more than others. Our work on distributional equity made it a little easier for us to leverage our mapping skills, so our team reached out to Minneapolis officials to see if a broad overview of food insecurity might help and they accepted. On June 20th, we provided an impact assessment of the loss of those grocery stores (see Memo below). We continued to provide updates on the assessments and based on these assessments, the city allocated an additional $1M towards food insecurity after a follow up assessment in November, 2020.

While this work was not explicitly related to golf courses and ecosystem services, it does speak to the broader geographic contexts into which urban planning decisions regarding golf courses are occurring.

Locations for Pop-up Grocery Stores to Alleviate Food Shortages in Minneapolis

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University of Minnesota - Twin Cities, Institute on the Environment

June 20, 2020

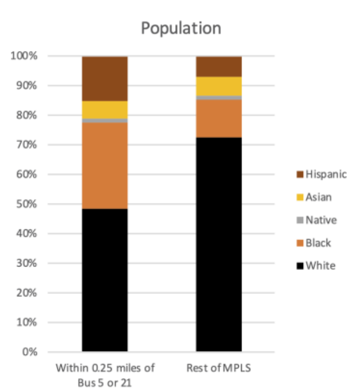
We have identified and mapped communities in Minneapolis that have been most affected by the loss of food access due to grocery store closures resulting from the recent civil unrest. These results are designed in response to a request from the City of Minneapolis to support decisions on where the community would benefit the most from rebuilding efforts to improve food security.

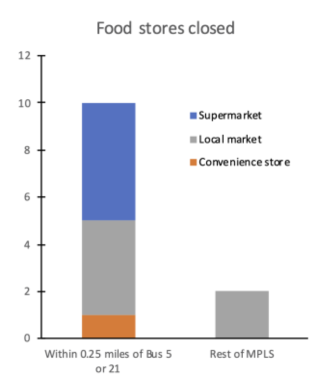
Three Things You Must Know

1. Eleven grocery stores (Map #1) and supermarkets have closed in Minneapolis due to damage sustained in the recent riots. This has decreased food access, by increasing the distance people need to travel to get to a grocery store and significantly reducing food supply in certain areas.

2. There are two critical locations for pop-up grocery stores to reduce the greatest loss of food access, especially for people with mobility limitations:
   - North Minneapolis in the area around the closed Cub, which has few alternative supermarkets (Map #2), and
   - the area surrounding the intersection of Lake and Hiawatha, which suffered the greatest number of store losses (Maps #1 and #3).

3. The areas affected by store closures are more racially and ethnically diverse and have lower incomes relative to city demographics as a whole (see figures to the right). Nearly all food stores closed (~80%) are within 0.25 mile of bus lines 5 or 21 - public transportation routes with the highest ridership.
   - The area near the bus routes contains 35% of the Minneapolis population, 49% of the Minneapolis' non-white population, and 50% of the city population living in poverty.





- There are 9 major supermarkets within 0.25 mile of these two bus lines; 5 of those stores are closed. There are 11 major supermarkets outside this buffer zone; none are closed.

Key Assumptions

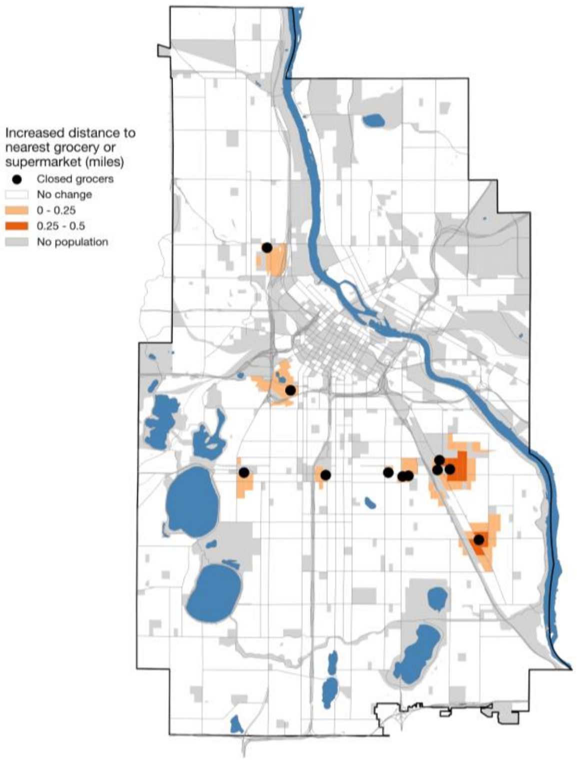
This analysis identified areas within 0.5 mile of recently closed or damaged stores that lack access to other nearby sources of supplies. Initial mapping is focusing on food access from “brick and mortar” food stores. Future work can extend the current analysis to additionally consider other basic needs such as toiletries and medications.

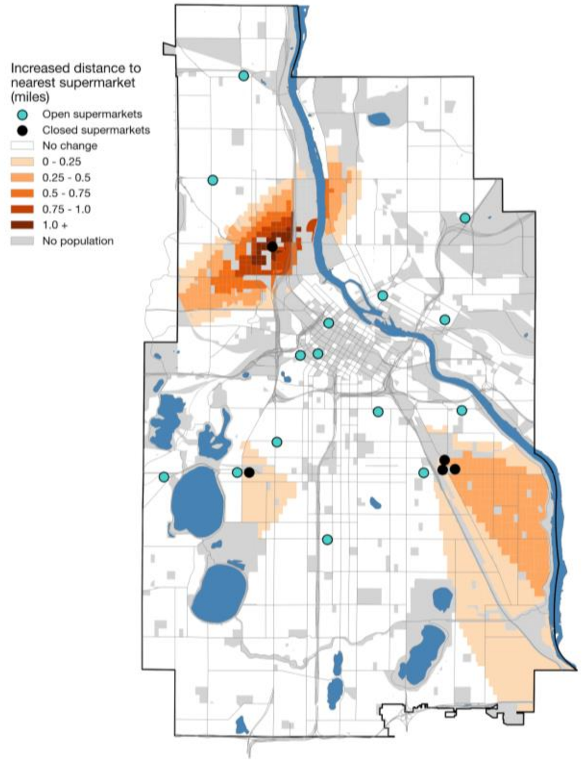
Store data: We identified the set of stores open prior to the riots using Minneapolis City’s list of stores licensed to sell food. We used data from an open-source website, Twin Cities Mutual Aid Project (https://tcmap.org/) to identify the stores from the list of Minneapolis City stores that are now closed.

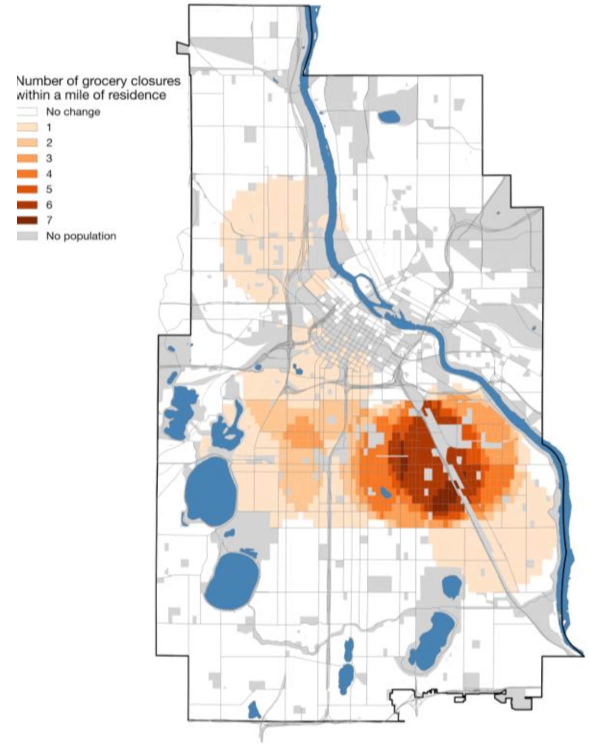
Demographic data: We used data from the 2010 Census to summarize the impacts on the population.

What is not included in this analysis: We included only those stores in the city’s licensed list for Minneapolis. This does not include farmers’ markets and food shelves. We did not include stores outside of Minneapolis. This exclusion could impact the assessment of store availability on the borders of the city (i.e., for people whose closest store is in another city/town). However, all of the closed stores in Minneapolis are greater than 1.0 away from the city border so the impact analysis would not be affected, and thus, our main conclusions are the same. We also did not include the impact of culturally-specific food source closures to different ethnic groups. Lastly, this analysis does not include the impact of non-food store closures.

Note: We cannot be sure that our list of store closures is complete, which means that our impact assessment is potentially underestimating the loss of food access. From what we understand of the damage, closures that we have not accounted for are likely to fall in the same areas as the ones we know about, but there may be areas of need that we have not identified. The process we used to create this data can be repeated once a comprehensive list of closures is complete.


Increased distance to nearest grocery or supermarket (miles)
Closed grocers
No change
0 - 0.25
0.25 - 0.5
No population


Increased distance to nearest supermarket (miles)
Open supermarkets
Closed supermarkets
No change
0 - 0.25
0.25 - 0.5
0.5 - 0.75
0.75 - 1.0
1.0 +
No population


Number of grocery closures within a mile of residence
No change
1
2
3
4
5
6
7
No population

Maplewood, MN

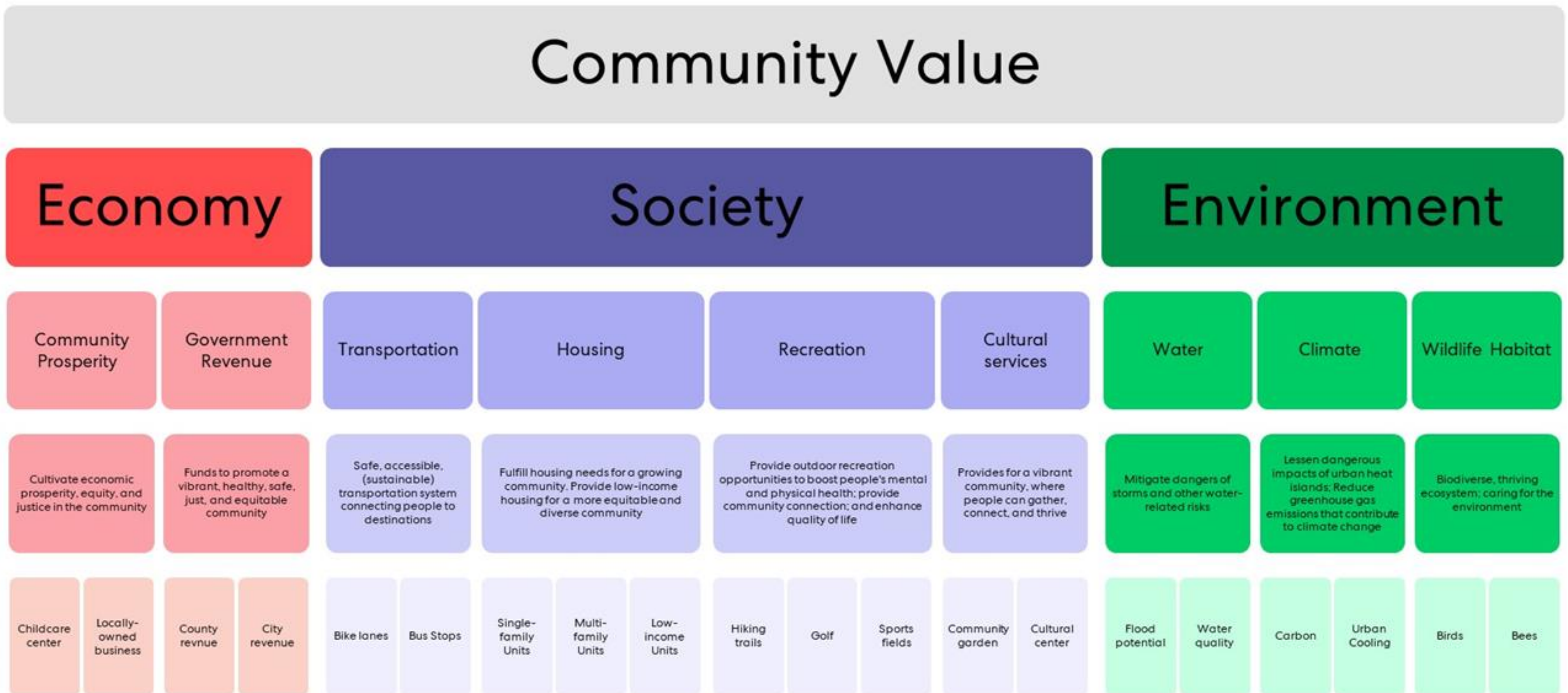
During the course of our project, Ramsey County (Minnesota) Commissioners decided to sell two parcels of county land within the city limits of Maplewood, a suburb of St. Paul. One parcel (Site A) is an undeveloped grassland while the other (Site B) is a 9-hole golf course called the Ponds at Battle Creek. This was a contentious issue from the beginning and continues to be so. The city of Maplewood controls zoning and does not want to sell the golf course but the county has argued the golf course is not making money. The debate over what the land should become has pitted one special-interest group after another against each other. The County and City agreed to hold a series of community meetings and hired Perkins and Will, an architecture firm, to facilitate the engagements. We reached out to Perkins and Will and learned that the mayor of Maplewood had also recommended that we provide input as they had heard about the work from a USGA publication. We worked with Perkins and Will to help construct the meetings and develop options that illustrated the potential values provided to the community from redevelopment as compared to the current land use patterns.

To effectively evaluate the impact of the Maplewood property redevelopment, it is critical to document the full costs and benefits that the properties currently or could provide the community. Throughout the community engagement meetings, individuals were given opportunities to express their concerns and desires for what benefits might be lost or gained from the development of the Maplewood properties. These expressions provide critical information as to what kinds of value the community currently or hopes to receive in the future from the sites. We used the feedback to develop an evaluation tool that would transparently evaluate the potential for each development scenario to address the multiple goals and objectives stated by community members.

Similar to our original community engagement meeting, we used a “values-hierarchy” approach to organize and describe the total value any scenario could provide based on the stakeholder comments. Rather than first debate what alternative development scenarios should be, it is important first determine what a community might want the redevelopment to provide. This value-focused thinking approach is described by Keeney (1992; 1996) who stated that “Alternatives are relevant only because they are means to achieve your values. Thus, your thinking should focus first on values and later on alternatives that might achieve them. Naturally there should be iteration between articulating values and creating alternatives, but the principle is 'values first'.” At the top of values hierarchy are a few fundamental (ultimate) objectives. As one moves down the hierarchy, the values describe how one would achieve those higher levels. Ultimately, some specific metrics could indicate how well one is doing to achieve those fundamental objectives.

For the Maplewood Properties, stakeholders highlighted the property’s ability to support the community through three fundamental pathways: the economy, society and the environment. Below, we lay out the details within each of these three objectives and describe the specific metrics that could indicate increased value for each of them. We used comments made by individuals or by government officials during the meetings to inform the selection of metrics. We then organized these insights into this value-focused approach for later scenario evaluation. The benefits of laying out and organizing the values is to provide an opportunity for feedback and reflection on the values that we think we have heard directly from the community, to make sure that those values are being included in the evaluation of the scenarios, and to show that there are multiple values and perspectives coming from the community.

Public comments indicated that these core values are ultimately shared by all members of the community. The disagreements we heard are likely about the priority or relative importance of each of these goals, so we first want to layout these goals and objectives without priorities debated and use them as aspirational goals. For example, all else being equal we do not feel that anyone wants to destroy wildlife habitat, or prevent anyone from having an affordable home to live in. A statement that someone likes golf does not mean they do not want some to have a place to live or vice-versa. Thus, we recommended laying out the full set of goals first and then see what can be done to achieve these goals collectively. We note that our golf project focused on environmental benefits (ecosystem services) but could fit into a broader decision context.



Community values hierarchy from Maplewood Community Engagement.

Using values to generate scenarios

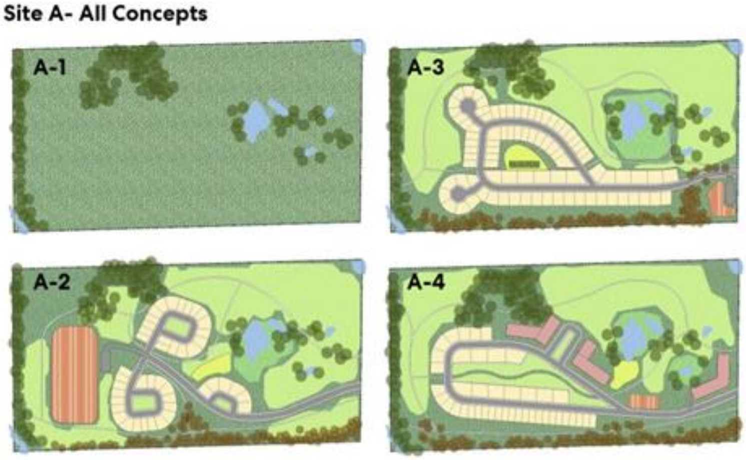
We used this values-focused framework to help generate the alternative scenarios for each of the redevelopment sites. However, there were six required attributes that all the scenarios were required to have related to the above broad goals and objectives:

1. Societal
    a. Some amount of publicly accessible open space via a trail network
    b. Some amount of community space for gardening/urban agriculture
    c. Improved access and connectivity to transit
    d. Sensitivity to neighbors
2. Environmental
    a. Preservation of some ecological sensitive areas and enhanced ecological systems (wetlands, grasslands, forested areas)
    b. An emphasis on enhanced stormwater management

Evaluating potential designs: We used the metrics and values above to explore how redevelopment scenarios of the two available spaces can each contribute to supporting community, city, and county interests. We used our judgment to determine the effect of each scenario on each metric. This is meant to reflect the intention of the design narratives. Since these are not actual designs, we thought it would be inappropriate to do too much quantitative analysis that would not reflect the intent of the design. These narrative designs are intended to achieve multiple goals, provide transparency, build trust in the design process and illustrate tradeoffs. We note that we did not include the revenue provided to Maplewood or Ramsey as this is simply outside of our team's expertise.

We considered four development scenarios of each site. The first scenario for each is the current land use and land cover with no building done. The next three scenarios show increasing intensity of building and provision for social and economic attributes.

Site A- All Concepts



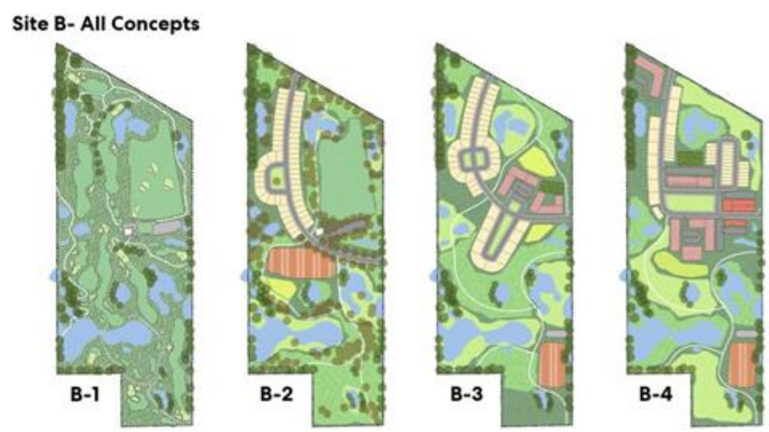
Site A scenarios' impacts

Scenario 1 (A1): The "No build" scenario will provide no social or economic value but contribute some environmental benefits for climate and wildlife habitat, particularly grassland habitat which is likely to support bees and birds.

Scenario 2 (A2): The “Cluster” scenario will add some clear value to society by providing housing in the form of single family homes and some subsidized, low-income units, creating recreation value through trails, park space, picnic and playgrounds, offering some transportation infrastructure with a bus-stop, parking and bike-lanes and generate cultural services with a community farm, public art space, an edible landscape of fruit trees and a community gathering space. The low intensity development will not include economic drivers and will still provide environmental benefits similar to the no build except that we envision the potential for renewable energy (solar) to be developed and some decrease in grassland habitat and thus a likely decrease in bird and bee habitat. The developed land will also reduce the summer urban cooling services provided to the neighboring homes.

Scenario 3 (A3): The “Neighborhood Enclave” scenario will expand its development intensity from A2. Key differences from A2: It will add more societal value by providing some units of all three kinds of housing and add ball fields to the recreation offered in A2. Instead of a community farm, community gardens will be provided. Transportation, economic and environmental metrics are similar to A2.

Scenario 4 (A4): The “Mixed Use Crossroads" scenario will increase development intensity from A3. While most of the societal values are the same, this scenario will have more multi-family and low-income housing provided. Most notably, this scenario would also contribute to economic goals by envisioning a job-training site, child care center and a café.



Site B scenarios’ impacts

Scenario 1 (B1): The “Golf” scenario will provide no other social value other than golf or economic value but contribute some environmental benefits for climate and wildlife habitat, particularly grassland habitat which is likely to support bees and birds. We do know from our flood modeling assessments that the golf course does provide flood mitigation service to adjacent homes during large rainfall events.

Scenario 2 (B2): The “Urban Reserve” scenario will lose the golf course but add some clear value to society. We envision having a driving range for golfer and then including housing in the form of single family homes and some subsidized, low-income units, creating recreation value through trails, park space, picnic and playgrounds, offering some transportation infrastructure with a bus-stop, parking and bike-lanes and generate cultural services with community garden opportunities, public art space, an edible landscape of fruit trees and a community gathering space. The low intensity development will not include economic drivers. The topography and soil prevent much of the parcel from being developed so it will still provide environmental benefits similar to the golf course except that we envision the potential for renewable energy (solar) to be developed. The developed land will also reduce the summer urban cooling services provided to the neighboring homes.

Scenario 3 (B3): The “Village” scenario will expand its development intensity from B2. Key differences from A2: The driving range is not included but it will add societal value by providing some units of all three kinds of housing (more multi-family and low-income), and add ball fields to the recreation offered in B2. Instead of community gardens, a community farm will be provided. Transportation, economic and environmental metrics are similar to B2.

Scenario 4 (B4): The “Neighborhood Center" scenario will increase development intensity from A3. While most of the societal values are the same, this scenario will have slightly more multi-family and low-income housing provided and a bit more parking. Most notably, this scenario would also contribute to economic goals by envisioning a job-training site, child care center and a café.

Summary Scorecard

We used a scorecard to show what each development scenario could provide for each metric and objective. We believe the scorecard allows the stakeholders to see the complexity of the decision without getting lost in the complexity of the analyses. For example, most of the metrics have different units of analysis: housing would be the number of units, trails are in linear feet or miles, community farms are in acres, etc. This exercise provides an opportunity to learn about the potential to achieve multiple goals or identify those that have tradeoffs.

Insights from scorecard: What seems clear from this exercise is that all there seems to be potential for adding many dimensions of value. The clearest tradeoff is between golf and many of the other metrics on site B. The current golf course does not provide any other societal value listed. The inclusion of site A in a decision context, however, could provide a way to contribute to societal and economic values. Combining scenario A4 with B1, for example, would allow for all kinds of housing, recreation, cultural services, transportation, and some economic well-being while allowing the golf course to remain.

Broad Goals	Specific Objective	A1	A2	A3	A4	B1	B2	B3	B4
Housing	Single family	○	◔	◑	◔	○	◑	◔	◔
	Multi Family	○	○	◔	●	○	○	●	●
	Low income/subsidized	○	◔	◑	●	○	◔	●	●
Recreation	Golf	○	○	○	○	●	◑	○	○
	Trails	○	●	●	●	○	◑	◑	◑
	Park space	○	◕	◑	◕	○	●	●	●
	Ball fields/courts	○	○	●	●	○	○	●	●
	Playground area	○	●	●	●	○	●	●	●
	Fishing (pond with access)	○	●	●	●	○	○	○	○
	Picnic /BBQ area	○	◑	◑	◑	○	◑	◑	●
Transportation	Bus stop(s)	○	●	●	●	○	●	●	●
	Parking	○	◔	◔	◕	○	◔	◕	●
	Bike lane connections	○	◔	◔	◔	○	◔	◔	◔
Cultural Services	Community farm	○	●	○	○	○	○	◕	◕
	Community garden	○	○	◔	◔	○	◕	○	○
	Historic preservation	○	○	○	○	○	○	○	○
	Public art space	○	●	●	●	○	●	●	●
	Edible landscape	○	●	●	●	○	●	●	●
	Community gathering space	○	●	●	●	○	●	●	●
Economic Well-being	Job training site	○	○	○	●	○	○	○	●
	Childcare center	○	○	○	●	○	○	○	●
	Locally-owned business	○	○	○	●	○	○	○	●
Water	Flood protection infrastructure	○	●	●	●	○	●	●	●
	Pesticide potential	◑	◑	◑	◑	◕	◑	◑	◑
	Nutrient Runoff	○	◑	◑	●	◑	◑	◑	◑
Climate	Climate Change Mitigation	●	◔	◑	◑	◑	◔	◑	◑
	Renewable Energy	○	●	●	●	○	●	●	●
	Urban cooling	●	◑	◑	◑	●	◑	◔	◑
Wildlife Habitat	Bird habitat	◕	◑	◑	◑	◑	◑	◔	◑
	Bees	◕	◑	◑	◑	◑	◑	◔	◑
	Forest	◑	◑	◑	◑	◑	◑	◔	◑
	Grassland	●	◔	◑	◑	◔	◑	◔	◑
	Aquatic	◑	◑	◑	◑	◔	◔	◔	◔

What's not captured in the assessment?

We identified two important aspects of the decision context that have not been reflected in the discussions; decisions on other parcels in Ramsey County and the values and services that could be provided elsewhere. It seems that evaluation of the parcels in Maplewood are part of a broader set of linked decisions within Ramsey County and the City of Maplewood. The revenue from the sale of the properties could be used to support some of the broader Ramsey County goals, so including what could be done with the sale could provide some clarity to stakeholders. They discussed other properties and locations.

The assessment also focused only on what is being provided by the two properties and did not put that in the context of what is currently available to different stakeholders who likely perceive value at different scales. Similar to the first point about what other parcels are available, the full set of values and how the scale of assessment could differ between the county, city, the neighbors and beyond (golfers) would provide a truer sense of the decision that could help build trust about what motivated the initial decision to sell the property and also what creative opportunities could be explored to achieve the multiple goals of all stakeholders. Currently, this motivation is hidden and the resulting distrust limits constructive discussion by not allowing each stakeholder to look at the problem rationally.

Warren, MN

Project overview: We collaborated with the University of Minnesota Design School’s “Design 4 Community Regeneration” (D4CR) pilot project in Warren, MN by engaging with community Point-of-View teams to reimagine the city of Warren, MN as sustainable and environmentally regenerative. Through bi-weekly community engagement meetings over 8 months, residents expressed their desires for community development, green infrastructure for ecosystem services, and a resilient future. Ultimately, residents compiled a “menu” of over 100 different interventions and a scoring system to evaluate alternative options. The project culminated in a game-style negotiations platform whereby teams would choose future visions for the city--for example, a “Golf Village” or “Regenerative School Campus”--and score these visions according to their ecological and social benefits.

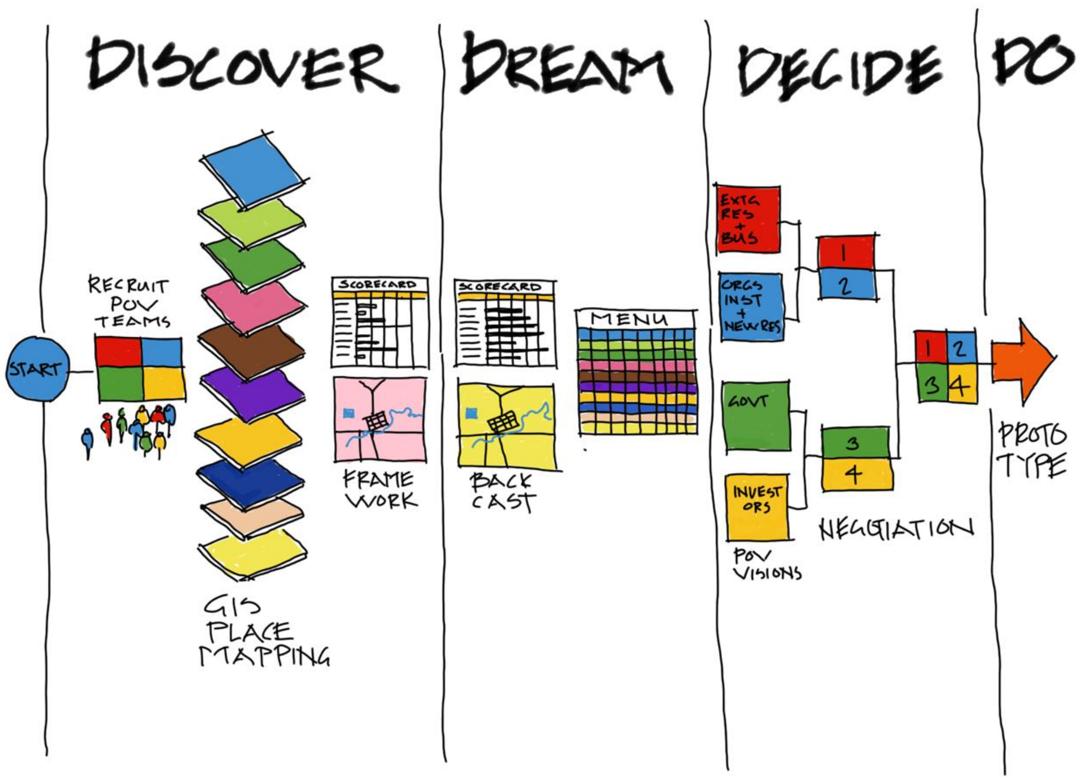


Image: Design 4 Community Regeneration pilot project process in four phases

Our involvement: We used Urban Invest models, our experience from the Maplewood project, and expert judgement to develop ecosystem service ‘scores’ for each of the potential “menu items” of community projects. This ecological score was then combined with economic scores identified by city officials, and cultural value scores determined by community representatives.

During the negotiation process, each Point-of-View team chose several projects to implement, and were able to see the potential ecological, economic, and cultural 'scores' of their choices, giving them a real-time sense of tradeoffs through the scorecard and an associated GIS mapping tool.

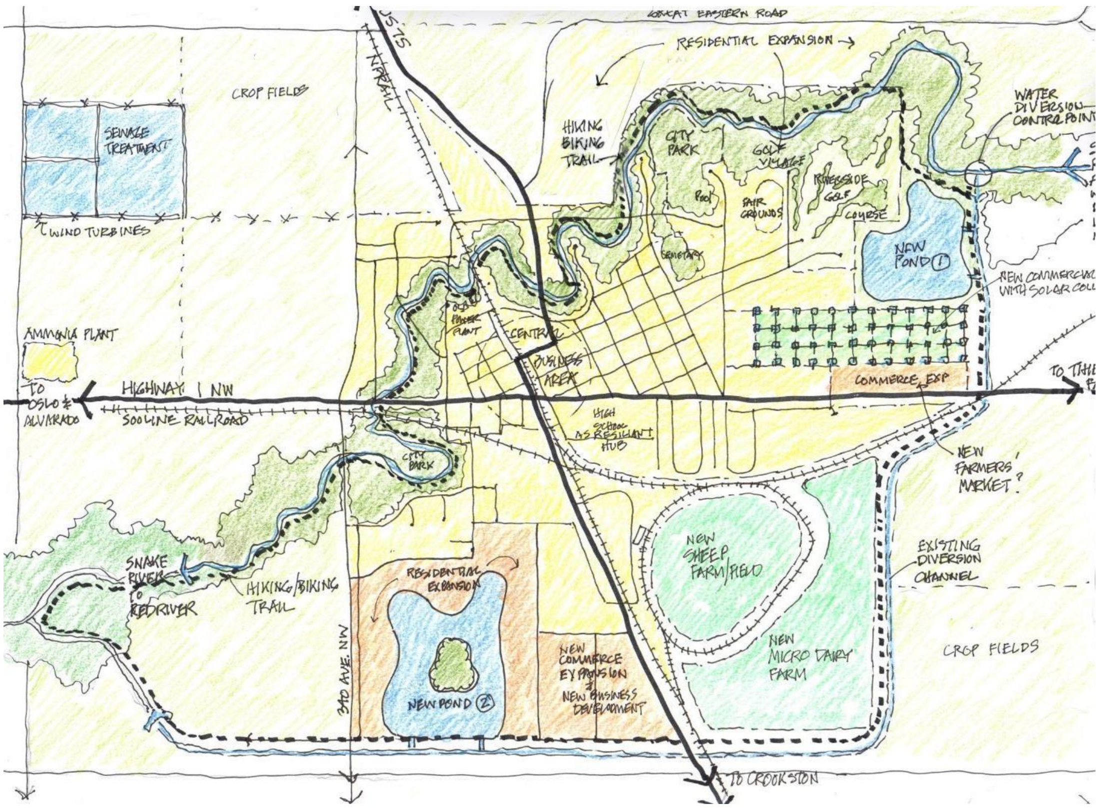
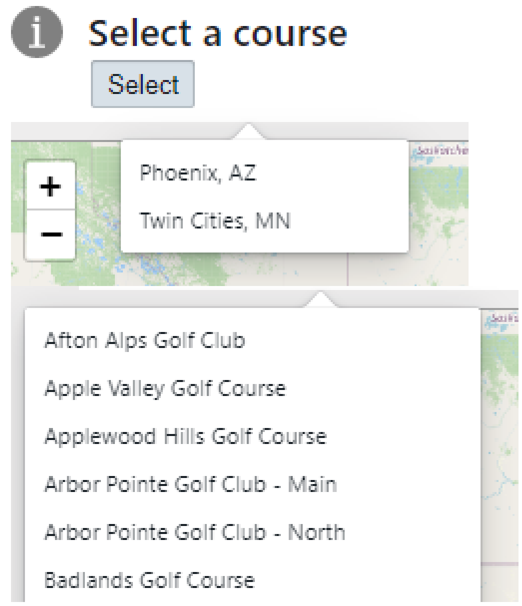


Image: sample Warren future from workshop, including redesigned "Golf Village"

Insights from Design 4 Community Regeneration: The pilot project (successfully receiving another round of funding) shows the need for people to understand the varied impacts and tradeoffs of their decisions as well as people's desire to incorporate ecosystem services into their values and visions for the future. However, we also learned that the scale of urban planning decisions, such as the ones Warren community members were making, are often smaller than our current models can show benefits for. We often had to 'de-construct' our models in order to assign scores. For example, we built the carbon model by inputting the known carbon sequestration of various plants (such as trees and grass) and had to revert back to these parameterizations to develop the scoring system for the project. This demonstrates a further need to ensure that models capture not only the scale of ecosystem services, but also are appropriate to the scale of decision-making when considering re-developing parcels or integrating green infrastructure projects with existing golf courses.

Interactive Tool

Inspired by our engagements that emphasized the need for communication and a broader view of sustainability beyond the ecosystem service assessments, we developed an interactive tool (https://phawthorne.github.io/single-parcel-es-tool/) that facilitates viewing the results of our work as a companion to the report. This tool is intended to help visualize the influence of different land use scenarios on a few environmental and social components of sustainability. We assessed several broad categories of factors either with judgment or through modeling efforts in this project. Climate, stormwater, wildlife habitat all relate to the environmental component of sustainability, while recreation, and cultural services connect to social sustainability. Within each category, several specific ecosystem services were assessed, such as pollinator abundance or urban heat island, as a function of the existing land use (golf) and several common urban land uses: city park, natural (undeveloped), high-density residential, low-density residential, and industrial.



We have loaded the results of our analysis of each course in two of the cities from our work: Twin Cities and Phoenix. In the example to the right, the Twin Cities was selected and an alphabetical list of all the courses is shown.

Once the course is selected, e.g. Les Bolstad (the UMN course), the tool zooms to that location (Figure 14a). The user can then select from one of the six scenarios. Once a scenario is selected, e.g. Golf, a clip of the National Land Cover Database map for the location will appear, as will the qualitative results of our "Ecosystem Services Report" (Figure 14b). The tool allows a user to view spatial patterns of ecosystem services for each scenario and see how those patterns change from golf (Figure 14c) to industrial (figure 14d), for example. The "other indicators" portion of the ES Report is meant to encourage a user to think about the other community values. We consider this a first prototype and plan to continue building off of it.

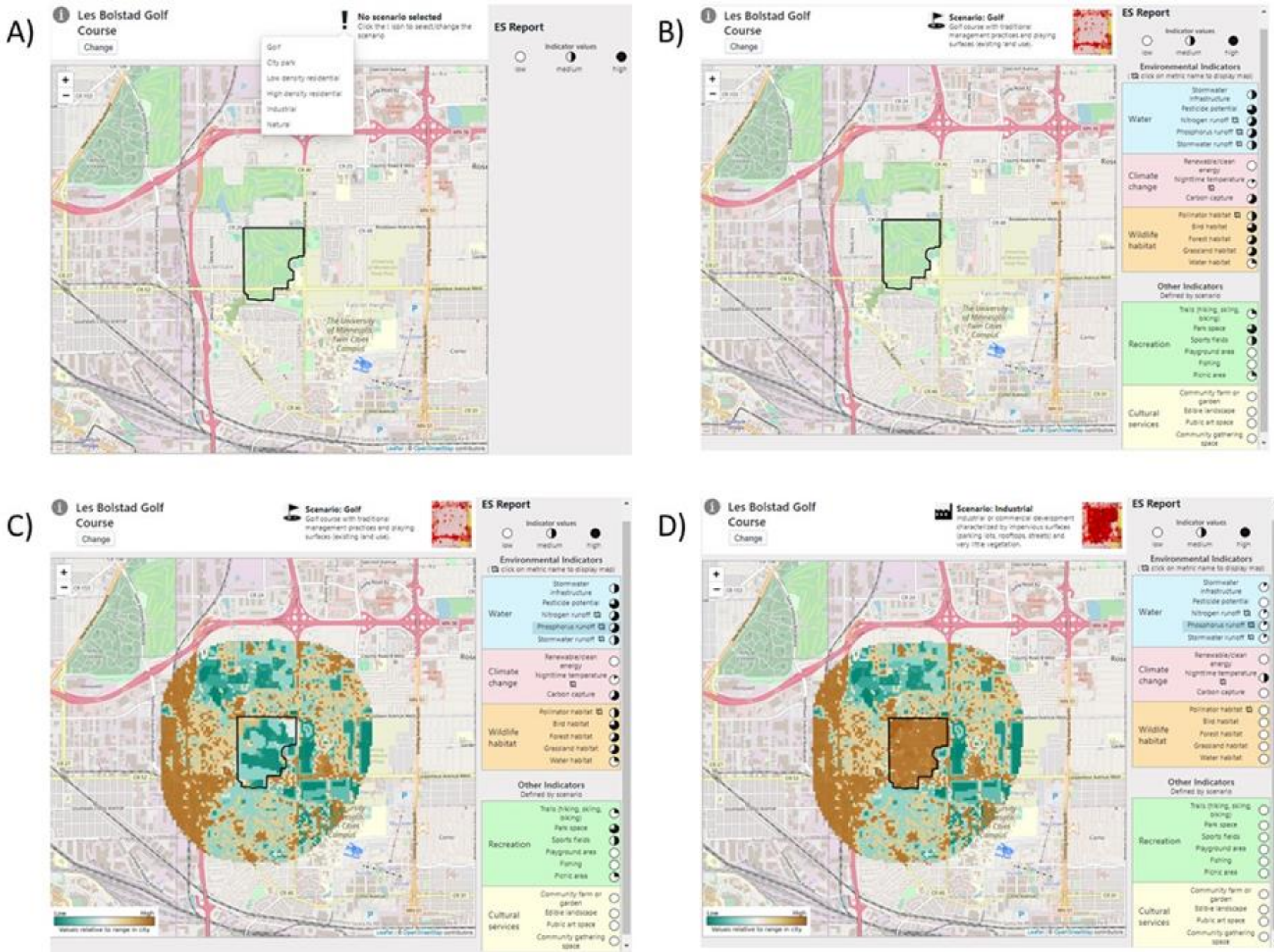


Figure 14. Example function of the interactive tool, showing (a) course selection, (b) scenario selection, and (c, d) ecosystem service mapping.

Conclusions

The US is in the midst of an urban planning debate between golf courses and affordable housing. Nationwide, it's estimated that there was a low-income housing shortage of 7 million units as of 2019 (National Low Income Housing Coalition). Intense discussions continue to play out in metro areas around the country around the remaining developable spaces - largely golf courses. In 2019, estimates of low-income housing availability in California suggest that there were 22 affordable rental units for every 100 low-income families, nearly the lowest in the nation. On February 12th, 2021 California state senator Christina Garcia introduced bill AB 672 which would "require a city, county, or city and county to rezone...certain sites used as a golf course to also allow for residential and open space use." Indeed, there are over 14,200 golf courses in the United States, and at approximately 160 acres per golf course, they collectively represent just over 2.27M acres of land (Environmental Institute for Golf, 2017). Fifty to 125 high density multi-family developments can be placed in a single acre and thus a golf course could potentially provide 8,000-20,000 units per course. Nationwide this would be an area equivalent to 110+ million potential housing units -10x more than enough to address the current housing deficit. But are these the only two options for golf course use? What is the best future for golf courses in the United States? What is or could be their total value to society?

We see an opportunity for the golf industry to take a lead in defining what a community-value centered golf course designed to address 21st Century sustainability challenges would like. At the moment, the general perception of golf courses is that they are unsustainable economically as public courses may be losing money, socially as they cater to the wealthy and take up space that could be used to address social issues, and ecologically, for their intense management. A deliberate strategy to address each of these three perceptions could show that the golf industry could lead in how to pivot towards a sustainable future where golf experiences are preserved and golf courses can be solutions. For example, in the USGA Distance Report, the relationships between distance and platform are discussed in a way that could essentially enhance golfing experiences that take up less space using variable-distance golf balls and creative design. There is an opportunity to illustrate how an urban golf course could be redesigned to address societal challenges like affordable housing using or increased access to parks *and* still have a multi-hole golf course.

Sustainability and resilience challenges in urban areas are likely to grow, and while our work clearly shows that golf courses are better than developed areas in the environmental services they provide, our engagement experiences suggest that this is but one part of the problem that all green spaces face. A future where golf courses remain single function, half-day getaways from the stress of urban life seems unlikely. Embracing golf-courses as inclusive, multi-function spaces that serve as broad a segment of society as possible and address as many components of sustainable urban design seems practical and critical if golf is to thrive.

References

Amorim Maia, A. T., F. Calcagni, J. J. T. Connolly, I. Anguelovski, and J. Langemeyer. 2020. Hidden drivers of social injustice: uncovering unequal cultural ecosystem services behind green gentrification. Environmental Science & Policy 112:254–263.

Arıoğlu Akan, M. Ö., D. G. Dhavale, and J. Sarkis. 2017. Greenhouse gas emissions in the construction industry: An analysis and evaluation of a concrete supply chain. Journal of Cleaner Production 167:1195–1207.

Bae, J., and Y. Ryu. 2015. Land use and land cover changes explain spatial and temporal variations of the soil organic carbon stocks in a constructed urban park. Landscape and Urban Planning 136:57–67.

Baró, F., Palomo, I., Zulian, G., Vizcaino, P., Haase, D., & Gómez-Baggethun, E. (2016). Mapping ecosystem service capacity, flow and demand for landscape and urban planning: A case study in the Barcelona metropolitan region. Land Use Policy, 57, 405–417. Retrieved from https://doi.org/10.1016/j.landusepol.2016.06.006

Bierman, P. M., Horgan, B. P., Rosen, C. J., Hollman, A. B., & Pagliari, P. H. (2010). Phosphorus runoff from turfgrass as affected by phosphorus fertilization and clipping management. Journal of Environmental Quality, 39(1), 282–292.

Boyle, C. A., and L. Lavkulich. 1997. Carbon Pool Dynamics in the Lower Fraser Basin from 1827 to 1990. Environmental Management 21:443–455.

Brauer, J. (2009, April 22). Design Concepts:Debating Fairway Widths. Retrieved July 30, 2020, from Golf Course Industry website: https://www.golfcourseindustry.com/article/design-concepts-debating-fairway-widths/

Chakraborty, T., and X. Lee. 2019. A simplified urban-extent algorithm to characterize surface urban heat islands on a global scale and examine vegetation control on their spatiotemporal variability. International Journal of Applied Earth Observation and Geoinformation 74:269–280.

Chaparro, L., and J. Terradas. 2010. Ecosystem services of urban forest.

Churkina, G., D. Brown, and G. KEOLEIAN. 2010. Carbon stored in human settlements: The conterminous United States. Global Change Biology 16:135–143.

Colding, J., & Folke, C. (2009). The role of golf courses in biodiversity conservation and ecosystem management. Ecosystems, 12(2), 191–206.

Cortinovis, C., & Geneletti, D. (2018). Ecosystem services in urban plans: What is there, and what is still needed for better decisions. Land Use Policy, 70, 298–312.

Daily, G. C. (1997). Nature’s services. Island Press, Washington, DC.

Davies, Z. G., J. L. Edmondson, A. Heinemeyer, J. R. Leake, and K. J. Gaston. 2011. Mapping an urban ecosystem service: quantifying above-ground carbon storage at a city-wide scale: Urban above-ground carbon storage. Journal of Applied Ecology 48:1125–1134.

Davis, A. Y., Lonsdorf, E. V., Shierk, C. R., Matteson, K. C., Taylor, J. R., Lovell, S. T., & Minor, E. S. (2017). Enhancing pollination supply in an urban ecosystem through landscape modifications. Landscape and Urban Planning, 162, 157–166. https://doi.org/10.1016/j.landurbplan.2017.02.011

Delegard, K., and K. Ehrman-Solberg. 2017. ‘Playground of the People’? Mapping Racial Covenants in Twentieth-century Minneapolis. Open Rivers: Rethinking The Mississippi.

Derkzen, M. L., van Teeffelen, A. J., & Verburg, P. H. (2015). Quantifying urban ecosystem services based on high-resolution data of urban green space: An assessment for Rotterdam, the Netherlands. Journal of Applied Ecology, 52(4), 1020–1032.

Dewitz, J. 2021. National Land Cover Database (NLCD) 2019 Products. U.S. Geological Survey.

Easton, Z. M., & Petrovic, A. M. (2004). Fertilizer source effect on ground and surface water quality in drainage from turfgrass. Journal of Environmental Quality, 33(2), 645–655.

ECOSOC, U. (2019). Special Edition: Progress towards the Sustainable Development Goals Report of the Secretary-General. Advanced Unedited Version. New York (US): United Nations.

Edmondson, J. L., Z. G. Davies, N. McHugh, K. J. Gaston, and J. R. Leake. 2012. Organic carbon hidden in urban ecosystems. Scientific Reports 2:963.

Elmqvist, T., Andersson, E., Frantzeskaki, N., McPhearson, T., Olsson, P., Gaffney, O., … Folke, C. (2019). Sustainability and resilience for transformation in the urban century. Nature Sustainability, 2(4), 267–273. https://doi.org/10.1038/s41893-019-0250-1

Environmental Institute for Golf. (2017). Land Use Characteristics and Environmental Stewardship Programs on U.S. Golf Courses. Golf Course Superintendents Association of America. Retrieved from https://www.gcsaa.org/docs/default-source/Environment/phase-2-land-use-survey-full-report.pdf?sfvrsn=c750ea3e_2

Escobedo, F., S. Varela, M. Zhao, J. E. Wagner, and W. Zipperer. 2010. Analyzing the efficacy of subtropical urban forests in offsetting carbon emissions from cities. Environmental Science & Policy 13:362–372.

Fissore, C., L. A. Baker, S. E. Hobbie, J. Y. King, J. P. McFadden, K. C. Nelson, and I. Jakobsdottir. 2011. Carbon, nitrogen, and phosphorus fluxes in household ecosystems in the Minneapolis-Saint Paul, Minnesota, urban region. Ecological Applications 21:619–639.

Gallo, T., Fidino, M., Lehrer, E. W., & Magle, S. B. (2017). Mammal diversity and metacommunity dynamics in urban green spaces: Implications for urban wildlife conservation. Ecological Applications, 27(8), 2330–2341. https://doi.org/10.1002/eap.1611

Geneletti, D., Adem Esmail, B., Cortinovis, C., Arany, I., Balzan, M., van Beukering, P., … Broekx, S. (2020). Ecosystem services mapping and assessment for policy-and decision-making: Lessons learned from a comparative analysis of European case studies. One Ecosystem, 5, e53111.

Ghimire, B., Chen, A.S., Guidolin, M., Keedwell, E.C., Djordjevic, S., and Savic, D.A. (2013). Formulation of a fast 2D urban pluvial flood model using a cellular automata approach. Journal of Hydroinformatics, 15(3), 676–686.

Goldstein, B., D. Gounaridis, and J. P. Newell. 2020. The carbon footprint of household energy use in the United States. Proceedings of the National Academy of Sciences 117:19122–19130.

Golf Course Superintendents Association of America. (2016). Nutrient Use and Management Practices on U.S. Golf Courses (Vol. II). Retrieved from https://www.gcsaa.org/environment/golf-course-environmental-profile

Golubiewski, N. E. 2006. Urbanization Increases Grassland Carbon Pools: Effects Of Landscaping In Colorado's Front Range. Ecological Applications 16:555–571.

Gómez-Baggethun, E., & Barton, D. N. (2013). Classifying and valuing ecosystem services for urban planning. Ecological Economics, 86, 235–245.

Gregory, R., & Keeney, R. L. (1994). Creating policy alternatives using stakeholder values. Management Science, 40(8), 1035–1048.

Guerry, A. D., Polasky, S., Lubchenco, J., Chaplin-Kramer, R., Daily, G. C., Griffin, R., … Elmqvist, T. (2015). Natural capital and ecosystem services informing decisions: From

promise to practice. Proceedings of the National Academy of Sciences, 112(24), 7348–7355.
Guo, Y., A. Gasparrini, B. Armstrong, S. Li, B. Tawatsupa, A. Tobias, E. Lavigne, M. de Sousa Zanotti Stagliorio Coelho, M. Leone, X. Pan, S. Tong, L. Tian, H. Kim, M. Hashizume, Y. Honda, Y.-L. L. Guo, C.-F. Wu, K. Punnasiri, S.-M. Yi, P. Michelozzi, P. H. N. Saldiva, and G. Williams. 2014. Global Variation in the Effects of Ambient Temperature on Mortality: A Systematic Evaluation. Epidemiology 25:781–789.
Haase, D., Larondelle, N., Andersson, E., Artmann, M., Borgström, S., Breuste, J., … Hansen, R. (2014). A quantitative review of urban ecosystem service assessments: Concepts, models, and implementation. Ambio, 43(4), 413–433.
Haith, D. A., & Duffany, M. W. (2007). Pesticide runoff loads from lawns and golf courses. Journal of Environmental Engineering, 133(4), 435–446. Retrieved from https://doi.org/10.1061/(ASCE)0733-9372(2007)133:4(435)
Hamel, P., A. D. Guerry, S. Polasky, B. Han, J. A. Douglass, M. Hamann, B. Janke, J. J. Kuiper, H. Levrel, H. Liu, E. Lonsdorf, R. I. McDonald, C. Nootenboom, Z. Ouyang, R. P. Remme, R. P. Sharp, L. Tardieu, V. Viguié, D. Xu, H. Zheng, and G. C. Daily. 2021. Mapping the benefits of nature in cities with the InVEST software. npj Urban Sustainability 1:25.
Hobbie, S. E., Finlay, J. C., Janke, B. D., Nidzgorski, D. A., Millet, D. B., & Baker, L. A. (2017). Contrasting nitrogen and phosphorus budgets in urban watersheds and implications for managing urban water pollution. Proceedings of the National Academy of Sciences, 114(16), 4177–4182.
Hoffman, J. S., V. Shandas, and N. Pendleton. 2020. The Effects of Historical Housing Policies on Resident Exposure to Intra-Urban Heat: A Study of 108 US Urban Areas. Climate 8:12.
Hueber, D., & Worzala, E. (2010). “Code Blue” for US Golf Course Real Estate Development:“Code Green” for Sustainable Golf Course Redevelopment. Journal of Sustainable Real Estate, 1–41.
Hutyra, L. R., B. Yoon, and M. Alberti. 2011. Terrestrial carbon stocks across a gradient of urbanization: a study of the Seattle, WA region: URBAN TERRESTRIAL CARBON STOCKS. Global Change Biology 17:783–797.
Ingram, M. A., Hoke, L., & Meyer, J. (2013). The declining economic viability of municipal golf courses. Public and Municipal Finance, 2(1), 46–55. Retrieved from https://businessperspectives.org/pdfproxy.php?item_id:5202
Janke, B. D., Finlay, J. C., & Hobbie, S. E. (2017). Trees and streets as drivers of urban stormwater nutrient pollution. Environmental Science & Technology, 51(17), 9569–9579.
Jo, H. 2002. Impacts of urban greenspace on offsetting carbon emissions for middle Korea. Journal of Environmental Management 64:115–126.
Kaye, J. P., R. L. McCulley, and I. C. Burke. 2005. Carbon fluxes, nitrogen cycling, and soil microbial communities in adjacent urban, native and agricultural ecosystems. Global Change Biology 11:575–587.
Keeler, B. L., Hamel, Perrine, McPhearson, Timon, Hamann, Maike H, Donahue, Marie L, Meza Prado, Kelly A, Arkema, Katie K, Bratman, Gregory N, Brauman, Kate A, Finlay, Jacques C, Guerry, Anne D, Hobbie, Sarah E, Johnson, Justin A, MacDonald, Graham K, McDonald, Robert I, Neverisky, Nick, and Wood, Spencer A. 2019. Social-ecological and technological factors moderate the value of urban nature. Nature Sustainability 2:29–38.
Keeney, R. L. (1992). Value-Focused Thinking: A Path to Creative Decisionmaking. Harvard University Press.

Kellett, R., A. Christen, N. C. Coops, M. van der Laan, B. Crawford, T. R. Tooke, and I. Olchovski. 2013. A systems approach to carbon cycling and emissions modeling at an urban neighborhood scale. Landscape and Urban Planning 110:48–58.

King, K. W., & Balogh, J. C. (2011). Stream water nutrient enrichment in a mixed-use watershed. Journal of Environmental Monitoring, 13(3), 721–731.

King, K. W., Balogh, J. C., & Harmel, R. D. (2007). Nutrient flux in storm water runoff and baseflow from managed turf. Environmental Pollution, 150(3), 321–328.

Koh, I., Lonsdorf, E. V., Williams, N. M., Brittain, C., Isaacs, R., Gibbs, J., & Ricketts, T. H. (2016). Modeling the status, trends, and impacts of wild bee abundance in the United States. Proceedings of the National Academy of Sciences, 113(1), 140–145. Retrieved from http://www.pnas.org/content/113/1/140.short

Kuittinen, M., C. Moinel, and K. Adalgeirsdottir. 2016. Carbon sequestration through urban ecosystem services: A case study from Finland. Science of The Total Environment 563–564:623–632.

Liotta, C., Y. Kervinio, H. Levrel, and L. Tardieu. 2020. Planning for environmental justice - reducing well-being inequalities through urban greening. Environmental Science & Policy 112:47–60.

Lonsdorf, E. V., C. Nootenboom, B. Janke, and B. P. Horgan. 2021. Assessing urban ecosystem services provided by green infrastructure: Golf courses in the Minneapolis-St. Paul metro area. Landscape and Urban Planning 208:104022.

Lonsdorf, E., Ricketts, T., Kremen, C., Winfree, R., Greenleaf, S., & Williams, N. (2011). Crop pollination services. In Natural capital. Theory and practice of mapping ecosystem services. (pp. 168–187). Oxford, UK: Oxford University Press. Retrieved from http://www.researchgate.net/profile/Rachael_Winfree/publication/230822625_Crop_pollination_services/links/09e415050bead162e5000000.pdf

Luo, S. H., Q. Z. Mao, K. M. Ma, and J. Wu. 2014. Spatial distribution of soil carbon and nitrogen in urban greenspace of Beijing. Shengtai Xuebao/ Acta Ecologica Sinica 34:6011–6019.

Maantay, J. (2002). Mapping environmental injustices: Pitfalls and potential of geographic information systems in assessing environmental health and equity. Environmental Health Perspectives, 110(suppl 2), 161–171. Retrieved from https://doi.org/10.1289/ehp.02110s2161

McDermott, M., Mahanty, S., & Schreckenberg, K. (2013). Examining equity: A multidimensional framework for assessing equity in payments for ecosystem services. Environmental Science & Policy, 33, 416–427. https://doi.org/10.1016/j.envsci.2012.10.006

McDonald, R. I., T. Biswas, C. Sachar, I. Housman, T. M. Boucher, D. Balk, D. Nowak, E. Spotswood, C. K. Stanley, and S. Leyk. 2021. The tree cover and temperature disparity in US urbanized areas: Quantifying the association with income across 5,723 communities. PLOS ONE 16:e0249715.

McPherson, E. G., Q. Xiao, and E. Aguaron. 2013. A new approach to quantify and map carbon stored, sequestered and emissions avoided by urban forests. Landscape and Urban Planning 120:70–84.

Merrick, J. R., Parnell, G. S., Barnett, J., & Garcia, M. (2005). A multiple-objective decision analysis of stakeholder values to identify watershed improvement needs. Decision Analysis, 2(1), 44–57. Retrieved from http://pubsonline.informs.org/doi/abs/10.1287/deca.1050.0033

Mohajerani, A., Bakaric, J., & Jeffrey-Bailey, T. (2017). The urban heat island effect, its causes, and mitigation, with reference to the thermal properties of asphalt concrete. Journal of Environmental Management, 197, 522–538.

Nelson, E., Mendoza, G., Regetz, J., Polasky, S., Tallis, H., Cameron, Dr., … Shaw, Mr. (2009). Modeling multiple ecosystem services, biodiversity conservation, commodity production,

and tradeoffs at landscape scales. Frontiers in Ecology and the Environment, 7(1), 4–11. https://doi.org/10.1890/080023
Nero, B. F., D. Callo-Concha, A. Anning, and M. Denich. 2017. Urban Green Spaces Enhance Climate Change Mitigation in Cities of the Global South: The Case of Kumasi, Ghana. Procedia Engineering 198:69–83.
Nesbitt, L., Meitner, M. J., Girling, C., Sheppard, S. R., & Lu, Y. (2019). Who has access to urban vegetation? A spatial analysis of distributional green equity in 10 US cities. Landscape and Urban Planning, 181, 51–79. Retrieved from https://doi.org/10.1016/j.landurbplan.2018.08.007
NOAA. 2021. NOWData: Monthly summarized data, 1981-2010, mean of monthly average temperatures. National Weather Service Forecast Office.
Nor, A. N. M., Corstanje, R., Harris, J. A., & Brewer, T. (2017). Impact of rapid urban expansion on green space structure. Ecological Indicators, 81, 274–284.
Norman, J., H. L. MacLean, and C. A. Kennedy. 2006. Comparing High and Low Residential Density: Life-Cycle Analysis of Energy Use and Greenhouse Gas Emissions. Journal of Urban Planning and Development 132:10–21.
Nowak, D. J. 1993. Atmospheric Carbon Reduction by Urban Trees. Journal of Environmental Management 37:207–217.
Nowak, D. J., and D. E. Crane. 2002. Carbon storage and sequestration by urban trees in the USA. Environmental Pollution 116:381–389.
Nowak, D. J., E. J. Greenfield, R. E. Hoehn, and E. Lapoint. 2013. Carbon storage and sequestration by trees in urban and community areas of the United States. Environmental Pollution 178:229–236.
Oke, T. R. (2006). Instruments and Observing Methods Report No. 8 (No. 81; p. 47). WORLD METEOROLOGICAL ORGANIZATION.
OpenStreetMap contributors. 2021. Planet dump retrieved from https://planet.osm.org.
Pascual, U., Balvanera, P., Díaz, S., Pataki, G., Roth, E., Stenseke, M., … Kelemen, E. (2017). Valuing nature’s contributions to people: The IPBES approach. Current Opinion in Environmental Sustainability, 26, 7–16.
Pistrika, A., Tsakiris, G., and Nalbantis, I. (2014). Flood Depth-Damage Functions for Built Environment. Environ Process 1:553–572. https://doi.org/10.1007/s40710-014-0038-2
Polasky, S., Nelson, E., Camm, J., Csuti, B., Fackler, P., Lonsdorf, E., … others. (2008). Where to put things? Spatial land management to sustain biodiversity and economic returns. Biological Conservation, 141(6), 1505–1524. Retrieved from http://www.sciencedirect.com/science/article/pii/S0006320708001213
Pouyat, R. V., I. D. Yesilonis, and D. J. Nowak. 2006. Carbon Storage by Urban Soils in the United States. Journal of Environmental Quality 35:1566–1575.
Raciti, S. M., L. R. Hutyra, and A. C. Finzi. 2012a. Depleted soil carbon and nitrogen pools beneath impervious surfaces. Environmental Pollution 164:248–251.
Raciti, S. M., L. R. Hutyra, P. Rao, and A. C. Finzi. 2012b. Inconsistent definitions of “urban” result in different conclusions about the size of urban carbon and nitrogen stocks. Ecological Applications 22:1015–1035.
Rice, P. J., & Horgan, B. P. (2011). Nutrient loss with runoff from fairway turf: An evaluation of core cultivation practices and their environmental impact. Environmental Toxicology and Chemistry, 30(11), 2473–2480.
Rice, P. J., & Horgan, B. P. (2013). Evaluation of nitrogen and phosphorus transport with runoff from fairway turf managed with hollow tine core cultivation and verticutting. Science of the Total Environment, 456, 61–68.
Ricketts, T. H., Regetz, J., Steffan-Dewenter, I., Cunningham, S. A., Kremen, C., Bogdanski, A., … others. (2008). Landscape effects on crop pollination services: Are there general patterns? Ecology Letters, 11(5), 499–515.

Roberto, E. 2016. The Divergence Index: A Decomposable Measure of Segregation and Inequality. arXiv:1508.01167 [physics, stat].
Rossman, L. A. (2010). Storm water management model user's manual, version 5.0. National Risk Management Research Laboratory, Office of Research and ….
Rossman, L.A. and Huber, W.C. (2016). Storm Water Management Model Reference Manual Volume I – Hydrology (Revised). EPA/600/R-15/162A, U.S. Environmental Protection Agency, Cincinnati, OH.
Schatz, J., & Kucharik, C. J. (2014). Seasonality of the urban heat island effect in Madison, Wisconsin. Journal of Applied Meteorology and Climatology, 53(10), 2371–2386.
Selbig, W. R. (2016). Evaluation of leaf removal as a means to reduce nutrient concentrations and loads in urban stormwater. Science of the Total Environment, 571, 124–133.
Sharp, R., Douglass, J., Wolny, S., Arkema, K. K., Bernhardt, J., Bierbower, W., … Wyatt, K. (2020). InVEST 3.8.7 User's Guide. The Natural Capital Project. Retrieved from http://data.naturalcapitalproject.org/nightly-build/invest-users-guide/html/#
Sharp, R., J. Douglass, S. Wolny, K. K. Arkema, J. R. Bernhardt, W. Bierbower, N. Chaumont, D. Denu, D. Fisher, K. Glowinski, R. Griffin, G. Guannel, A. D. Guerry, J. Johnson, P. Hamel, C. Kennedy, C. K. Kim, M. Lacayo, E. Lonsdorf, L. Mandle, L. Rogers, J. M. Silver, J. Toft, G. Verutes, A. L. Vogl, S. A. Wood, and K. Wyatt. 2021. InVEST 3.9.0 User's Guide. The Natural Capital Project, Stanford University, University of Minnesota, The Nature Conservancy, and World Wildlife Fund.
Shuman, L. M. (2002). Phosphorus and nitrate nitrogen in runoff following fertilizer application to turfgrass. Journal of Environmental Quality, 31(5), 1710–1715.
Smoliak, B. V., Snyder, P. K., Twine, T. E., Mykleby, P. M., & Hertel, W. F. (2015). Dense network observations of the twin cities canopy-layer urban heat island. Journal of Applied Meteorology and Climatology, 54(9), 1899–1917.
Soldat, D. J., Petrovic, A. M., & van Es, H. M. (2009). The effects of soil phosphorus and nitrogen and phosphorus fertilization on phosphorus runoff losses from turfgrass. In M. T. Nett, M. J. Carroll, B. P. Horgan, & M. Petrovic (Eds.), The Fate of Nutrients and Pesticides in the Urban Environment (pp. 93–106). American Chemical Society. Retrieved from https://doi.org/10.1021/bk-2008-0997.ch006
Strohbach, M. W., and D. Haase. 2012. Above-ground carbon storage by urban trees in Leipzig, Germany: Analysis of patterns in a European city. Landscape and Urban Planning 104:95–104.
Tang, Y., A. Chen, and S. Zhao. 2016. Carbon Storage and Sequestration of Urban Street Trees in Beijing, China. Frontiers in Ecology and Evolution 4.
Theobald, D. M. 2014. Development and Applications of a Comprehensive Land Use Classification and Map for the US. PLoS ONE 9:e94628.
Threlfall, C. G., Walker, K., Williams, N. S., Hahs, A. K., Mata, L., Stork, N., & Livesley, S. J. (2015). The conservation value of urban green space habitats for Australian native bee communities. Biological Conservation, 187, 240–248.
Tidåker, P., T. Wesström, and T. Kätterer. 2017. Energy use and greenhouse gas emissions from turf management of two Swedish golf courses. Urban Forestry & Urban Greening 21:80–87.
Tobias, S. (2013). Preserving ecosystem services in urban regions: Challenges for planning and best practice examples from Switzerland. Integrated Environmental Assessment and Management, 9(2), 243–251.
United States Census Bureau. 2020. 2013 – 2018 American Community Survey. U.S. Census Bureau's American Community Survey Office.
United States Department of Agriculture, Natural Resources Conservation Service (NRCS). (2009). Hydrologic Soil Groups. Chapter 7 in: Part 630, National Engineering Handbook. Report No. 210–VI–NEH. Washington, DC. 13 pp.

Vodyanitskii, Yu. N. 2015. Organic matter of urban soils: A review. Eurasian Soil Science 48:802–811.
Webber, J.L., Fu, G., and Butler, D. (2018). Rapid assessment of surface water flood intervention performance. Water Science & Technology, 77(8), 2084–2092.
Webber, J.L., Fu, G., and Butler, D. (2019). Comparing cost-effectiveness of surface water flood management interventions in a UK catchment. Journal of Flood Risk Management 12:1–12. https://doi.org/10.1111/jfr3.12523
Weston, D. P., Holmes, R. W., & Lydy, M. J. (2009). Residential runoff as a source of pyrethroid pesticides to urban creeks. Environmental Pollution, 157(1), 287–294. Retrieved from https://doi.org/10.1016/j.envpol.2008.06.037
Wittmer, I. K., Bader, H.-P., Scheidegger, R., Singer, H., Lück, A., Hanke, I., … Stamm, C. (2010). Significance of urban and agricultural land use for biocide and pesticide dynamics in surface waters. Water Research, 44(9), 2850–2862. Retrieved from https://doi.org/10.1016/j.watres.2010.01.030
Yoon, T., K. Seo, G. Park, Y. Son, and Y. Son. 2016. Surface Soil Carbon Storage in Urban Green Spaces in Three Major South Korean Cities. Forests 7:115.
Zardo, L., Geneletti, D., Pérez-Soba, M., & Van Eupen, M. (2017). Estimating the cooling capacity of green infrastructures to support urban planning. Ecosystem Services, 26, 225–235.
Zhao, J., L. Gladson, and K. Cromar. 2018. A Novel Environmental Justice Indicator for Managing Local Air Pollution. International Journal of Environmental Research and Public Health 15:1260.
Ziter, C., and M. G. Turner. 2018. Current and historical land use influence soil-based ecosystem services in an urban landscape. Ecological Applications 28:643–654.