HISTORICAL LEGACIES OF LAND USE IN CITIES; PARKS, OPEN SPACES AND POTENTIAL FOR GREEN INFRASTRUCTURE- IDEAS OF CITY NATURE IN AN URBANIZING PLANET

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HISTORICAL LEGACIES OF LAND USE IN CITIES; PARKS, OPEN SPACES AND POTENTIAL FOR GREEN INFRASTRUCTURE- IDEAS OF CITY NATURE IN AN URBANIZING PLANET

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EDITORIAL
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Editorial: Historical Legacies of Land Use in Cities; Parks, Open Spaces and Potential for Green Infrastructure- Ideas of City Nature in an Urbanizing Planet

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Editorial on the Research Topic

Historical Legacies of Land Use in Cities; Parks, Open Spaces and Potential for Green Infrastructure- Ideas of City Nature in an Urbanizing Planet

Research on urban nature, green infrastructure, and nature-based solutions to environmental problems has burgeoned over the past two decades. A good deal of that research is undergirded by the premise that trees, greenspaces, parks, and urban ecosystems are self-evidently good. Much less critical analysis has interrogated the assumptions that legitimize and privilege urban greening, in scholarly research, urban planning, and urban management. Comparatively little has been written about how the historical origins and legacies of urban land use shape greening outcomes. And very little has been written about critiques of the "urban greening project". Redressing these lacunae is important.

As global climate change impacts manifest ever more strongly, urban greening has moved from a nice idea to becoming what many consider an essential adaptive response. Against the backdrop of a global climate emergency, it is increasingly difficult to question the motives, assumptions, principles, and processes that enact urban greening. And with ramped-up scholarly attention to greenspace and green infrastructure driven by the global COVID19 pandemic—it seems almost heretical to question the doctrine of urban greening. Yet urban greening can have pernicious and unintended consequences. Urban greening can heighten the risk of wildfire, increase property values and displace marginalized residents, increase human wildlife conflict, promote vector borne disease, and increase other health hazards, such as asthma from pollen, biogenic volatile organic compounds, and trapped particulates, among others (Gibbs, 2019).

As Pataki et al. observe, it is vitally important that urban greening advocates use evidence, rather than received wisdom, to inform policy-making and decision-making. Urban greening initiatives can also distract from far more fundamental physical problems that drive urban inequality (e.g., urban heat, flooding and polluting land uses), problems that are embedded within the modernist urban morphology. Pavement type, street orientation, building densities, fossil fuel transport dependence, and exclusionary zoning, all contribute to urban inequality. But the focus on greening masks these more difficult, historically accreted issues, and eludes the question of how we should build and rebuild cities into the future to promote social inclusion, human diversity, equity, justice, and ecological sustainability.

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In this special issue, we have sought contributions that attempt to unpack the contemporary ideologies driving investments in green infrastructure and greenspace in cities across the world, including urban ecosystem science. Our aim was to take a step back and to try to objectively question the idea of "green is good" (Angelo, 2019). We asked ourselves and each other whether there might be a role for other colors within efforts to restore urban ecologies, provide habitat, alleviate environmental inequalities, and combat myriad urban environmental ills. Is there a place for the browns of grasslands, the grays of rocky outcrops, and the reds and oranges of lichens (Cohen, 2013)? We also questioned whether green might be a code word for a particular cultural politics of nature (Byrne, 2012), that privileges large trees over grassland, prairies, and moors; wealthy people over the poor; and White communities over people of color—(re)producing and entrenching environmental inequality (Schell et al., 2020). What is this urban vision of green and can it truly be disentangled from western imperialism, including gardening traditions exported across the globe (Doherty, 2017).

Interestingly, several submissions entirely ignored this call, focusing on quantification of GI, and/or pointing to the lack of GI and its reputed, but in many cases unstudied, value in the case discussed. Some articles have considered key questions about who has access to urban greenspace, who benefits and who does not. Collectively though, they are a first attempt to push scholarship in a more critical and reflexive direction, to pay attention to place and to history, to urban morphology and to the "what" and "why" of green infrastructure. It is quite surprising to observe, for example, that GI solutions appear to be generalized across cities, regardless of where they are situated and their land use or open space histories.

Smart et al. for instance, assess tree density and distribution in cities in different climate zones. They find that cultural factors play a significant role in both greening outcomes, but also in legitimizing urban greening programs, and note that street trees in European and North American cities were uncommon until the late nineteenth and early twentieth centuries. And there tends to be a "wealth effect" where prosperous neighborhoods have more trees. Moreover, they observe that urban planning, urban form, historical perceptions of street trees as a nuisance, and climatic factors all configure urban greening outcomes. In contrast, Berland et al., explain that higher levels of tree abundance in some cities is associated with systematic disinvestment, neglect, and perceptions of some places as undesirable. Perceptively, they observe that simply quantifying the presence or absence of vegetation is insufficient for explaining socio-ecological relationships. We also need to understand the role of residents' perceptions of urban greening.

In their article on perceptions of green infrastructure, Brown et al., point to the importance of understanding both green infrastructure services and disservices, noting that not all green infrastructure is perceived the same way. Both the form of the green infrastructure and the socio-demographic characteristics of nearby residents are important considerations. While people's attitudes and values are relatively stable and hard to change, Brown et al. point to the role that preferences can play in determining what is acceptable or not for different residents. Felson and Ellison build on this insight, showing that it is possible to integrate ecological theory with landscape architecture to devise urban greening interventions that do not necessarily "look messy" and which provide critical benefits for non-human species. Yet Allen et al., also show how the taken for granted benefits of greening for non-human species can have unintended or unforeseen impacts. They reveal how seemingly innocuous actions such as dog walking can have pervasive and enduring impacts on nitrogen distribution in built environments, and that high levels of nitrogen can accumulate in soils over time, potentially harming urban ecologies.

It is also important to consider how even well-intentioned planning can smuggle-in historical legacies of racism and colonialism. Shackleton and Gwelda in their article point to the ongoing social and ecological perturbations caused by apartheid in South Africa. Decisions made at a point in time can persist for decades or even centuries. Judgements about the suitability of species, weed invasion, naturalness and belonging resonate uncomfortably with racialized policies and practices and notions of who belongs and who is unwanted. Connolly and Anguelovski probe the relationship between greening and whiteness in US cities. Compellingly, they find a relationship between the economic trajectory of a city, degree of greening and whiteness—cities with sustained economic growth tend to be greener and whiter whereas the inverse applies in cities that are contracting or have punctuated periods of economic growth and stagnation. As Ossola et al. argue, we therefore need to understand not only the spatial, but also the temporal dimensions of the urban greening project.

To those interested in the future of cities and in the role of urban greening in addressing complex socio-ecological issues, we commend this special issue.

AUTHOR CONTRIBUTIONS

All authors listed have made a substantial, direct and intellectual contribution to the work, and approved it for publication.

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Influences of Environmental and Social Factors on Perceived Bio-Cultural Services and Disservices

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Although the value of urban ecological infrastructure (UEI) is widely recognized, insufficient research has investigated how people perceive the wide variety of UEI. To address this gap, we investigated residents' perceptions of the coupled value of aesthetic and biological qualities as related to diverse UEI and other environmental and social factors (including personal beliefs and demographics), collectively referred to as bio-cultural services and disservices. We evaluated whether people positively view their neighborhood environments as natural-looking while providing diverse plants and wildlife habitat (services), in contrast to negatively perceived disservices that we evaluated as messy-looking with weeds and pests (disservices). We analyzed survey data from residents (n = 495) in metropolitan Phoenix, Arizona, United States, coupled with environmental variables (UEI and vegetative cover) compiled from diverse sources. We ran three regression models to compare the relative influence of social and environmental factors independently and combined on the perception of bio-cultural services and disservices. Our results demonstrate the influence of social factors, particularly place identity, neighborhood cohesion, and income, on both bio-cultural services and disservices. Additionally, environmental factors such as vegetation cover increased perceptions of bio-cultural services while decreasing perceived disservices. The effects of proximity to UEI were more varied. While proximity to cropland increased perceived bio-cultural disservices, proximity to desert parks reduced disservices. Although UEI can promote biodiversity and human well-being, all UEI are not perceived the same. Our results underscore the added value of considering both the form of UEI and perceptions among people who live nearby when designing and implementing infrastructure to promote bio-cultural services that are both ecologically and socially valued.

Keywords: ecosystem services and disservices, urban ecological infrastructure, perceptions, urban planning, urban nature

INTRODUCTION

People's everyday relationships with their local environments largely depend on the structure and design of natural areas within urban settings. Globally, urban areas expanded by 10,000 km^2 per year between 1985 and 2015 (Li et al., 2018). The projected urban population of 5.2 billion by 2030 (up from 4.0 billion in 2015; United Nations, Department of Economic and Social Affairs, Population Division, 2019) underscores the importance of understanding complex relationships between people and nature in urban ecosystems (Pickett and Cadenasso, 2008; McPhearson et al., 2016). Although urban areas frequently experience ecological homogenization, especially compared to their outlying native ecosystems (Groffman et al., 2014, 2017), they also exhibit a high degree of ecological variability at local scales (Pickett et al., 2017). Research into the heterogeneity of urban landscapes, both within and between metropolitan areas, provides a deeper understanding of how local features influence the perceived benefits of varied landscapes, including those related to biological outcomes (Cadenasso et al., 2007, 2013). In this paper, we investigate how environmental features of people's neighborhoods, including urban ecological infrastructure (UEI), coupled with social factors, affect public perceptions of their aesthetic and biological value.

Urban ecological infrastructure encompasses all infrastructure in a city that supports ecological structure and function, and by extension, provides ecosystem services to urban residents (Childers et al., 2019). UEI is a broad, all-encompassing concept for "nature in cities." This idea includes commonly recognized forms of infrastructure, such as parks, residential yards, community gardens, lakes and rivers, and street trees. But UEI also includes less recognized forms, such as vacant lots, agricultural fields, canals, and water retention basins. Childers et al. (2019) categorized UEI into terrestrial, aquatic, and wetland ecosystem types because each type supports unique ecological structures and functions and thus provides different ecosystem services. Understanding how UEI influences people's perceptions can lead to improved city design and an increase in resident satisfaction with, and stewardship of, their local environments (Paul et al., 2014; Artmann et al., 2019).

At the turn of the twentieth century and through the Progressive Era, city planners and developers focused on gray infrastructure that provided basic services (e.g., water delivery, waste removal, flood control, etc.) in a mostly successful effort to make urban areas "sanitary" (Melosi, 2008; Pincetl, 2010; MacKinnon and Derickson, 2013). More recent urban solutions have promoted "green" infrastructure, or UEI broadly, that are more adaptive and flexible than gray infrastructure while also providing multiple benefits to residents (Campbell, 1996; Colding and Barthel, 2013; Elmqvist et al., 2015; Li et al., 2017). Integration of UEI can simultaneously improve both human well-being as well as biodiversity in urban environments (Tzoulas et al., 2007; Ekkel and de Vries, 2017). Exposure to biodiversity and natural areas has positive impacts on human well-being (Cox et al., 2017a,b), but these impacts can vary based on people's subjective views of local landscapes (Syrbe and Walz, 2012; Shwartz et al., 2014; Pett et al., 2016). Importantly, for urban residents, perceived biodiversity is more important in driving psychological well-being than actual biodiversity (i.e., species richness) (Dallimer et al., 2012; Pett et al., 2016).

While the benefits and conflicts surrounding nature in cities are widely acknowledged (Soulsbury and White, 2016), research has not fully linked different types of UEI to public perceptions of their aesthetic qualities. For example, green spaces that have a diversity of native plants often benefit wildlife by providing habitat in cities (Marzluff and Rodewald, 2008; Niemelä et al., 2010); meanwhile, these features can also benefit people through aesthetic appreciation and other positive outcomes for human well-being (Casalegno et al., 2013; Hernández-Morcillo et al., 2013; Fish et al., 2016). However, certain types of UEI that provide habitat can also be perceived as messy and unkempt (van Heezik and Ludwig, 2012). These UEI might elicit negative responses by urban residents because of an association with pest or nuisance species (Lyytimäki et al., 2008; Maruthaveeran and van den Bosh, 2015). From this point on, we refer to these linked biological and cultural ecosystem services and disservices as bio-cultural services and disservices. In particular, our analysis centers on natural-looking landscapes and their perceived ability to provide wildlife habitat as a bio-cultural service, as well as perceptions regarding messy-looking landscapes that attract pests as a disservice.

Research has explored how local wildlife contribute to bio-cultural services, specifically in terms of perceptions of birds (Lerman and Warren, 2011; Belaire et al., 2015). More broadly, research has revealed an array of perceptions in relation to diverse environmental features (e.g., trees and parks) across different urban contexts (Flannigan, 2005; Fernandes et al., 2019). However, the literature largely fails to examine public perceptions of bio-cultural services and disservices flowing specifically from proximity to diverse UEI, along with other environmental and social factors. Moreover, research often focuses on social or environmental explanations for ecosystem services, often through biophysical evaluations or economic valuation of ecosystem services (Casalegno et al., 2013; Buizer et al., 2016; Li et al., 2017). Further, existing research on perceptions of UEI tends to focus on temperate climates in which the distribution of vegetation and UEI may be more uniform than in arid climates. Herein, we couple social and environmental datasets in a desert metropolis to examine how an array of potential social and environmental factors influence people's subjective views of a specific type of bio-cultural service, focusing on perceptions of vegetation and wildlife.

In this research, we asked: what social and environmental features, including proximity to varied UEI, influence perceived bio-cultural services and disservices. Overall, since personal values and experiences strongly influence public perceptions (Bruvold, 1973; Rokeach, 1973; Bell et al., 1996), we hypothesize that social factors (including personal beliefs and demographics) will better explain perceived services and disservices than local UEI and other environmental conditions. Regarding environmental features, we specifically test whether distinct UEI differentially influences perceived bio-cultural services and disservices. By understanding patterns in perceived bio-cultural services and disservices, planners and practitioners can more

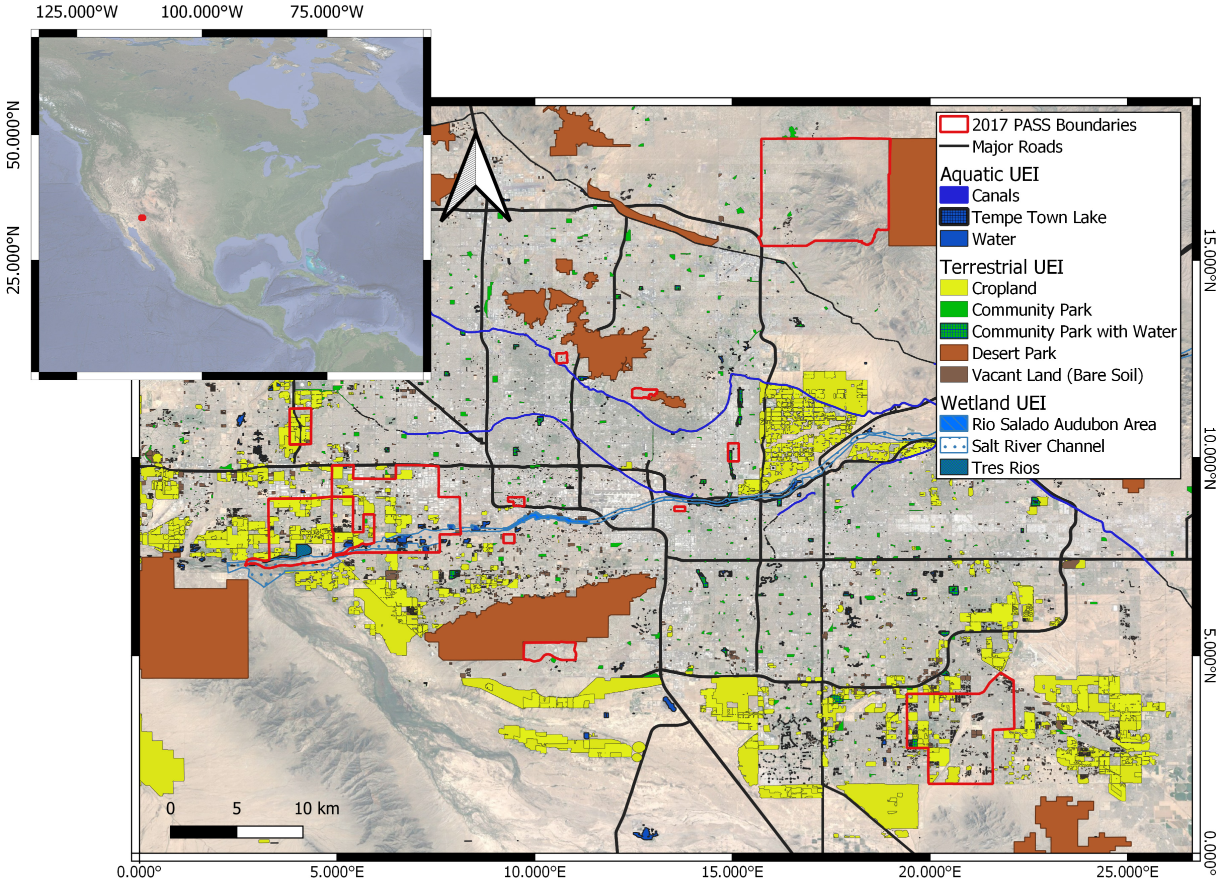

FIGURE 1 | The Phoenix study area (indicated in red on the map of North America). PASS neighborhoods outlined in red. Aquatic, terrestrial, and wetland UEI are shown throughout the study area. Base maps are from Google Earth 2017.

effectively promote biodiversity and the design of UEI that are appreciated by the public (Buizer et al., 2016; Larson et al., 2019). Designing UEI according to societal preferences is critical since people's values and attitudes are often steadfast and difficult to change (e.g., through education and outreach efforts; Heberlein, 2012). Ultimately, how the public perceives and values UEI will affect support for their implementation, management, and continued investments.

MATERIALS AND METHODS

Study Site

The Phoenix Metropolitan Area (**Figure 1**) was originally settled by the Hohokam people who created a series of irrigation canals that branched from the Salt River and allowed farming to occur in the arid landscape (Trimble and Trimble, 2003). Hundreds of years after the Hohokam left the region, Anglo-Americans settled there in the 1800s. As Phoenix developed, some of the original Hohokam canals paved the path for modern canals that deliver water throughout Phoenix. Due to restoration efforts, some areas of the canals and Salt River Channel are bordered by native vegetation, but most canals are hardscaped with concrete. Given the upstream storage of the Salt and Verde River water in dams built in the early twentieth century, most of the river channels that run through the metropolitan region are ephemeral.

Agricultural settlements developed along the canals to facilitate widespread irrigation use. Although much of the farmland has been developed into residential and other urban land uses, agrarian land remains throughout the region (**Figure 1**; Keys et al., 2007; Kane and York, 2017). Currently, farmland includes dominant crops of hay, cotton, and wheat (York et al., 2020). The region is also home to a large dairy industry.

Despite the arid environment, which receives 13 cm of rain annually, the regional landscape is traditionally characterized as an oasis, distinct from the surrounding desert (Larson et al., 2009). These lush landscapes include irrigated agriculture as well as ample irrigated grass and more than 1,400 artificial bodies of water (Larson and Grimm, 2012). Although residential lawns are ubiquitous, drought-tolerant xeric landscapes with gravel groundcover and low water-use plants have become increasingly common in recent decades (Martin, 2015). Meanwhile, large undeveloped swaths of the Sonoran Desert—including several desert parks and preserves—exist within and at the edges of the metro area.

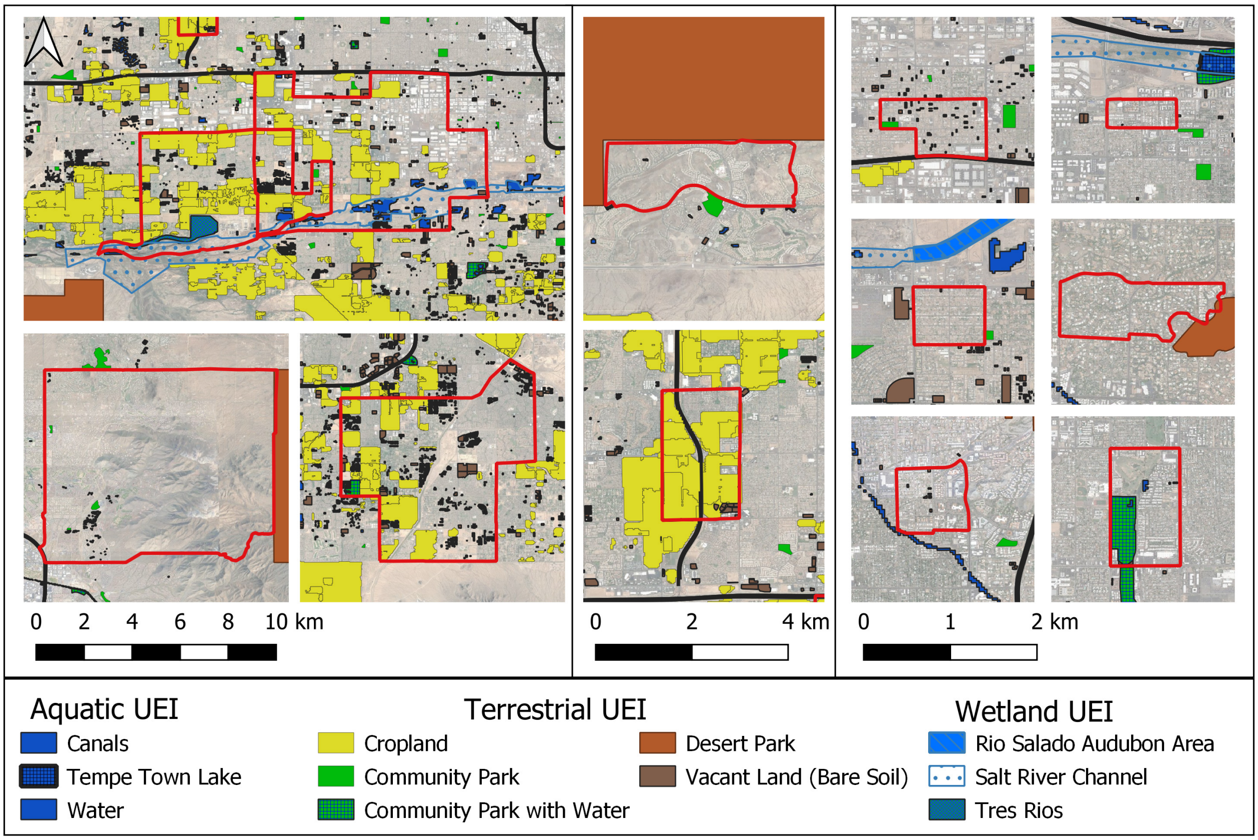

FIGURE 2 | Close ups of the various PASS neighborhoods from the study area. PASS neighborhoods outlined in red. Aquatic, terrestrial, and wetland UEI are shown throughout the study area. Base maps are from Google Earth 2017.

Phoenix Area Social Survey (PASS)

To measure people's perceptions of bio-cultural services and disservices and select explanatory factors in our analysis, we used data collected from the 2017 Phoenix Area Social Survey (PASS[1]; Larson et al., 2019). The PASS is a longitudinal survey effort conducted as a part of the Central Arizona-Phoenix Long-Term Ecological Research Program. The 2017 survey targeted 12 neighborhoods, delineated by Census Block Groups, strategically located to capture a range of demographic characteristics (e.g., low to high income levels) and central, suburban, and exurban locations in proximity to diverse UEI (**Figures 1**, **2**). The survey was sent to 1,400 addresses between May and September of 2017; 188 addresses represented households that responded to a previous PASS (2011), and the other 1,212 surveys were randomly drawn from addresses provided by the Marketing Systems Group, which come from the U.S. Postal Service's Delivery Sequence Files.

The University of Wisconsin survey lab administered the questionnaires to households via four-wave mailing, including three full questionnaires and a reminder postcard sent in-between mailings (Larson et al., 2019). Individuals could request a Spanish version with a postage-paid postcard. Regardless of response, the survey lab sent a $5 incentive as well as post-response incentives (ranging from $5 to $40; see Smith et al., 2020 for details) to increase participation. A total of 39.4% of contacted households responded for a total of 496 completed surveys. We dropped one respondent who removed their identifier, since we could not map their location and link their responses to the environmental data including local UEI features. See Larson et al. (2020) for all survey questions.

Bio-Cultural Services and Disservices Variables

We measured the ecosystem service and disservice variables using a Likert scale with responses ranging from strongly disagree (1) to strongly agree (5). In particular, we asked the following question: "To what extent do you disagree or agree that the following statement describes the environment in your neighborhood" (Larson et al., 2019). We specifically asked respondents to consider their local "environment" as, "the grass, plants, and/or trees in the area, along with the streets, sidewalks, patios, porches, and built structures as well as parks and open spaces." We purposefully included both ecological and built infrastructure since both influence ecosystem services and disservices in urban areas (Shackleton et al., 2016).

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[1] https://sustainability.asu.edu/caplter/research/long-term-monitoring/phoenix-area-social-survey/

TABLE 1 | Dependent variables in composite scales (i.e., bio-cultural services and disservices) with their means and standard deviations.

Variable (alpha)		Mean (SD)	Number of responses
Bio-cultural services (0.71)		3.72 (0.88)	495
	Looks natural	3.68 (1.13)	494
	Offers a variety of plants	3.68 (1.07)	490
	Provides habitat for birds	3.82 (1.07)	489
Bio-cultural disservices (0.71)		2.41 (1.04)	493
	Looks messy	2.23 (1.24)	493
	Attracts unwanted animals or pests	2.62 (1.25)	493
	Has too many weeds	2.39 (1.28)	490

Cronbach's alpha is reported for both composite scales, and the mean, SD, and number of valid responses for the individual variables are also reported.

The statements analyzed herein include: "looks natural," "offers a variety of plants," and "provides habitat for birds," in addition to "looks messy," "attracts unwanted animals or pests," and "has too many weeds." We created two composite survey scales for our dependent variables, with the first three variables averaged for each respondent to capture bio-cultural services and the latter three averaged to capture bio-cultural disservices. Larson et al. (2019) informed the development of the survey scales, along with the scholarly literature and expert input among our interdisciplinary research team. We deemed the two scales reliable through a standard test of internal consistency; each scale had Cronbach alpha values greater than the 0.7 criterion (**Table 1**, see also **Supplementary Figure 1**).

Environmental Variables

Urban Ecological Infrastructure

We measured UEI to link larger scale and landscape features to perceptions of bio-cultural services. Following Childers et al. (2019), we first classified UEI as aquatic (containing perennial water), terrestrial (no water), or wetlands (both terrestrial and aquatic due to ephemeral water features). The three broad categories are ecologically distinct and thus provide disparate ecosystem services broadly. However, while these broad classifications can be useful for distinguishing between ecosystem services (specifically provisioning or regulating services), they may not adequately measure bio-cultural services. To better understand perceptions of bio-cultural services, we investigated 11 different forms of UEI (**Table 2**), identified through a combination of expert opinion and pre-existing data sources (Maricopa Association of Governments, 2014; Zhang and Li, 2017; Smith et al., 2017). Local governments, including the cities of Phoenix and Tempe, have restored or redesigned segments of the river channel. We therefore classified these UEI features separately (i.e., Tempe Town Lake, Rio Salado Audubon, and Tres Rios Wetlands) relative to the remaining river channel, which is ephemeral due to upstream dams and the distribution of water throughout the region (**Table 2**). We also included agricultural land and vacant land (Smith et al., 2017; Childers et al., 2019). See **Supplementary Figure 2** for example images of the varying UEI.

We used QGIS version 3.12.2 to calculate the amount of UEI (in hectares) within a 1 km radius of a respondent's home for all UEI except the Salt River and canals. Since the Salt River and canals are predominantly linear features, we measured the distance (in meters) from respondents' homes to these features instead of the area within 1 km of the respondent these features occupied. We defined residents' local environments as those within a 1 km radius since this distance captures how most respondents (84%) defined their neighborhood. Moreover, planners commonly consider this distance as walkable and, thus, individuals are likely to interact with UEI within this radius (Macedo and Haddad, 2016). Descriptive statistics for all UEI are provided in **Table 2**.

Although we classified UEI broadly as aquatic, terrestrial, or wetland, we did not that expect all UEI within a classification would be perceived similarly due to differences in accessibility (e.g., open vs. closed to the public), appearance (e.g., relatively managed vegetation in community parks vs. natural desert preserves), and function (e.g., croplands producing commercial agricultural products vs. public parks used for recreation). For instance, UEI that was not publicly accessible, such as croplands and the Tres Rios Wetlands, are less likely to impact perceptions since people cannot visit and interact with these UEIs as readily as public spaces such as parks (de Groot et al., 2010). Additionally, UEI that lacks active management of vegetation or other features (e.g., wetlands, vacant lots) may be perceived as messy and thereby be perceived negatively (Rega-Brodsky et al., 2018; Stoffel, 2020). Finally, we predicted that natural areas established to protect and promote biodiversity, such as desert preserves, will enhance perceptions of bio-cultural services.

Vegetation Cover

Vegetation density and the extent to which the study neighborhoods had xeric landscaping provided further information about the vegetative structure of a respondent's neighborhood not captured by the UEI. We used two metrics, Normalized Difference Vegetation Index (NDVI and neighborhood yard composition (proportion of xeric yards in a neighborhood), to assess local vegetative cover. The NDVI data was measured at a 1 meter resolution from the National Agriculture Imagery Program 1–4 June 2017. To reflect local neighborhood environments, we used the average NDVI value within a 1 km radius of each respondent. We also included a proxy for neighborhood yard composition (percent xeric) based on survey responses since we did not have information on all yards within a neighborhood. Specifically, respondents were

TABLE 2 | Classifications and descriptions of UEI used in this study as well as associated data sources.

UEI form	UEI	Description	Data sources
Aquatic	Canals	Canals have a hard, artificial banks that contain flowing water. The area adjacent to canals sometimes includes walking and biking paths as well as other amenities.	2008 data from the ASU Geospatial Repository and Google Earth Imagery from 2017
	Tempe town lake	An artificial lake created by damming a portion of the Salt River Channel and pumping water to keep it filled year-round. The lake is flanked by pathways and recreational infrastructure including a grassy park, playground, amphitheater, and boat dock. It is located to the north of downtown Tempe and is managed by the City. The boundaries (**Figure 1**) were delineated around the permanently filled water body. This feature is differentiated from other bodies of water due to its unique management scheme as well as the amount of development and amenities associated with this artificial lake.	Landsat TM5 imagery from Zhang and Li (2017) at a 30 m resolution and 2014 park delineation data from the City of Phoenix and the ASU Geospatial Repository.
	Other water	Lakes and other small bodies of water that are perennial.	Landsat TM5 imagery from Zhang and Li (2017) at a 30 m.
Terrestrial	Community parks	These are green spaces that are designed primarily for human use. They typically contain mesic landscaping and amenities such as benches, paths, playgrounds, and open recreation areas. These areas are also open to the public and designed for public use and are sometimes the location of local and community events. These areas are maintained by municipalities and are much smaller than the desert preserves. Community parks in this study have an average size of 0.832 ha and a maximum size of 795.191 ha.	Landsat TM5 imagery from Zhang and Li (2017) at a 30 m resolution, and 2014 park delineation data from the City of Phoenix and the ASU Geospatial Repository.
	Community parks with water	These have the same features as community parks but also contain one or more permanent water features. We distinguish these parks from community parks because the presence of water may be attractive to both people and wildlife.	Landsat TM5 imagery from Zhang and Li (2017) at a 30 m resolution, and 2014 park delineation data from the City of Phoenix and the ASU Geospatial Repository.
	Cropland	Cropland is defined as a combination of both active cropland (classified as vegetated croplands) and inactive cropland (classified as bare soil croplands).	Landsat TM5 imagery from Zhang and Li (2017) at a 30 m resolution.
	Desert preserves	These are large, municipally maintained natural areas. They consist of large contiguous desert and natural vegetation with hiking and biking trails. Human access to these preserves is restricted to designated areas, with the remaining land dedicated to preserving wildlife. Within this study, desert preserves have an average size of 1,852.943 ha with a maximum size of 11,114.006 ha.	Landsat TM5 imagery from Zhang and Li (2017) at a 30 m resolution, and 2014 park delineation data from the City of Phoenix and the ASU Geospatial Repository.
	Vacant land	Classified as 'vacant by Maricopa county (maricopa.gov) and include parcels of land that may be developed but currently unoccupied and typically underutilized (Smith et al., 2017). These unmaintained areas have minimal vegetation and consist primarily of bare soil or gravel.	Smith et al., 2017
Wetlands	Rio Salado Audubon area	The Nina Mason Pulliam Rio Salado Audubon center is managed by the National Audubon Society with a mission of protecting birds and their associated habitat. It serves as the headquarters of the Audubon Society in Arizona and contains a large visitor center. The Area consists of ∼2,400 ha of restored riparian habitat within and directly adjacent to the Salt River channel. It is home to over 200 species of birds. In addition to wildlife habitat, the Rio Salado Audubon Area also has variety of hiking trails and the center hosts events for the public.	Delineated using expert opinion and information from riosalado.audubon.org
	Salt River channel	The Salt River is approximately 320 km long and is the largest tributary of the Gila River. The Tres Rios Wetlands, Rio Salado Audubon Area, and Tempe Town Lake all are within the Salt River channel, which also includes some permanent and semi-permanent bodies of water. This area contains several "accidental" wetlands (per Suchy et al., 2019) as well as natural vegetation and human built structures for water management. This urban stretch of the Salt River has not seen perennial flow since 1938, but during large storms flow in the channel may exceed 4,000 m^3/s (United States Geological Survery, 2010).	Delineated using expert opinion, Landsat TM5 imagery from Zhang and Li (2017) at a 30 m resolution, and Google Earth Engine 2017.
	Tres Rios Wetlands	Tres Rios includes both a large constructed treatment wetland and a riparian restoration project. The former treats effluent from the 91st Avenue wastewater treatment plant, the goal of the latter is to serve as a public amenity; both are habitat for over 150 species of birds and other animals.	Delineated using expert opinion, Landsat TM5 imagery from Zhang and Li (2017) at a 30 m resolution, and Google Earth Engine 2017.

asked what percent of their front yard was composed of grass. We classified yards that were less than 50% grass as xeric yards and then measured the percent of xeric yards (0.00–1.00) in a neighborhood as the number of xeric yards reported divided by the total number of respondent's yards in the neighborhood.

Social Variables

We analyzed social variables from the PASS survey to represent personal values and beliefs, in addition to demographic attributes. We also included the age of housing (from county tax assessor data), since we expected historical patterns of development might influence perceptions of local landscapes (Tengberg et al., 2012; Locke et al., 2020).

Personal Values and Beliefs

To capture environmental values, we used the standard 15-question New Ecological Paradigm (NEP) scale (Dunlap et al., 2000). The NEP reflects broad-based beliefs about the relationship between people and the natural environment or, in other words, the extent to which people hold biocentric (or pro-ecological) versus anthropocentric (i.e., human-centered) views. We averaged responses from the NEP scale to create a single variable ranging from 1 to 5 (**Table 3**). Values closer to 1 represent anthropocentric orientations, whereas values closer to 5 captured biocentric views.

As one dimension of place attachment, we measured neighborhood identity using the average of five different variables (following a standard scale developed by Williams and Vaske, 2003). The verbatim survey question included statements such as, "I feel my neighborhood is a part of me." and "I identify strongly with my neighborhood." (see Larson et al., 2020 for the complete list of questions). This index ranged from 1 to 5 with 1 indicating an individual who feels a weak connection to their neighborhood and 5 representing an individual with a strong connection to their neighborhood (**Table 3**). Individuals who strongly identify with their neighborhood are likely to perceive their local environments more positively than others (for example, as shown by Brown and Raymond, 2007).

TABLE 3 | Descriptive statistics for environmental and social explanatory variables.

Variables	Mean	Std. Dev.	Min	Max	Valid N
Vegetative cover					
NDVI	0.00	0.05	−0.10	0.25	495
Percent xeric	0.77	0.13	0.6	1	495
Cognitive factors					
Neighborhood identity$^+$	3.66	1.09	1	5	493
Pro-ecological orientations$^+$	3.70	0.70	1.5	5	495
Social cohesion$^+$	3.07	0.70	1	5	494
Demographics and housing					
Age	51.37	17.88	18	96	486
Age of house	30.23	20.88	1	89	432
Education	4.58	1.21	1	6	483
Gender	1.60	0.49	1	2	486
Income	5.31	3.19	1	11	456
LatinX	1.22	0.41	1	2	475
Years current address	12.27	11.54	0	64	442
Years in valley	26.78	18.39	0	90	479

*See **Table 4** for UEI variables. $^+$These are composite variables. Please see **Supplementary Table 1** for more details.*

TABLE 4 | Average, minimum, and maximum amount (in ha) of each UEI that was measured within a 1 km radius relative to all respondents.

Variables	Average	Minimum	Maximum
Canals (A)	8.94 km	0.12 km	24.96 km
Tempe town lake (A)	0.31 ha	0.00 ha	13.51 ha
Water (A)	2.20 ha	0.00 ha	60.63 ha
Community parks (T)	1.57 ha	0.00 ha	16.34 ha
Community parks with water (T)	1.88 ha	0.00 ha	37.07 ha
Cropland (T)	23.56 ha	0.00 ha	197.29 ha
Desert preserves (T)	6.28 ha	0.00 ha	148.28ha
Vacant land (T)	5.65 ha	0.00 ha	61.26 ha
Rio Salado Audubon area (W)	0.31 ha	0.00 ha	20.42 ha
Salt River Channel (W)	10.57 km	0.11 km	29.53 km
Tres Rios (W)	0.00 ha	0.00 ha	30.47 ha

For canals and the Salt River Channel which were measured as the distance of the feature from the respondent, the average, minimum, and maximum distance relative to all respondents is reported (km). The UEI classification for each variable is denoted as aquatic (A), terrestrial (T), or wetland (W). The average total UEI (sum of all averaged UEI) is also reported.

Lastly, we used a measure of social cohesion to reflect one aspect of social capital. The concept of social capital conveys the value of interpersonal relationships and networks in taking collective actions, among other human assets (Putnam and Putnam, 2000; Locke et al., 2020). Following Larsen et al. (2004), we assessed social cohesion using the average response to four statements, such as, "I live in a close-knit neighborhood." and "I can trust my neighbors" (see **Supplementary Material** for the complete list of statements). The scale ranges from 1 to 5, with 1 indicating individuals who felt a weak connection to their neighbors and 5 indicating that individuals felt a strong connection with their neighbors (**Table 3**).

Demographics and Housing

In the survey, each respondent provided their date of birth, from which we calculated their age. Age is associated with the amount of free time an individual has (Goerres, 2007) and is associated with how frequently individuals visit parks and local neighborhood features (Godbey and Blazey, 1983). Older individuals often prefer bio-cultural features such as natural areas in community parks as opposed to recreational areas (Alves et al., 2008).

For tenure of residence in the Phoenix area and at the surveyed home, we asked respondents: "How many years have you lived in the Phoenix metropolitan area?" and "How many years have you lived at your current address?" The amount of time an individual has lived in one location can positively influence their perception of that area and their willingness to care for it (York et al., 2017; Sorensen et al., 2018). People who have lived in an area longer may feel stronger attachment to that area and are therefore more likely to perceive it positively (Vaske and Kobrin, 2001; Altman and Low, 2012).

We coded gender on a binary scale with 0 as male and 1 as female. Gender can influence perceptions of risk (Flynn et al., 1994; Slovic, 2000), which may in turn, for example, influence women's perceptions of messy, pest-ridden UEI as being dangerous (Sreetheran and van den Bosch, 2014). An individual's engagement with environmental issues and their perceived importance of natural spaces has shown to vary by gender, with women perceiving natural systems as more important than men (Mohai, 1997; Momsen, 2007).

To capture socioeconomic status, we included income and education. Respondents reported their household income on an 11-point scale in $20,0000 increments, from $20,000 to over $200,000. People with higher income levels likely have more control over their local landscape, which might influence their perception of bio-cultural services and disservice that relate to the biodiversity in the neighborhood (Lerman and Warren, 2011; Locke et al., 2020). We measured education on a 7-point scale, from completion of grades 1 to 8 to attainment of a graduate or professional degree. Education was included because of its association with income (Morgan and David, 1963; Gregorio and Lee, 2002) and its positive association with environmental attitudes (Lundmark, 2007).

For ethnicity, we asked respondents whether they identified as LatinX (including Mexican, Mexican-American, Chicano, Hispanic, or LatinX). We coded the LatinX variable on a binary scale with 1 as not LatinX and 2 as LatinX. We limited the response to this binary variable as the ethnic composition of the study area and respondents is primarily white/Caucasian and LatinX (white/Caucasian and LatinX respondents accounted for 91.4% of respondents in this study). Previous work has shown that LatinX ethnicity is associated with environmental views. For example, LatinX are less likely to perceive the "natural" desert environment positively (Andrade et al., 2019). LatinX individuals may more likely perceive wildlife as a potential risk (Ramer et al., 2019; Larson et al., 2020), and thus, may perceive disservices more strongly than services.

Finally, we obtained housing age by linking respondent addresses to the Maricopa County Assessor's Office (Parcel Secured Master 2017), which includes the year the home was built. In the Phoenix metropolitan area, newer construction at the fringe of the city is associated with increased exposure to potentially dangerous or negatively perceived animals like snakes (Pitts et al., 2017).

Analyzing the Influence of Environmental and Social Variables

We used generalized linear models since these models do not assume that independent variables are pulled from a normal distribution. We first ran four models to compare how environmental (**Tables 5**, **6**) versus social factors (**Table 7**) explained the bio-cultural service and disservice variables separately. Additionally, to test whether broad UEI classifications (terrestrial, aquatic, and wetland) influence perceived bio-cultural services and disservices, we ran another model in which we centered and scaled all UEI variables and then combined all values within a single broad category (e.g., scaled scores for canals, Tempe Town Lake, and water were combined as "aquatic UEI") and also included average NDVI and xeric yard variables (**Tables 6**). In the more specified environmental model, we used the distinct UEI variables (**Tables 2**, **5**) as well as the other environmental variables. The social models incorporated the varied personal beliefs and demographic variables as explanatory factors. The final models combined all environmental (at the specific UEI level) and social variables. To ensure variables were not co-linear, we also ran a variable inflation factor to ensure we did not have co-linearity with our predictor variables. No independent variables had a score over 5 indicating non-significant co-linearity of predictor variables within our model (James et al., 2014). Additionally, since the independent variables were measured on different scales, we standardized all variables by subtracting the mean value of each variable and dividing by its

TABLE 5 | Environmental models predicting perceptions of bio-cultural services and disservices. Model fits (R^2) for dependent variables are presented.

Environmental variable	Bio-cultural services (R^2 0.20)	Bio-cultural disservices (R^2 0.25)
Vegetative cover		
NDVI average	0.24***	−0.16**
Xeric yard percent	–	–
UEI		
Canals	–	–
Tempe town lake	–	–
Water	–	−0.13**
Cropland	−0.24***	0.16**
Community park	–	–
Community park with water	–	−0.15**
Desert park	0.16**	−0.19***
Vacant land (bare soil)	−0.14**	0.14**
Rio Salado Audubon	–	0.10*
Salt River channel	0.22**	−0.27***
Tres Rios	–	–

Standardized beta values for each model are reported with significance indicated by "," "**," and "***" representing $p < 0.05$, $p < 0.01$, and $p < 0.001$. Beta values for non-significant variables not included.*

TABLE 6 | Environmental models at the broadly classified UEI level predicting perceptions of bio-cultural services and disservices.

Environmental variable	Bio-cultural services (R^2 0.07)	Bio-cultural disservices (R^2 0.08)
Vegetative cover		
NDVI average	0.20***	−0.12*
Xeric yard percent	0.09*	−0.12**
UEI		
Aquatic	–	–
Terrestrial	−0.22***	0.14**
Wetland	–	0.18***

Model fits (R^2) for dependent variables are presented next to each dependent variable. Standardized beta values for each model are shown below with significance indicated by "," "**," and "***" representing $p < 0.05$, $p < 0.01$, and $p < 0.001$. Beta values for non-significant variables not included.*

standard deviation before running our analysis. All analyses were conducted in R version 3.6.1 (R Core Team, 2019).

RESULTS: FACTORS INFLUENCING PERCEIVED BIO-CULTURAL SERVICES AND DISSERVICES

UEI Distribution

On average, the survey respondents lived within 1 km of eight of the eleven types of UEI (**Table 4**). Cropland was the most abundant in terms of land cover within 1 km of our survey respondents, likely because the four largest neighborhoods from our study had high amounts of cropland within them. Access to restored or redeveloped portions of the Salt River channel was low among survey respondents overall, given their relatively small area coverage in specific locations of the region.

Environmental Models

Out of the two sets of environmental models, the UEI specific models (**Table 5**) outperformed the broadly classified UEI models (**Table 6**). For bio-cultural services and disservices separately, the broadly defined UEI models had an R^2 of 0.07 and 0.08 (**Table 6**), whereas the UEI specific models had a fit of 0.20 and 0.25 (**Table 5**).

The environmental models that captured distinct UEI identified five of the thirteen environmental variables and explained 20% of the variation in perceived bio-cultural services (**Table 5**). Increasing neighborhood NDVI and proximity to desert preserves significantly increased perceptions of bio-cultural services while increasing the amount of cropland and vacant land decreased perceived bio-cultural services (**Table 5**).

TABLE 7 | Social model results for predicting perceptions of bio-cultural services and disservices.

Social variable	Bio-cultural services (R^2 0.35)	Bio-cultural disservices (R^2 0.29)
Cognitive factors		
Neighborhood identity	0.27***	−0.17**
Pro-ecological orientations	–	−0.11*
Social cohesion	0.20***	−0.14*
Demographics and housing		
Age	–	−0.15*
Age of house	–	0.22***
Education	0.14*	–
Gender	–	–
Income	0.21***	−0.16**
LatinX	–	–
Years current address	–	–
Years residency in region	–	–

Model fits (R^2) for dependent variables are presented next to each dependent variable. Standardized beta values for each model are shown below with significance indicated by "," "**," and "***" representing $p < 0.05$, $p < 0.01$, and $p < 0.001$. Beta values for non-significant variables not included.*

By comparison, eight of the thirteen environmental variables captured 25% of the variation in perceived disservices (**Table 5**). The same variables that enhanced perceptions of bio-cultural services decreased perceptions of bio-cultural disservices. Similarly, the variables that diminished perceptions of bio-cultural services increased perceived bio-cultural disservices. Additionally, increasing proximity to the Rio Salado Audubon Area increased perceptions of bio-cultural disservices. Two additional variables, community parks with water and open water, lowered perceptions of bio-cultural disservices but did not increase perceived bio-cultural services.

In the broadly defined UEI model, terrestrial UEI decreased perceived bio-cultural services while increasing bio-cultural disservices. In this model, wetland UEI also increased perceived bio-cultural disservices but had no influence on bio-cultural services (**Table 6**). Meanwhile, aquatic UEI were not significantly associated with either perceived bio-cultural services or disservices. Additionally, the results from this model show that higher NDVI and more xeric yards also increased perceived bio-cultural services while reducing perceived bio-cultural disservices (**Table 6**).

Social Models

As hypothesized, the social models explained more variation in perceived bio-cultural services (R^2 = 0.35) as well as disservices (R^2 = 0.28) than the environmental models. Of the eleven social variables, four and seven variables were significant in the ecosystem services and disservices models, respectively (**Table 7**). Identification with one's neighborhood, as one dimension of local place attachment, was the most influential factor explaining perceived bio-cultural services, followed by income and perceived social cohesion. Education levels, another measure of socioeconomic status, also significantly explained the perceived services.

The same variables significantly explained perceived bio-cultural disservices but in the opposite direction, apart from education which was not significant (**Table 7**). Additional significant variables included age of housing, which had the largest influence on perceived disservices, with older homes associated with lower perceived bio-cultural disservices. People who have lived in the region longer also tended to perceive more bio-cultural disservices compared to others, while older people perceived fewer disservices. Lastly, pro-ecological worldviews were associated with perceptions of bio-cultural disservices.

Combined Models

The combined models, which included both the social and environmental variables (**Table 8**), explained the most variation in perceived bio-cultural services (R^2 =0.39) and disservices (R^2 =0.35). Social variables still proved to be strong predictors of perceptions; specifically, neighborhood identity, social cohesion, and income followed the same patterns as in the social model. Environmental variables less strongly explained perceived bio-cultural services, though average NDVI within a neighborhood was positively associated with perceived bio-cultural services while cropland and Tempe Town Lake were negatively associated with bio-cultural services. All other environmental and social

TABLE 8 | The combined model combines environmental and social variables for bio-cultural services and disservices.

Variables	Bio-cultural services (R^2 0.39)	Bio-cultural disservices (R^2 0.35)
Environmental factors		
Vegetative cover		
NDVI average	0.17**	–0.16*
Xeric yard percent	–	–
UEI		
Canal	–	–
Tempe town lake	–0.14**	–
Water	–	–
Cropland	–0.19*	0.09**
Community park	–	–
Community park with water	–	–0.15*
Desert park	–	–0.19**
Vacant land (bare soil)	–	–
Rio Salado Audubon	–	–
Salt River channel	–	–
Tres Rios	–	–
Social factors		
Cognitive factors		
Neighborhood identity	0.27***	–0.14*
Pro-ecological orientations	–	–
Social cohesion	0.18**	–0.13*
Demographics and housing		
Age	–	–
Age of housing	–	0.18*
Education	–	–
Gender	–	–
Income	0.18**	–
LatinX	–	–
Years current address	–	–
Years residency in region	–	–

Model fits (R^2) are presented next to each dependent variable. Standardized beta values for each model are shown below with significance indicated by "," "**," and "***" representing near significance, $p < 0.05$, $p < 0.01$, and $p < 0.001$. Beta values for non-significant variables not included.*

variables were statistically insignificant in the combined model for perceived bio-cultural services.

In the combined models for perceived bio-cultural disservices, the same variables were significant for perceived bio-cultural services except proximity to Tempe Town Lake (**Table 8**). Several additional variables uniquely explain perceptions about bio-cultural disservices, including proximity to desert parks (negative relationship), distance to the Salt River and water (both negatively related). Among the social variables, the age of housing was significant with older homes being associated with more perceived disservices.

DISCUSSION

Our research illustrates that people's perceptions of bio-cultural services and disservices are related to both local landscape features (including varied UEI) and social factors including personal beliefs and demographics. While these factors can be investigated independently, our results showed that coupling environmental and social factors best explain variations in perceptions of ecosystem services and disservices. Our study also demonstrates the importance of considering social factors, such as neighborhood identity and social cohesion, as drivers of perceived ecosystem services. This consideration is important since the ultimate value placed on bio-cultural services by people depends largely on public perceptions of them, which are often highly subjective and contingent upon personal beliefs and experiences (Heberlein, 2012; Larson et al., 2019). However, since our sample does not represent the entire population of the study area, and instead captures predominantly older individuals with high levels of income and education, we suggest caution in generalizing the results to Phoenix as a whole. However, this work may provide more direct insight into the perception of UEI in arid land systems than other studies conducted in more temperate climates.

While ecologists, planners, and other professionals tout the value of UEI (e.g., Li et al., 2017), our research shows that UEI is not always linked to positive perceptions of local environments among nearby residents. Our results also show that many forms of UEI are negatively perceived (e.g., croplands and wetlands) and thus we recommend careful consideration of how to integrate these features in landscape design and planning. While our dependent variables, bio-cultural services and disservices, only captured a narrow set of perceptions regarding bio-cultural services, as related to their aesthetic quality coupled with vegetation and wildlife features, our results indicate that people's views vary in relation to local environmental and social factors.

Our analysis of perceived bio-cultural services underscores that, although social factors were more important predictors of people's perceptions, UEI still plays a role in explaining perceived bio-cultural services and disservices. Our findings suggest that landscape elements such as vegetation density and preserved natural areas can increase people's aesthetic appreciation of, and support for, UEI while also reducing perceived ecosystem disservices. Similarly, features such as open water can influence aesthetic appreciation (Cottet et al., 2013). However, much of the restored or redeveloped areas of the Salt River channel do not significantly influence perceived bio-cultural services and disservices. Our findings echoed those of Sokolow (2003) and Hamilton et al. (2014) finding negative perceptions of cropland. Therefore, in urbanizing areas with adjacent residential and agricultural land, planners and decision makers may wish to look for ways to mitigate negative perceptions of cropland adjacent to residential areas. Overall, our findings stress the importance of considering the specific characteristics of different UEI as well as the landscape context around UEI when assessing public perceptions of these infrastructure and their associated ecosystem services and disservices.

As one of the most dominant UEI in our study area, cropland was among the most significant predictors of perceived bio-cultural disservices. While agricultural areas can have strong cultural value to those in rural and agrarian communities (Lincoln and Ardoin, 2016), these areas may be perceived negatively by those that live in urban, suburban, and exurban areas (Thomson and Kelvin, 1996; Hamilton et al., 2014). This finding is notable given modern interests in urban and near-urban agriculture for varied purposes such as food security, the reduction of food waste and the carbon footprint associated with food transportation, and open space preservation (Coley et al., 2009; McClintock, 2010; Mok et al., 2014). The form of agriculture present in Phoenix may be particularly unappealing to urban residents, due to its uniform, large, flat monocrop fields with little topographic or vegetative variation, and long periods with bare soil (York et al., 2020). Inclusion of habitat refugia in agricultural lands, which can positively influence perceptions of ecosystem services (Diekötter et al., 2008), might be one angle for enhancing biological conservation and societal appreciation of agricultural lands in metropolitan regions. Efforts to make agricultural areas more suitable for wildlife may also reduce the negative impacts croplands have on perceived biocultural services. Ultimately, the ability to enhance the co-benefits from agriculture will depend on support for farmers (Eakin et al., 2016) and regional development policies.

Our research further illustrates the role of spatially distributed nature preserves and community parks in promoting perceptions of aesthetically pleasing local environments that are also biologically valuable. The lack of significant effects of community parks and other accessible UEI on perceptions might be due to substantial variability across parks in terms of specific park features that influence their perceived bio-cultural value. Given the perceived bio-cultural value of preserved deserts in our Phoenix-based study, combined with overwhelming preferences for neat, orderly landscapes (Nassauer, 1995; Larson and Brumand, 2014), incorporating these design elements into parks and other accessible UEI sites may increase their perceived bio-cultural value among diverse residents. Nonetheless, we recommend additional research to clarify how specific environmental features (e.g., varied vegetation structure and land use contexts) among different UEI influence perceptions in diverse contexts. Since the results from our neighborhood-based survey may be highly context specific, especially in relation to features in local parks such as Tempe Town Lake and Indian Bend Wash (**Figure 1**), additional research is needed to validate results across different contexts.

Although water influences individual's perceptions of beauty (Burmil et al., 1999; Völker and Kistemann, 2011), the type of water matters. Respondents positively perceived water when present in community parks, likely due to the aesthetic value placed on open water (e.g., ponds and lakes) (Burmil et al., 1999; Asakawa et al., 2004). But in our study, two large features that contain water were negatively perceived: Tempe Town Lake and the Salt River channel. In the case of Tempe Town Lake, nearby residents may correctly perceive the lack of bio-cultural values offered by this site since the lake's edge is largely surrounded by concrete paths devoid of significant vegetation, which detracts from its "natural" aesthetic and value to wildlife. In terms of the Salt River Channel, a couple of potential factors may contribute to negative bio-cultural perceptions. The channel and its banks are largely unmanaged and, thus, may appear disorderly or unkempt (Nassauer, 1995). Unhoused people occupy these spaces (Palta et al., 2016; DeMyers et al., 2017), and as a result, the channel and banks may be viewed as messy (i.e., due to associated trash found at these sites) or otherwise unsafe or undesirable. Additionally, evaluations of wetlands are often influenced by factors such as the lack of visible open water, dense vegetation, and pests such as mosquitos (Cottet et al., 2013; Landau and van Leeuwen, 2012).

Another explanation for negative views of aquatic and wetland UEI in the study region is that broader landscape context matters, and in the case of the Salt River, much of the channel is flanked by agricultural or industrial lands. Since industrial areas often produce pollution, noise, and other disservices, they are negatively perceived by people who live near them (Burningham and Thrush, 2004; Flanquart et al., 2013). Therefore, we may be capturing the negative response of individuals to the industrial areas surrounding the Rio Salado Audubon site, as opposed to the restored river channel itself. This may reflect the legacy of historic urban development and the management of rivers as functional conduits (e.g., to deliver water and remove waste).

As investments in UEI grow with the shift from gray to green infrastructure designs (Elmqvist et al., 2015; Li et al., 2017), researchers and practitioners might evaluate how the broader land use context of UEI influences how specific sites are perceived, used, and valued by people. In doing so, it will likely take time to overcome the legacy effects of historic gray infrastructure designs, such that public sentiment shifts from relatively negative views of river channels and associated UEI toward appreciation. Such a shift seems underway in metropolitan Phoenix, as some local governments have started to invest in features (e.g., signage, lights, paved pathways) alongside long-established canals to enhance their recreational amenities (Tenny, 2020). However, these investments are often inequitably distributed (Kuras et al., 2020), thereby potentially contributing to the socioeconomic disparities in perceived services in our analyses (**Table 7**). Further, the addition of recreational amenities or features such as lights and sidewalks, which increase the usability and value of spaces for people, may negatively impact wildlife and thus reduce bio-cultural services (Bennett et al., 2009).

As public investments in UEI increase, whether for social (e.g., recreational) or ecological (e.g., biological conservation) purposes, the perceived aesthetics are imperative to establish and maintain public support and appreciation. This objective entails designing with "cues of care" that demonstrate intentional maintenance and design features that people value (e.g., curvilinear features, some mown or trimmed vegetation; Nassauer, 1995). Meanwhile, to enhance biological conservation, designing sites with native vegetation, a variety of plants, and significant vegetation structure can help to preserve native biodiversity and provide wildlife habitat (Lerman and Warren, 2011; Standish et al., 2013). We realize, however, that these actions may not be possible in some cases. For instance, the rules established by the Federal Aviation Authority prevent establishment of significant vegetation along Tempe Town Lake, since this UEI is in the flight zone and attracting birds to the area may obstruct or cause concerns for air travel. Additional policies and other factors may constrain UEI features and designs, depending on the primary goals of the infrastructure along with applicable regulations, property ownership, and resources available for investing in UEI.

While the prevalence of UEI in a respondent's neighborhood predicted perceptions of bio-cultural services, as we hypothesized, social factors, especially neighborhood identity and social cohesion, better predicted perceived ecosystem services and disservices than UEI. This result supports previous findings showing that people who feel a stronger sense of community are more likely to positively perceive their surrounding environment, including the beauty of their neighbors' homes and yards (Dempsey, 2008; Murphy-Dunning, 2009). This finding is also true for people who feel a strong connection to their neighborhood, as they are more likely to take actions which promote bio-cultural services and reduce bio-cultural disservices (Goltsman et al., 2009; Locke et al., 2020). Interventions that promote social cohesion and increase neighborhood connection, such as scheduling social meetings, gardening, or clean-up events, may be one way to increase the perceived value of local landscapes (King et al., 2010; Yamamoto, 2011). As such, facilitating social cohesion events, including participation in decisions to change and maintain UEI in neighborhoods, may enhance perceptions of local bio-cultural services (Yamamoto, 2011).

CONCLUSION

Characterizing diverse UEI provides a framework for evaluating their ecological and social benefits and impacts, both real and perceived. The evaluation of perceived ecosystem services and disservices is particularly important for understanding societal appreciation and support for specific UEI and associated characteristics. Although many types of UEI investigated herein did not significantly influence perceived bio-cultural services and disservices, future studies can refine the attributes of UEI beyond broad classifications (such as terrestrial or aquatic). Together, such research can inform specific design features that are valued by people. In developing or redesigning UEI into the future, we recommend that planners, landscape architects, and other site designers and managers work with researchers and community members to create UEI in ways that are socially valued both aesthetically and for biological conservation purposes.

DATA AVAILABILITY STATEMENT

All data analyzed in this study is publicly available and can be found at the CAP LTER data portal (https://data.sustainability.asu.edu/cap-portal/home.jsp).

ETHICS STATEMENT

The studies involving human participants were reviewed and approved by Arizona State University (ID: 662) IRB—Social & Behavioral Research Group. The patients/participants provided their written informed consent to participate in this study.

AUTHOR CONTRIBUTIONS

KL conceptualized the project idea. KL and JB decided upon the methodology and wrote the manuscript. JB conducted all data compilation, analyses, and created all tables and figures for the project. SL acted as lead editor of the manuscript and contributed to methods and writing. DC helped classify urban ecological infrastructure, contributed writing, and provided comments. RA contributed writing and provided comments for the manuscript. HB contributed to the methodology and provided comments. SH, PW, and AY contributed comments and reviewed the manuscript. KL and AY co-directed the social survey referenced in this study. All authors contributed to the article and approved the submitted version.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fevo.2020.569730/full#supplementary-material

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Beauty or Blight? Abundant Vegetation in the Presence of Disinvestment Across Residential Parcels and Neighborhoods in Toledo, OH

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Urban vegetation can generate social and ecological benefits, so vegetation abundance is commonly treated as a proxy for greater benefits. A repeated finding in environmental justice research related to urban vegetation is that commonly marginalized populations live in neighborhoods with less vegetation. However, urban vegetation can function as amenity or disamenity depending on the context and the characteristics of the vegetation. In areas of disinvestment, overgrown vegetation may indicate neglect and lead to negative social outcomes. For example, previous research in the shrinking city of Toledo, Ohio, showed that areas with concentrated residential vacancy and high representation of traditionally marginalized populations also had relatively high vegetation abundance. This can be largely attributed to spontaneous, weedy vegetation in areas of concentrated vacancy. Equal vegetation cover therefore should not necessarily be equated with environmentally just outcomes. Here, we used several high-resolution data sets to study the relationships among vegetation abundance, vegetation quality, and property parcel occupancy on residential land in Toledo. Our results demonstrate that vacant residential land had more abundant vegetation than comparable occupied parcels according to two common metrics (tree canopy cover and the normalized difference vegetation index). Compared to occupied parcels, vacant parcels also had higher rates of blight associated with overgrown vegetation, as recorded during a citywide ground-based survey of property conditions. There were more vacant parcels overall in areas of disinvestment, and on a per-parcel basis, vacant parcels in these high-vacancy areas were also greener relative to nearby occupied parcels than vacant parcels in low-vacancy areas. This indicates that vacant parcels play a disproportionately large role in greening on residential land in areas of disinvestment. These results reinforce the idea that simply quantifying vegetation abundance may be insufficient for understanding urban social-ecological outcomes. Incorporating parcel occupancy data along with multiple strands of information about vegetation type and condition provides context to understand where abundant vegetation functions as amenity vs. disamenity. These perspectives are especially relevant in shrinking cities like Toledo where legacies of urban socioeconomic change have produced widespread areas of disinvestment and land abandonment.

Keywords: shrinking cities, normalized difference vegetation index (NDVI), urban tree canopy, residential vacancy, land abandonment

INTRODUCTION

Despite dramatic urbanization globally, hundreds of cities around the world have experienced multi-decadal decline (Hollander et al., 2009; Martinez-Fernandez et al., 2016). These shrinking cities (also termed legacy cities) have undergone sustained population loss and economic retraction driven by the decline of industry, suburbanization, and white flight, among other factors (Martinez-Fernandez et al., 2016). Shrinking cities are characterized by areas of concentrated vacancy and disinvestment, and these patterns can be reinforced through a weakening tax base, the legacy of racial segregation, and persistent poverty (Xie et al., 2018).

Toledo, OH is a shrinking city in the North American "Rust Belt." Toledo experienced substantial population growth through the middle of the 20th century as a prominent manufacturing center, but the city has seen a decline in population from over 383,000 in 1970 to <279,000 today (US Census Bureau, 2020). As the population in Toledo has declined, vacancy rates have increased. This has left a patchwork of vacant lots across the landscape, embedded among parcels with unoccupied and occupied residences, businesses, and industrial sites. While areas of disinvestment may be particularly prominent in shrinking cities, not all neighborhoods within a shrinking city experience decline (Tighe and Ganning, 2015). Like many U.S. shrinking cities, some of Toledo's neighborhoods have experienced concentrated vacancy while others have minimal vacancy at levels indicative of a healthy real estate market (Mallach, 2018).

The heterogeneity in vacancy across Toledo intersects with the uneven distribution of amenities and hazards long documented by environmental justice researchers (Bullard, 1994; Schlosberg, 2009; Taylor, 2014; Agyeman et al., 2016), but see Baró et al. (2019) for a recent exception that did not find uneven distributions. One important point of intersection between vacancy and environmental justice is urban vegetation, often considered an environmental good or amenity due to the associated ecosystem services (e.g., shade, stormwater interception, human health benefits, biodiversity conservation) provided by trees and other vegetation (Seamans, 2013; Dobbs et al., 2014; Willis and Petrokofsky, 2017; Ibáñez-Álamo et al., 2019). However, context often dictates whether urban vegetation and its attendant social-ecological functions act as amenity or disamenity (Heynen et al., 2006; Hoalst-Pullen et al., 2011; Pearsall and Christman, 2012; Herrmann et al., 2016; Roman et al., 2018). For example, abundant spontaneous vegetation in a vacant lot may perform beneficial ecosystem services such as stormwater interception, but it is unlikely to provide social benefits for neighborhood residents (Riley et al., 2018). Conversely, carefully manicured vegetation may be socially desirable even though it does not maximize ecological benefits (Gobster et al., 2020).

Recognizing the importance of context highlights two shortcomings of existing research in which authors have used the unequal distribution of vegetation within urban areas to illustrate potential cases of environmental injustice (e.g., Heynen et al., 2006; Landry and Chakraborty, 2009; Danford et al., 2014), and particularly where a proposed remedy is to increase the amount of vegetation in underserved areas. First, greening as a solution to unequal vegetation distributions is often accompanied by concerns from residents about gentrification and displacement (Wolch et al., 2014; Anguelovski, 2016), the lack of genuine inclusion in deciding how greening efforts will proceed (Carmichael and McDonough, 2018), or burdens imposed by the need to maintain newly planted vegetation (Heynen et al., 2006; Pincetl, 2010). Note that we use the term *greening* to describe a general increase in vegetation, regardless of intent. This is in line with previous studies that have used the abundance of vegetation to study (in)equitable distributions of urban vegetation (e.g., Landry and Chakraborty, 2009; Pham et al., 2012), but it differs from studies that use greening to refer specifically to strategies intended to restore beauty and reduce blight (e.g., Krusky et al., 2015; Gobster et al., 2020). We characterize blight as recognizable elements of physical disorder such as unmowed lawns, untrimmed trees and hedges, and abundant weeds. Second, the distribution of vegetation is typically assessed quantitatively but not qualitatively, because qualitative data are not readily available. This is a shortcoming because the abundance of vegetation says little about its quality (Leslie et al., 2010; Hoalst-Pullen et al., 2011; Schwarz et al., 2018). We do not know then whether residents perceive the vegetation as amenity or disamenity. However, qualitative and quantitative indicators of vegetation desirability and evidence of stewardship or maintenance present opportunities to address this shortcoming (Nassauer, 2011; Camacho-Cervantes et al., 2014; Gobster et al., 2020).

Greening and the desirability of urban vegetation must be further situated in its changing contexts. In the case of shrinking cities, indicators of urban greenness gradually respond to population loss and land abandonment (Hoalst-Pullen et al., 2011). Previous work in Toledo showed that the relationship between residential vacancy and the normalized difference vegetation index (NDVI), an indicator of greenness, changed over the several decades following the onset of population loss (Schwarz et al., 2018). In 1980, NDVI was lower in census tracts with higher percentages of residential vacancy, but by 2014 there was not a statistically significant relationship between NDVI and vacancy rates. This was attributed to increasing vegetation in tracts with higher vacancy rates, plausibly driven in large part by spontaneous vegetation on vacant parcels that was not likely to be considered amenity vegetation by the community (Schwarz et al., 2018). This suggests that vacancy and vegetation quantity and quality intersect in important ways, potentially altering the balance between whether urban vegetation is considered beauty or blight.

The intersection of these vacancy and vegetation templates is crucial to understand and support sustainable and equitable city futures (Herrmann et al., 2016; Riley et al., 2018). Disinvestment in legacy cities is uneven, realized through the heterogeneous distribution of both vacancy and amenities/hazards. Failing to understand the intersection among amenities/hazards and vacancy in shrinking cities will likely perpetuate an uneven recovery as well. One important aspect of this intersection that remains understudied is the role of heterogeneous vacancy in driving urban vegetation patterns. It is important to understand how different types of vegetation (i.e., understory vs. canopy) influence observed patterns in shrinking cities, how perception of vegetation quality relates to vegetation quantity, or how these patterns emerge at multiple relevant spatial scales.

Here, we use both parcel- and neighborhood-level data to explore patterns in urban vegetation (NDVI, tree canopy cover, and a qualitative indication of overgrown lawn) in relation to residential vacancy within the city of Toledo. Parcel- and neighborhood-level data are complementary in interpreting urban vegetation outcomes. Parcels importantly have specific identities as vacant or occupied, and represent the scale at which most urban vegetation management decisions are made. Neighborhood-level data integrate many parcels to capture emergent patterns, and neighborhoods provide the immediate context that can influence parcel management decisions. Using these complementary data levels, we address the following research questions:

1) **What are the neighborhood-scale relationships among vacancy and the following greenness indicators: tree canopy, NDVI, and the prevalence of overgrown lawns?** We hypothesize that tree canopy and NDVI will be higher in both low- and high-vacancy areas, and lowest in areas with intermediate levels of vacancy. This could be explained by an abundance of amenity vegetation in low-vacancy areas. In high-vacancy areas, this would reflect the emergence of spontaneous vegetation on vacant parcels, and we expect this to be reinforced by a higher prevalence of overgrown lawns in high-vacancy areas.

2a) **Are vacant residential parcels greener than comparable occupied parcels?** If we control for confounding factors by pairing parcels that are similarly sized and in close proximity to one another, we expect that vacant parcels will generally have higher tree canopy cover, higher NDVI values, and higher prevalence of overgrown lawns than occupied parcels. This will reflect the emergence of spontaneous vegetation on vacant parcels. Additionally, the relative greenness of vacant lots with no structures will be amplified, because these parcels do not have residential structures and thus have more space for vegetation to grow.

2b) **If vacant parcels are greener than comparable occupied parcels, does the magnitude of this difference vary across geographic space?** By definition, we know there will be more vacant parcels overall in high-vacancy areas. But it is also possible that the difference in greenness between vacant and occupied parcels is greater in high-vacancy areas compared to low-vacancy areas, indicating that vacant parcels in high-vacancy areas make a disproportionately large contribution to neighborhood greening on a per-parcel basis. In high-vacancy areas, this could potentially be explained by lower social pressure or capacity to actively maintain vegetation on vacant parcels, coupled with the relative paucity of amenity vegetation on occupied parcels. In low-vacancy areas, we expect occupied parcels to be greener on account of more abundant amenity vegetation, and stricter social norms may encourage proactive maintenance of vegetation on vacant parcels. Therefore, we expect the difference in greenness between vacant and occupied parcels in low-vacancy areas to be weaker (or negative), signaling that vacant parcels make a proportionally smaller contribution to neighborhood greening on a per-parcel basis in low-vacancy areas.

MATERIALS AND METHODS

Study Area

Toledo (41.65° N, 83.54° W) is situated in Lucas County, Ohio, where the Maumee River flows into Lake Erie. Toledo is located in the humid continental climate zone where vegetation can grow without water subsidy. Toledo and Lucas County have a history of park and green space investment through the City of Toledo Division of Parks, Recreation and Forestry, and through Metroparks (Rozick, 2010), a regional program that has several large holdings with high quality conservation and recreation assets on more than 12,000 acres. Historically, there has been less investment in parks concentrated in downtown Toledo. In 2017, parks and green space only accounted for 2% or 32 acres of downtown Toledo (Downtown Toledo Master Plan., 2017). Shifting the distribution of open space has been a focus of urban planning, with the Downtown Toledo Master Plan highlighting the need for a network of parks to better connect people to the historic focus of the city, the Maumee River. The first urban Metropark, Middlegrounds, opened in 2016 and is part of a planning vision to transform Toledo's waterfront and connect East and West Toledo (Downtown Toledo Master Plan., 2017).

Following a prosperous era as a glass manufacturing center, Toledo experienced substantial economic contraction in the second half of the 20th century associated with deindustrialization (Floyd, 2015). Like other shrinking cities in the North American "Rust Belt," the loss of manufacturing jobs, suburban development, and social factors in Toledo led to a sustained population decline from 383,818 in 1970 to 282,275 in 2015 (US Census Bureau, 2020). The population loss and economic decline was not experienced evenly throughout Toledo, as some neighborhoods were hit harder than others. Disinvestment was evident in redlined neighborhoods where financial institutions withheld mortgage lending, and where government-sponsored urban renewal projects disrupted communities by constructing highways through predominantly African American neighborhoods (Abdelazim et al., 2016).

In 2015, the poverty rate in Toledo was 27.8%, which was nearly double the poverty rates in the state of Ohio (15.8%) and the US as a whole (15.5%) (US Census Bureau, 2020). The 2015 citywide residential vacancy rate in Toledo was 15.1%, which Mallach (2018) classifies as high vacancy. Many areas of the city had higher vacancy rates (>20%) that can be characterized as hypervacancy; in these areas, abandoned properties are likely to remain vacant due to poor market conditions (Mallach, 2018). Collectively, vacant parcels are conservatively estimated to cost Toledo \$9.2 million annually through direct costs (e.g., code enforcement, policing), lost tax revenue from vacant parcels, and lost tax revenue attributable to reduced property values for occupied parcels near vacant parcels (Immergluck et al., 2016). Due to the widespread presence of vacant residential properties in Toledo and the decades-long economic contraction and population loss, Toledo is well-suited to studying the interactions among disinvestment, residential vacancy, urban greening, and blight associated with overgrown vegetation.

Data Collection and Preparation

Parcel Vacancy Data

Parcel-scale analysis offers insights at the fundamental geographic unit of land management for residential parcels in the US (Manson et al., 2009). To characterize vacancy at the parcel scale, we obtained Toledo Survey data in table format from the Lucas County Land Bank. The Toledo Survey is a comprehensive walking survey of property condition that was completed in 2014–2015 for over 121,000 parcels in Toledo. The survey data include observations on whether a structure was present, whether a structure appeared occupied, and indicators of physical disorder such as peeling paint or missing siding, missing or broken windows, and evidence of fire damage. These indicators of physical disorder were recorded as part of the Toledo Survey's efforts to better understand the prevalence and geographic distribution of property blight within the city. The Toledo Survey data table was joined to property parcel polygons from the Lucas County Auditor in a geographic information system (GIS). Parcel information from the Auditor was used to identify and extract residential parcels. Using the Toledo Survey data, we classified residential parcels into three mutually-exclusive categories based on observed occupancy and the presence/absence of a residential structure: (1) *occupied structures*, which were parcels with occupied residential structures (86.6% of total); (2) *vacant structures*, defined as parcels with unoccupied residential structures (4.4% of total); and (3) *vacant lots*, which were residential parcels without a structure present (9.0% of total). We use the term *vacant parcels* to collectively describe both vacant structures and vacant lots.

Vegetation Data

We obtained or derived three parcel-scale vegetation metrics from existing open data: (1) overgrown lawns as an indicator of blight, (2) tree canopy cover, and 3) NDVI. The Toledo Survey documented the presence or absence of overgrown lawn and/or dumping, henceforth overgrown lawn. Tree canopy cover was obtained from a 2012 tree canopy map at 0.3 m resolution developed for Toledo using object-based classification techniques (Li, 2017). We assessed the overall accuracy of the canopy cover map as 91% accurate, based on visual interpretation of 300 randomly distributed points using 1 m resolution air photos from 2015. Tree canopy at the parcel scale was calculated as the proportion of each parcel's land area occupied by tree canopies when viewed from above. NDVI is an indicator of vegetation abundance and health. NDVI ranges from −1 to 1, where lower values indicate water, bare soil/rock, or impervious surfaces, and increasingly higher values indicate more vigorous vegetation. Note that while average NDVI values tend to be substantially lower for NAIP imagery compared to Landsat imagery (Zhang et al., 2019), the interpretation that higher NDVI values indicate more vigorous vegetation remains applicable. NDVI was derived based on 1 m resolution, growing season color-infrared aerial photography from the National Agriculture Imagery Program (NAIP) acquired from OCM Partners (2020). A mean NDVI value was calculated for the pixels located within each residential parcel to derive a parcel-scale measure for subsequent analysis.

Conceptually, we expected these three greenness indicators to provide somewhat different insights. Due to the slow growth of trees (Roman et al., 2018), tree canopy cover registered tall vegetation that was planted or otherwise began growing years earlier. On the other hand, NDVI is responsive to any green vegetation regardless of its height or age. Thus, NDVI detected trees and low-lying vegetation such as turf grass, as well as ruderal herbaceous plants and woody shrubs that colonized sites in the absence of proactive yard maintenance. Whereas, both tree canopy cover and NDVI are quantitative measures of vegetation abundance, the overgrown lawn indicator was used as a qualitative indicator of perceived blight related to reduced yard maintenance.

Data Analysis

Patterns in Vacancy and Greenness at the Neighborhood Scale

Parcel-scale data were aggregated by census block groups to analyze neighborhood-scale patterns within Toledo. Block groups are designed to be relatively homogeneous with respect to population characteristics. Each residential parcel was assigned to the block group that contained the parcel's centroid to avoid slivers and spurious associations. Toledo contained 303 block groups. We excluded block groups with fewer than 20 residential parcels, resulting in 297 block groups included in analyses. The mean and median number of residential parcels per included block group was 348 (st. dev. = 150.3), and 338, respectively.

We used ordinary least squares (OLS) regression to analyze how vacancy related to indicators of greenness among block groups. We aggregated parcel greenness metrics to the block group scale to construct separate regression models for mean percent tree canopy cover, mean NDVI, and percent of parcels with overgrown lawns. The mean canopy cover and NDVI metrics were calculated as the mean of parcel means for all residential parcels located in each block group. In each regression model, percent vacancy was the independent variable and the greenness metric was the dependent variable. We tested models with a linear term only, and with a linear and quadratic term because exploratory data analyses showed potentially non-linear relationships. The model with the lowest Akaike information criterion (AIC) value was chosen as the model with the best, parsimonious fit.

Spatial dependence among observations can lead to faulty inferences when applying OLS regression (Anselin and Bera, 1998), so we tested the OLS model residuals for spatial autocorrelation using the Moran's I statistic. When Moran's I indicated spatial autocorrelation in OLS residuals, we implemented simultaneous autoregressive models (SAR) to control for spatial dependence. The Lagrange multiplier test was used to guide the selection of the appropriate SAR model, following guidance from Anselin (2005). A first-order queen contiguity spatial matrix was implemented in the SAR models, following earlier studies (e.g., Pham et al., 2012; Grove et al., 2014) and based on the expectation that observed values for a block group are partially dependent on the values of adjacent neighboring block groups. We report Nagelkerke pseudo-R^2-values because simple R^2-values are not appropriate for SAR models. While pseudo-R^2-values do not report the proportion of variability explained by the model, they can be compared among models as a relative measure of model explanatory power.

Patterns in Greenness and Occupancy Status at the Parcel Scale

A primary goal of this study was to compare greenness across occupancy categories at the parcel scale. We used independent, two-way ANOVA to test for effects of occupancy class, overgrown lawn status, and their interaction on both tree canopy and NDVI. When the interaction effects were significant, we used Tukey's HSD *post-hoc* test to determine which combinations of occupancy class and overgrown status were different from one another.

We conducted an additional parcel-scale analysis to control for potential influences of neighborhood-scale factors. Vacant parcels are spatially concentrated (**Figure 1**), so observed differences in greenness based on occupancy status could be influenced by confounding geographic patterns at the neighborhood scale like urban form, development history, and socioeconomic status (Roman et al., 2018). For example, if vacant parcels are typically found in neighborhoods with smaller parcel sizes, then parcels with vacant structures may appear less green simply because their structures occupy a larger proportion of the parcel on average, leaving less space to grow vegetation. Conversely, if occupied parcels are more frequently located in neighborhoods with larger parcel sizes where structures occupy proportionally less space, they will generally have more space to grow vegetation. Failing to control for such confounding factors could essentially result in a comparison of greenness attributable to neighborhood-scale geographic patterns across high- vs. low-vacancy areas, rather than an analysis isolating patterns related to occupancy status.

We controlled for these confounding neighborhood-scale factors by matching each vacant parcel with up to three comparable occupied parcels (Aschengrau and Seage, 2008). Comparable occupied parcels were defined as those meeting the following two matching criteria: (1) the parcel land area was within ±10% of the vacant parcel's land area, and (2) the parcel centroid was within 100–300 m of the vacant parcel's centroid. The land area criterion increased the likelihood that matched pairs had similar yard sizes, thereby increasing the likelihood that greenness comparisons were capturing vegetation management rather than differences in yard size. The proximity criterion controlled for unobserved neighborhood factors such as socioeconomics or social norms that could correspond with different vegetation management practices across different neighborhoods within Toledo. The minimum distance threshold of 100 m guarded against matching abutting parcels that could register overhanging tree canopy from the neighboring parcel's trees. The minimum distance threshold also increased the chances that matched pairs would constitute independent observations with respect to vegetation management, because very close neighbors tend to exhibit mimicry in yard maintenance (Minor et al., 2016). From the list of candidate occupied parcels that met both criteria, three parcels were selected at random as matches. This matching procedure was implemented separately for vacant structures and for vacant lots. The parcel matching strategy facilitated more meaningful comparison of greenness according to occupancy status than simply comparing all occupied parcels to all vacant parcels, because the matching process controlled for other confounding factors that likely influence greenness. To assess matching success, we calculated the mean number of occupied matches per vacant structure and per vacant lot, and we compared the mean parcel sizes of matched pairs to ensure they were similar.

Using the matched parcels, we compared vacant parcels and their occupied matches for each vegetation metric (tree canopy cover, NDVI, and overgrown lawn status). For both NDVI and tree canopy cover, paired t-tests were used to test for a difference in means between vacant parcels and their occupied matches. McNemar's test was used to assess whether the proportion of parcels with overgrown lawns was different between vacant parcels and their occupied matches.

Geographic Discrepancies in the Relative Contribution of Vacant Parcels to Neighborhood Greenness

By definition, vacant parcels are more abundant in high-vacancy neighborhoods than low-vacancy neighborhoods, and thus we would expect vacant parcels to play a larger overall role in greenness patterns within high-vacancy areas. But we also hypothesized that vacant parcels in high-vacancy areas make a larger contribution to greenness *on a per-parcel basis* compared to low-vacancy areas. We expected that low-vacancy areas have relatively more amenity vegetation on occupied parcels, and stricter vegetation maintenance norms compared to high-vacancy areas. These strict social expectations could translate to more proactive vegetation maintenance on vacant parcels in low-vacancy areas, including activities like regular mowing and brush clearing. In high-vacancy areas, vacant parcels may be greener because lawns are more likely to grow taller, and weedy trees

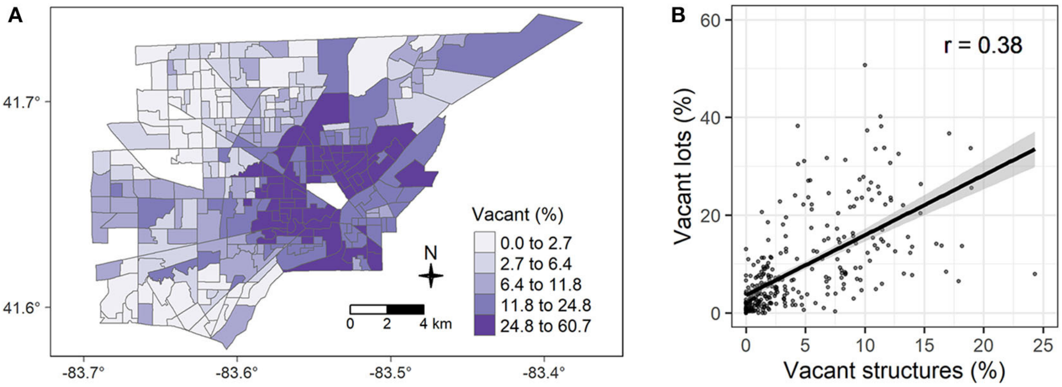

FIGURE 1 | (A) Map of residential vacancy rates (vacant structures plus vacant lots) in Toledo. Filled polygons represent census block groups with $\geq$20 residential parcels. For reference, the downtown core is located near the center of the map in the unfilled (white) block groups. **(B)** Scatterplot demonstrating the positive relationship between vacant structures and vacant lots within Toledo block groups. Data: Toledo Survey.

and shrubs grow unchecked. We expected the discrepancy in greenness between high- and low-vacancy areas to be stronger for NDVI than for tree canopy, because NDVI captures the low-lying, spontaneous vegetation typical of vacant parcels while tree canopy does not.

To test this hypothesis, we calculated the mean difference in NDVI between vacant parcels and their occupied matches by block group, excluding block groups with low sample sizes of <10 matched pairs. Then we used the Getis-Ord G_i* statistic with a first-order queen contiguity spatial matrix to identify geographic clusters of high and low values. High values (hot spots) indicated block groups with a large, positive difference in NDVI between vacant parcels and their occupied matches that were surrounded by other block groups that also exhibited large, positive differences. Clusters of low values (cold spots) were those block groups and their neighbors that had particularly small positive differences or negative differences in NDVI between vacant parcels and their occupied matches. The analysis was conducted separately for vacant structures and vacant lots, and then we repeated the analysis for tree canopy cover. The Getis-Ord G_i* statistic is reported as *z*-scores indicating whether a block group and its neighbors have differences between vacant parcels and occupied matches that are higher or lower than expected.

RESULTS

Patterns in Vacancy and Greenness at the Block Group Scale

All three OLS regression models used to assess the relationships between vacancy rates and greenness indicators at the block group scale showed significant spatial autocorrelation in their residuals. As recommended by the Lagrange multiplier test, spatial lag models were implemented for both tree canopy cover and NDVI, while a spatial error model was used for overgrown lawn. For each greenness indicator, the SAR with a quadratic term fit better than the model with a linear term only. The Nagelkerke pseudo-R^2-values were 0.65, 0.66, and 0.73 for tree canopy cover, NDVI, and overgrown lawns, respectively. These relationships are visualized in the scatterplots in **Figure 2**, and the detailed output from the SAR models is given in **Supplementary Material 1**. In general, tree canopy cover and NDVI are higher in both low- and high-vacancy areas, and lower in block groups with intermediate vacancy rates. The prevalence of overgrown lawns increases with vacancy rates (**Figure 2**).

Patterns in Greenness and Occupancy Status at the Parcel Scale

The results from the two-way ANOVA tests are summarized in **Figure 3**. In the two-way ANOVA for tree canopy cover, overgrown status ($F = 20.5$, $p < 0.0001$) and occupancy class ($F = 299.7$, $p < 0.0001$) were both significant, as was their interaction ($F = 5.77$, $p = 0.0031$). The results of Tukey's HSD test indicated that tree canopy cover was 2.8% higher ($p < 0.0001$, 95% confidence interval (CI) of the difference = 1.1–4.5%) for occupied parcels that did not have overgrown lawns, there was no difference for parcels with vacant structures ($p = 0.99$), and there was no difference for vacant lots based on overgrown status ($p = 0.86$; **Figure 3A**). When lawns were not overgrown, tree canopy cover on vacant lots was 9.8% ($p < 0.0001$, CI = 8.3–11.3%) greater than parcels with vacant structures and 5.7% ($p < 0.0001$, CI = 4.9–6.4%) greater than occupied structures, and parcels with occupied structures had 4.1% ($p < 0.0001$, CI = 2.9–5.4%) greater tree canopy cover than vacant structures (**Figure 3A**). When lawns were overgrown, tree canopy cover was higher on vacant lots by 8.5% ($p < 0.0001$, CI = 5.7–11.2%) and 7.5% ($p < 0.0001$, CI = 4.8–10.3%) compared to vacant structures and occupied parcels, respectively, and there was no difference between vacant structures and occupied parcels (p=0.88).

In the two-way ANOVA for NDVI, overgrown status ($F = 221.6$, $p < 0.0001$) and occupancy class ($F = 7,153.2$, $p < 0.0001$) were both significant, as was their interaction ($F = 8.48$, $p = 0.0002$). The results of Tukey's HSD test for NDVI indicated that there was no difference between occupied parcels

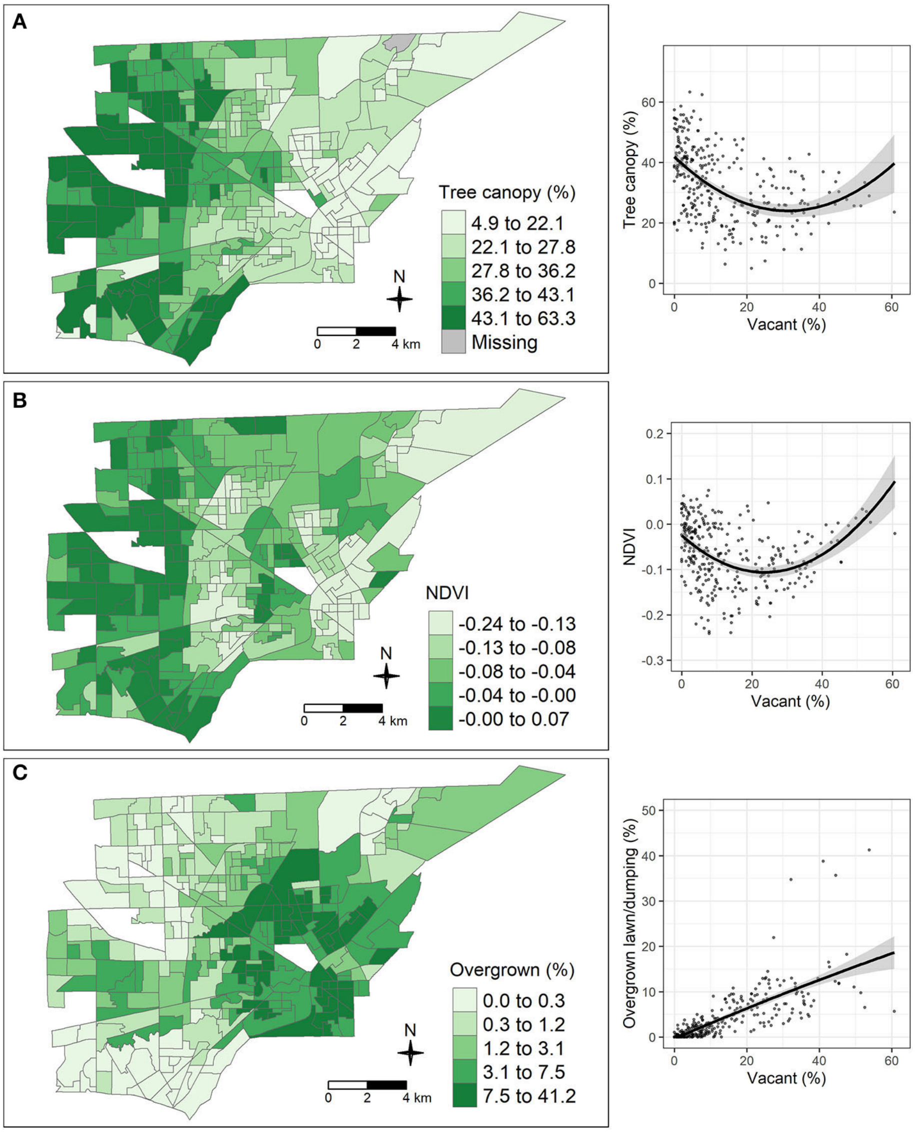

FIGURE 2 | Maps of greenness indicators and their respective relationships with percent vacant parcels (vacant structures plus vacant lots) within Toledo block groups. On the maps, filled polygons represent census block groups with ≥20 residential parcels. Within each block group, **(A)** tree canopy represents the mean percent canopy cover for all residential parcels, **(B)** NDVI represents the mean parcel NDVI value, and **(C)** overgrown lawn and/or dumping represents that percent of parcels noted as such during the Toledo Survey. On the scatterplots, block groups are represented by points, and quadratic regression lines are displayed because they yielded a better fit than linear models.

with and without overgrown lawns ($p = 0.99$), parcels with vacant structures had an NDVI value that was 0.015 higher for overgrown parcels compared to those that were not overgrown ($p = 0.0032$, 95% CI of the difference=0.003–0.026), and vacant lots had an NDVI value that was 0.017 higher for overgrown parcels compared to those that were not overgrown ($p = 0.0008$, CI = 0.005–0.029; **Figure 3B**). When lawns were not overgrown, NDVI values on vacant lots were higher than parcels with occupied and vacant structures by 0.163 ($p < 0.0001$, CI = 0.159–0.168) and 0.176 ($p < 0.0001$, CI = 0.168–0.183), respectively (**Figure 3B**). Parcels with occupied structures had NDVI values that were 0.012 ($p < 0.0001$, CI = 0.005–0.019) higher than parcels with vacant structures. For parcels with overgrown lawns, NDVI on vacant lots was higher than occupied parcels by 0.182 ($p < 0.0001$, CI = 0.168–0.197) and vacant structures by 0.178 ($p < 0.0001$, CI = 0.164–0.193). There was no difference between vacant structures and occupied parcels ($p = 0.93$).

We used spatial proximity and parcel land area criteria to match parcels in an effort to control for geographic influences other than occupancy status. On average, vacant structures were successfully matched with 2.86 occupied parcels, resulting in 12,460 total matches. The mean and median numbers of occupied candidates for each vacant structure were 53.3 and 41, respectively. Eighty-seven (or 2% of total) vacant structures

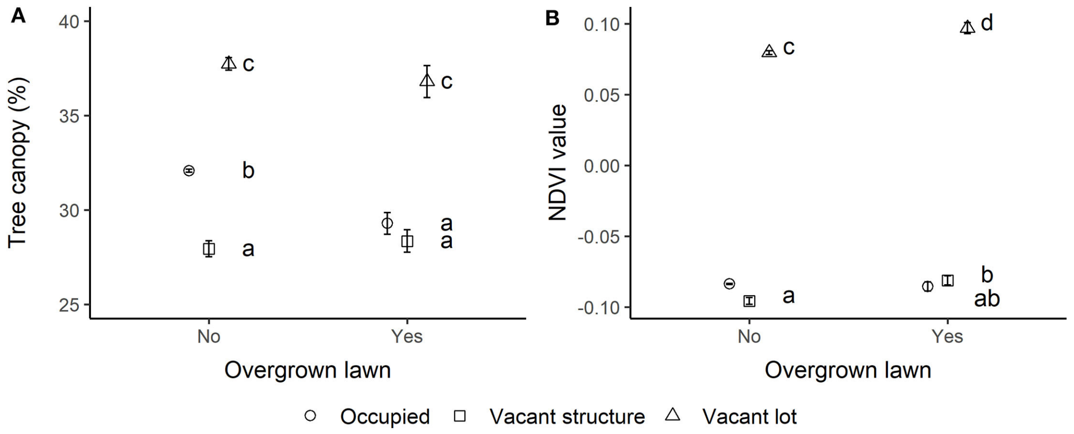

FIGURE 3 | Comparison of **(A)** tree canopy and **(B)** NDVI according to the interaction between occupancy class and overgrown status. Significant differences are indicated by different letters.

TABLE 1 | Comparison of vacant structure parcels and their occupied matches.

	Vacant structures	Occupied matches	Difference
Mean parcel size (m^2)	456.6	454.8	1.8
Mean tree canopy cover (%)	28.1	25.0	3.1*
Mean NDVI	−0.091	−0.132	0.041*
Overgrown lawn and/or dumping (%)	35.0	4.0	31.0*

*$p < 0.001$.

TABLE 2 | Comparison of vacant lot parcels and their occupied matches.

	Vacant lots	Occupied matches	Difference
Mean parcel size (m^2)	472.4	472.6	−0.2
Mean tree canopy cover (%)	37.7	26.6	11.1*
Mean NDVI	0.081	−0.126	0.207*
Overgrown lawn and/or dumping (%)	11.6	3.2	8.4*

*$p < 0.001$.

had no occupied matches, and they were not included in the analysis. There were no significant differences in parcel size between the vacant structures and their occupied matches, suggesting that the matching criterion was successfully met. Vacant structures were significantly greener than their occupied matches. Vacant structures had 3.1 percentage points more tree canopy than their occupied matches, significantly higher NDVI, and had an overgrown lawn rate that was 31 percentage points higher (**Table 1**).

In the paired parcel comparisons, the average vacant lot was successfully matched with 2.49 occupied parcels, resulting in 19,548 total matches. The mean and median numbers of occupied candidates for each vacant lot were 39.3 and 27, respectively. There were 954 vacant lots with no occupied matches (or 11% of total), and they were excluded from the analysis. Again, the matching procedure was successful, as the mean parcel size did not differ significantly between vacant lots and their occupied matches. Vacant lots had an average of 37.7% tree canopy cover, which was 11.1 percentage points greater than the occupied matches (**Table 2**). NDVI was significantly higher in vacant lots, and the prevalence of overgrown lawns was 8.4 percentage points higher on vacant lots compared to their occupied matches (**Table 2**).

Geographic Discrepancies in the Relative Contribution of Vacant Parcels to Neighborhood Greening

Mapping the results of the Getis-Ord Gi* analyses of tree canopy for vacant structures shows a heterogeneous geographic pattern with a small number of statistically significant clusters of strong and weak differences between vacant structures and occupied matches (**Figure 4A**). For tree canopy on vacant lots, cold spots indicating weak or negative differences between vacant lots and their occupied matches are concentrated in high-vacancy areas near downtown Toledo, while hot spots indicating larger differences between vacant lots and occupied matches are located further to the west (**Figure 4B**). For NDVI, the Getis-Ord G_i* analysis shows similar patterns for both vacant structures (**Figure 4C**) and vacant lots (**Figure 4D**). Stronger differences in NDVI between vacant parcels and occupied matches were clustered in high-vacancy areas near the center of Toledo, while weaker or negative differences were concentrated in lower-vacancy areas of western Toledo.

DISCUSSION

This study supports the notion that citywide heterogeneity in vacancy intersects with the uneven distribution of vegetation

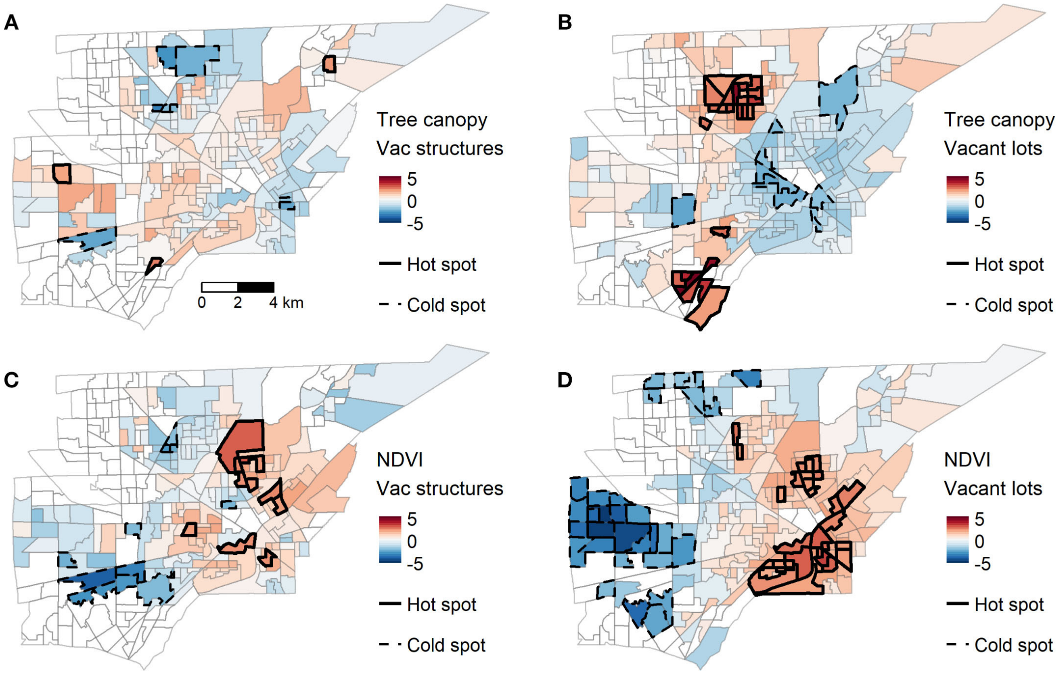

FIGURE 4 | (A–D) Results of Getis-Ord G_i^* analysis of the discrepancy in greenness between vacant parcels and their comparable occupied matches. Mapped values are *z*-scores, where darker reds indicate increasingly strong positive differences between vacant parcels and their occupied matches, and darker blues indicate differences that are increasingly smaller or negative. Hot (cold) spots are clusters of block groups that have values that are significantly higher (lower) than expected at the $p < 0.05$ level. Hot spots can be interpreted as areas where vacant parcels are particularly green compared to their occupied matches. Cold spots can be interpreted as areas where occupied parcels are either greener than or quite similar in greenness to vacant parcels. White block groups had <10 matched pairs and were excluded from the analysis.

in ways that are especially important to understanding distributional equity in shrinking cities. Using parcel-scale data to explore the connections among parcel occupancy status, vegetation abundance, and blight associated with overgrown lawns, our findings underscore the noteworthy contributions of vacant residential parcels to urban vegetation abundance, particularly in high-vacancy areas. Previous research by Schwarz et al. (2018) showed that high-vacancy areas in Toledo increased in vegetation over time, plausibly due to spontaneous vegetation growth on vacant parcels. They speculated that while this greening reduced the geographic disparity in the quantity of vegetation, it was likely that disparities in the quality or desirability of vegetation persisted across neighborhoods. The findings presented here add multiple lines of complementary evidence supporting this idea, and illustrate that vacancy plays a disproportionately large role in the greening of Toledo's high-vacancy areas. Moreover, much of this greening is driven by overgrown vegetation.

At the neighborhood scale, tree canopy and NDVI both exhibited non-linear associations with vacancy rates. Low-vacancy and high-vacancy block groups had the highest tree canopy and NDVI (**Figure 2**). This is consistent with earlier findings from Toledo at the census tract scale showing that high-vacancy areas have increased in vegetation over time, presumably as vacant parcels became greener due to building demolition and/or reduced vegetation maintenance (Schwarz et al., 2018). Note that the regression curve for NDVI peaked higher for high-vacancy block groups than for low-vacancy block groups, while the curve for tree canopy did not (**Figure 2**). This can plausibly be explained by the fact that NDVI captures low-lying vegetation like grasses and shrubs that are not included in the tree canopy classification, and so low-lying vegetation (weedy or otherwise) growing abundantly in high-vacancy block groups contributed to higher NDVI values. The positive association between vacancy and overgrown lawns (**Figure 2**) suggests that undesirable, weedy vegetation contributes to greening in higher-vacancy block groups. While there are opportunities to advance both social and ecological goals via planned greening initiatives on urban vacant land (Green et al., 2016; Anderson and Minor, 2017), the proliferation of overgrown vegetation in high-vacancy areas of Toledo (**Figure 2**) indicates that many vacant parcels have greened in a way that could produce negative social outcomes.

The parcel-scale analyses enrich the perspectives from the block group scale by explicitly accounting for occupancy status in comparisons of greenness metrics. An important finding is that vacant parcels were greener than their occupied matches according to tree canopy cover and NDVI (**Tables 1**, **2**). This was expected for vacant lots where building demolition creates additional space for vegetation to grow. But even on parcels with vacant structures we observed higher tree canopy and NDVI than on comparable occupied parcels (**Table 1**). Given that the prevalence of overgrown lawns is 31% higher for vacant structures than occupied parcels, it is likely that unmaintained vegetation on vacant structures is responsible for a substantial portion of this difference. Indeed, the two-way ANOVA tests showed that both vacant structures and vacant lots had higher NDVI values on parcels with overgrown lawns than those without overgrown lawns (**Figure 3**); this difference was not observed for occupied parcels. Tree canopy cover did not vary on vacant parcels depending on overgrown lawn status (**Figure 3**), so the convergence of evidence indicates that the differences in NDVI were driven by low-lying vegetation such as grasses and shrubs.

Vacant parcels are geographically concentrated in neighborhoods near downtown Toledo (**Figure 1**), where they will influence neighborhood greenness patterns based on their sheer abundance alone. Moreover, when we analyzed spatial patterns in the NDVI discrepancy between vacant parcels and their occupied matches, we found that high-vacancy areas had larger discrepancies (**Figure 4**). On a per-parcel basis, vacant parcels in high-vacancy areas contribute more to neighborhood greenness than vacant parcels in low-vacancy areas. While this pattern was clear for NDVI, there was almost no discernable pattern for tree canopy on vacant structures, and we saw the opposite pattern for tree canopy on vacant lots (i.e., cold spots were concentrated in high-vacancy areas) (**Figure 4**). We interpret this as an indication that the large discrepancies in NDVI between vacant and occupied parcels in high-vacancy areas were driven by the proliferation of low-lying vegetation on vacant parcels in high-vacancy areas; this interpretation is consistent with the findings presented in **Figures 2**, **3B**, and **Tables 1**, **2**. On the other hand, vacant parcels in low-vacancy areas did not contribute much to neighborhood greenness overall because there were not many vacant parcels in these areas; occupied parcels in these areas are relatively green in general, so vacant parcels are not particularly green in comparison; and there may be a neighborhood expectation to maintain vegetation on vacant parcels in low-vacancy areas (Krusky et al., 2015), such that these parcels are less likely to become overgrown.

This study highlights the value of incorporating multiple lines of evidence at multiple spatial scales to contextualize observed patterns in urban greenness. Whereas, earlier studies have been limited by data availability, our analysis benefited from parcel-scale data including qualitative data about overgrown lawns from the Toledo Survey. This allowed us to demonstrate that vacant parcels are greener than comparable occupied parcels (**Tables 1**, **2**), and that vacant parcels with overgrown lawns are greener than those without overgrown lawns (**Figure 3B**), a pattern that is not explained by tree canopy cover (**Figure 3A**). These conclusions would not have been apparent if we had studied patterns in tree canopy cover or NDVI at the neighborhood scale alone. While the Toledo Survey has demonstrable value for the practical management of land in Toledo, our research also shows the promise of such data sets for research purposes. Where comprehensive parcel condition data are not available, researchers have developed protocols to assess parcel condition that provide context to complement remotely sensed measures like tree canopy cover or NDVI (e.g., Krusky et al., 2015; Troy et al., 2016; Gobster et al., 2020). Such approaches can provide valuable data about human perception that inform whether vegetation functions as beauty or blight (Gobster et al., 2019).

There were limitations to our analyses that may impact our findings. First, the data sets upon which our analyses were based were captured at different times. The 2015 imagery used to calculate NDVI was acquired within 1 year of the Toledo Survey data, which was collected over a 12-month period. Mowing, trimming, and other management interventions may have altered the vegetation abundance and overgrown lawn status in the period between these data acquisitions. The tree canopy cover map was based on 2012 imagery, but its assessed accuracy was 91% with respect to 2015 reference imagery, so we deemed it suitable for use. Second, each data set captured conditions at a single point in time, and there may have been errors in the data sets such as miscategorized occupancy status in the Toledo Survey. However, with a large data set of over 100,000 parcels, we expect that representative patterns were apparent even in the presence of some error. Third, different members of the Toledo Survey field crew may have had different interpretations of what constitutes an overgrown lawn based on their neighborhood social norms and their personal preferences (Hostetler, 2020). Furthermore, we operationalized the Toledo Survey category "overgrown lawn and/or dumping" as an indicator of overgrown vegetation, but there were likely instances of dumping in which the vegetation was not overgrown. While we cannot control for such cases, the relationships between NDVI and the overgrown lawn and/or dumping indicator (**Figure 3B**) suggest that the indicator was frequently recording overgrown vegetation.

In Toledo, heterogeneity in vacancy and vegetation quantity and quality intersect in ways that point to inequitable social-ecological outcomes. Urban vegetation is often touted for providing ecosystem services (e.g., Seamans, 2013; Dobbs et al., 2014), and environmental justice analyses related to urban greening have generally assumed, implicitly or explicitly, that higher amounts of vegetation are desirable (e.g., Pham et al., 2012; Danford et al., 2014). However, urban vegetation does not always function as an amenity (Schwarz et al., 2018), and here we found that vacant parcels played a substantial role in neighborhood-scale greening in high-vacancy areas. So while vacant parcels may perform important ecological functions like stormwater control and urban heat island amelioration, they can simultaneously evoke negative perceptions of the neighborhood (Herrmann et al., 2016; Riley et al., 2018), creating a potential misalignment between ecosystem services and amenities. This may be particularly unjust in instances where overgrown vegetation in high-vacancy areas provides ecological benefits that are experienced citywide, while the overgrown vegetation simultaneously generates negative social outcomes that are localized within those high-vacancy areas, potentially perpetuating localized disinvestment. This highlights the need to understand both vegetation quantity and quality when assessing the equity of urban vegetation patterns (Schwarz et al., 2018).

Future work in this area of research could further advance our understanding of drivers and consequences of vacancy and unequal vegetation distributions. For example, longitudinal data sets would permit assessment of how patterns in greenness change over time following changes in occupancy status. Incorporating parcel data about ownership could shed light on whether vegetation maintenance varies depending on ownership; this type of information could inform land banks as they work to return vacant parcels to uses that are positive for the community. We have demonstrated clear connections between vacancy, vegetation abundance, and overgrown vegetation functioning as blight in Toledo. Conducting similar studies in additional cities would indicate whether these patterns are generalizable across shrinking cities, as well as in areas of disinvestment within cities that are generally prosperous. Vacant parcels can play a key role in increasing the amount of vegetation in shrinking cities. However, it is not only the quantity, but also how that vegetation is experienced by the community in which it is embedded, that determines whether "greening up" is resisting or perpetuating environmental inequities.

DATA AVAILABILITY STATEMENT

Publicly available datasets were analyzed in this study. This data can be found at: Toledo parcel data were obtained from the Lucas County Auditor (http://co.lucas.oh.us/331/Auditor). Toledo Survey data were obtained from the Lucas County Land Bank (https://lucascountylandbank.org/). Xi Li provided the urban tree canopy classification data that was developed for her Ph.D. dissertation (http://rave.ohiolink.edu/etdc/view?acc_num=toledo1493417634359152). NDVI was calculated based on 2015 NAIP imagery available at https://coast.noaa.gov/dataviewer/#/.

AUTHOR CONTRIBUTIONS

AB, DL, KS, and DH designed the study. AB prepared data and conducted analysis. AB, DH, and DL generated visualizations. AB, KS, DL, and DH wrote the paper. All authors contributed to the article and approved the submitted version.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fevo.2020.566759/full#supplementary-material

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Street Tree Density and Distribution: An International Analysis of Five Capital Cities

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Municipal leaders around the world are demonstrating significant interest in urban greening to realize a range of socioecological benefits. The urban greening toolkit often includes street trees, an essential component of urban design informed by historic legacies of both human and environmental factors. To date, there has been little comparative analysis of street tree density and distribution across international and intercontinental settings, and associated research has not been situated within the broader discussion of historical legacies. This study focuses on five capital cities (Ottawa, Stockholm, Buenos Aires, Paris, and Washington, DC) situated in two climate zones and it addresses two research questions: (1) What are the density and distribution of street trees across a given city and its street hierarchy? and (2) How do these metrics compare within and between cities by climate zone? The analysis draws upon up-to-date datasets from local authorities and includes geospatial analysis of street trees across hierarchical street classes within the central zones of each city. The results show clear differences in street tree density in cities within and between climate zones as well as differences in street tree distribution in cities within the same climate zone. Substantial differences within climate zones further suggest that cultural factors—including but not limited to urban form, aesthetic norms, and governance regimes—may play a pivotal role in the distribution and density of street trees. This illustrates the importance of place-specific cultural and environmental legacies as determinants of street tree density and distribution and supports further comparative research on the topic.

Keywords: street trees, legacy effects, urban greening, comparative analysis, globalization, urban history

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INTRODUCTION

Urban greening is a common practice around the world today that aims to realize a range of socioecological benefits. Urban greening activities involve organized or semi-organized efforts to introduce, conserve, and maintain outdoor vegetation in urban areas (Eisenman, 2016; Feng and Tan, 2017). Such efforts take on a multitude of material expressions, municipal policies, and incentives (Beatley, 2017; Tan and Jim, 2017). This includes large-scale urban tree planting initiatives in which street trees figure prominently (Young, 2011; Roman et al., 2015; Breger et al., 2019). Scholars have described these campaigns as an urban forestry movement (Campbell, 2017) and a popular trend (Pincetl et al., 2013). Of note, the systematic planting of trees across the urban fabric and along streets was not common in many European and North American cities until the nineteenth and early twentieth centuries (Campanella, 2003; Lawrence, 2006; Dümpelmann, 2019).

Today, street trees are a prominent type of flora in the urban greening toolkit and they are an essential component of urban design. They define the space of a street, delimit the pedestrian realm, calm traffic, filter sunlight, promote visual order, soften the streetscape, and introduce beauty in the form of nature (Massengale and Dover, 2014). They are also an important component of urban forestry practice. Street trees are the primary focus of many urban tree inventories (Keller and Konijnendijk, 2012), and in the United States, expenditures on street trees account for the largest portion of municipal tree management budgets, eclipsing the amount spent on park trees by a factor of 2.7 (Hauer and Petersen, 2016). This is noteworthy, as street trees only constitute 10% of urban trees by population and 25% by leaf area (Miller et al., 2015) in the United States. In dense settlements, street trees may constitute a greater portion of the urban canopy than in less dense areas (Pham et al., 2013).

Current patterns of urban tree distribution (or structure) and species composition are a legacy of both human and environmental (or biophysical) factors (Roman et al., 2018). Environmental legacies include extreme weather events, wildland urban interface fires, and outbreaks of pest and disease. Human legacies include those of historical periods such as national and regional identity, colonial history, species symbolism, and urban park and city beautification movements as well as long-term changes in neighborhood form and socioeconomic demographics. These legacies are set within a bioregional context—native biome, climate, topography, and preexisting vegetation and land use—that establishes the environmental conditions for the development of cities comprising socio-ecological systems that are built by and for humans (Groffman et al., 2014). A city's urban form may, in turn, respond to both environmental conditions such as topography and landscape setting, as well as cultural drivers such as military defense (e.g., road widths, medieval moats and walls), political and economic control (e.g., the grid), periodic trends (e.g., baroque street diagonals, freeways-to-greenways), public policy (e.g., urban renewal), and technological and socio-demographic change (e.g., automobile infrastructure, suburban expansion) (Kostof, 1991; Birch, 2009; Horte and Eisenman, 2020). Because of this, urban form varies widely across geography and culture (Huang et al., 2007; Berghauser Pont et al., 2019). This suggests a need to understand cities as distinct biomes that can be classified by typology to better inform urban greening aspirations (Pincetl, 2015). The distribution of urban trees may, in turn, differ between cities and this is likely due to local legacy effects (Roman et al., 2018).

Street tree inventories are commonplace in urban forestry practice and research, and they typically address a range of factors including species composition, age, vigor, size, management costs, inventory methods, and estimates of ecosystem services such as carbon sequestration (Sjöman et al., 2012; Tanhuanpää et al., 2014; Strunk et al., 2016; Tigges et al., 2017). However, there have been relatively few studies focusing on the spatial dimension of street tree density and distribution to date. In this study, street tree density is defined as the number of trees per 100 m of street length and street tree distribution is defined as the number of trees per 100 m of street length across hierarchical street classes. This study addresses street tree density and distribution as indicators of environmental and human legacy effects, and it builds upon related scholarship. Kuruneri-Chitepo and Shackleton (2011) calculated street tree density in the Eastern Cape, South Africa using 200 m transects to highlight distributional disparities between different towns. They found that relatively more affluent suburbs in these towns had larger mean street tree densities. Gwedla and Shackleton (2017) also calculated street tree density in multiple towns of the Eastern Cape using 200 m transects and found that both smaller towns and those that had been marginalized during apartheid had significantly lower street tree density and diversity. Nagendra and Gopal (2010) sampled 200 m transects across Bangalore, India to analyze the relationship between street tree density and narrow, medium-width, and wide roads, while Shams et al. (2020) sampled 100 m transects to do the same in Karachi, Pakistan. Both studies found that narrower roads had fewer street trees than their wider counterparts. Deb et al. (2013) studied street trees in Sylhet City, Bangladesh to compare density and distribution between main roads and link roads, and found that, on average, the former exhibited higher tree density than the latter. In these studies, street tree density was defined as the measure of trees per unit distance (Nagendra and Gopal, 2010; Shams et al., 2020) whereas street tree distribution was defined as the measure of such trees across space (Nagendra and Gopal, 2010), and, more specifically, across street types.

These studies illustrate the influence of cultural variables such as economic factors, development history, and street type on street tree density and distribution. This literature has, in turn, assessed street tree density and distribution in individual cities, or cities within the same region of a country. However, there has been little comparative analysis of street tree density and distribution across international and intercontinental settings and these metrics have not been included in broader discussions of legacy effects. This is noteworthy because there is substantial evidence that both national and continental contexts inform urban tree discourse, practice, and preferences (Fraser and Kenney, 2000; Campanella, 2003; Lawrence, 2006; Konijnendijk, 2008; Keller and Konijnendijk, 2012; Shackleton, 2012; Dümpelmann, 2019), as well as urban ecology writ large (Ernstson and Sörlin, 2019).

In response to the aforementioned gaps, this article addresses two research questions: (1) what are the density and distribution of street trees across a given city and its street hierarchy? and (2) how do these metrics compare within and between cities by climate zone? The analysis of these metrics introduces broader questions of how environmental and human legacies inform the spatial structure of urban trees. The study includes five capital cities—Ottawa, Stockholm, Buenos Aires, Paris, and Washington, DC—which have shared characteristics including sites of display and monumentality, places of tourist pilgrimage, and hosts to diplomatic quarters (Gordon, 2006). The cities are located in two climate zones as defined by the Köppen–Geiger classification system (Kottek et al., 2006): Ottawa and Stockholm are in the Dfb climate zone (snow, fully humid, warm summer) while Buenos Aires, Paris, and Washington, DC are in the relatively similar Cfa and Cfb climate zones (warm temperate, fully humid, and hot/warm summer, respectively). This provides for relative control of climatological factors, allowing sociopolitical and morphological factors to emerge. An underlying tenet of the study is that street patterns are closely related to urban form and that a streetscape therefore serves as the physical scaffolding upon which street trees are planted and maintained. Thus, the study employs a common transportation engineering classification scheme of *local*, *collector*, and *arterial* streets to assess street tree distribution in each city. This classification scheme responds to city-specific traits such as traffic volume, speed, and building density; and while road width generally increases from local, to collector, and arterial streets, this functional classification does not translate to universal street design standards (USDOT, 2013; Massengale and Dover, 2014; European Commission, 2020). However, the classification scheme is commonly employed by municipalities and it is a helpful tool for understanding the spatial distribution of urban street trees.

MATERIALS AND METHODS

Study Areas and Data Sources

As noted above, this study involves an analysis of the density and distribution of street trees in five capital cities that are located in two Köppen–Geiger climate zones. Publicly accessible street tree inventories and street hierarchy classification systems were used to conduct a geospatial analysis of street tree density and distribution.

Each of the selected cities is situated in a larger metropolitan region. Central zones were identified in each city and are characterized as medium- to high-density with high frequency of street use and diverse land use (Hillier et al., 1993; Berghauser Pont et al., 2019). The entire municipal boundaries of Buenos Aires (*Ciudad Autónoma de Buenos Aires*), Paris (*Paris Intra-Muros/Département 75*), and Washington, DC (*District of Columbia*) were designated as central zones while in Ottawa, the central zone was defined by the *urban wards* and in Stockholm the central zone was defined by the *innerstan* (inner-city). The resulting central zone designations are henceforth referred to by the names of the five cities. Maps of the central zones and street hierarchies for each city are provided in **Figure 1**.

Urban tree inventory research has emerged over the past four decades and has grown to encompass a variety of methods, including satellite-assisted data collection, airplane-assisted data collection, and both ground-level photography and manual visual inspection methods (Nielsen et al., 2014). In our study, spatial data for trees, streets, jurisdictional areas, and bodies of water (used to identify bridges) in each city were obtained from open data portals and third-party sources including public institutions, municipalities, national governments, and research organizations (see **Table 1**). The exception to this is the street dataset for Paris, the content of which was sourced from OpenStreetMap (OpenStreetMap, 2019).

The datasets for each of these cities are routinely updated and the most recent version of each inventory is from 2019. There is variability in the methods of data collection by each locality; this constitutes one of the inherent challenges of doing comparative research across cities and countries (Keller and Konijnendijk, 2012; Östberg et al., 2013; Cowett and Bassuk, 2014). For example, the tree inventory in Ottawa is updated through daily operations by multiple city departments when new trees are planted, which occurs through a number of planting programs. Private contractors are, in turn, responsible for accurately reporting the establishment of new trees to the municipality's planning department (Urban Forest Innovations Inc. et al., 2017). The city's inventory includes trees located on city-controlled public land except for new subdivision development[1]. The tree inventory in Stockholm was retrieved from a city-wide mapping database that is continuously updated using satellite-supported control points and photogrammetry (City of Stockholm, 2019); it contains at least 90% of the street trees in the inner city beyond those that are newly planted or located in newly built areas[2]. In Buenos Aires, the tree inventory data is the product of annual on-site surveys of the city's trees[3]; officials in this city did not respond to inquiries about completeness. Launched in 2001, the tree database for the City of Paris is updated weekly when trees are serviced and contains all the street trees planted in the city's public domain[4] (City of Paris, 2019). Tree inventory data in Washington, DC is continuously updated (District of Columbia Department of Transportation [DDOT], 2020); it contains all the trees under its management purview and is at least 95% complete according to city officials[5].

Data Collection and Analysis

Data were processed in ArcMap to categorize each city's street network and to facilitate quantitative analysis of street tree density and distribution in Excel and MATLAB. There was variation in the definition of the urban core and the street classification system for each city. For example, the urban core of Ottawa was derived from the twelve urban wards (constituting a land area of 323 km^2), while the urban core of Stockholm was derived from four inner city districts (constituting a land area of 36 km^2). Ottawa and Stockholm are the largest and smallest urban cores in the study with Buenos Aires (204 km^2), Paris (103 km^2) and Washington, DC (158 km^2) having intermediate spatial extents. With respect to street classification, the number of functional classes ranged from 4 in Buenos Aires to 32 in Paris.

To establish a consistent hierarchy of streets, the street network in each city was normalized to three classes commonly used in traffic engineering: local, collector, and arterial (USDOT, 2013; European Commission, 2020). In all cities except Paris,

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[1] Genevieve Raymond (Section Manager at the City of Ottawa), email to Nicholas Smart, 24 August 2020.

[2] Jennifer Gustavsson (Engineer at the City of Stockholm), email to Nicholas Smart, 1 September 2020.

[3] City of Buenos Aires, email to Nicholas Smart, 18 February 2020.

[4] Rederic Toussaint (Chef de la Cellule Méthode et Patrimoine), email to Nicholas Smart, 5 October 2020.

[5] Earl Eutsler (Associate Director at DDOT Urban Forestry Division), email to Nicholas Smart, 31 August 2020.

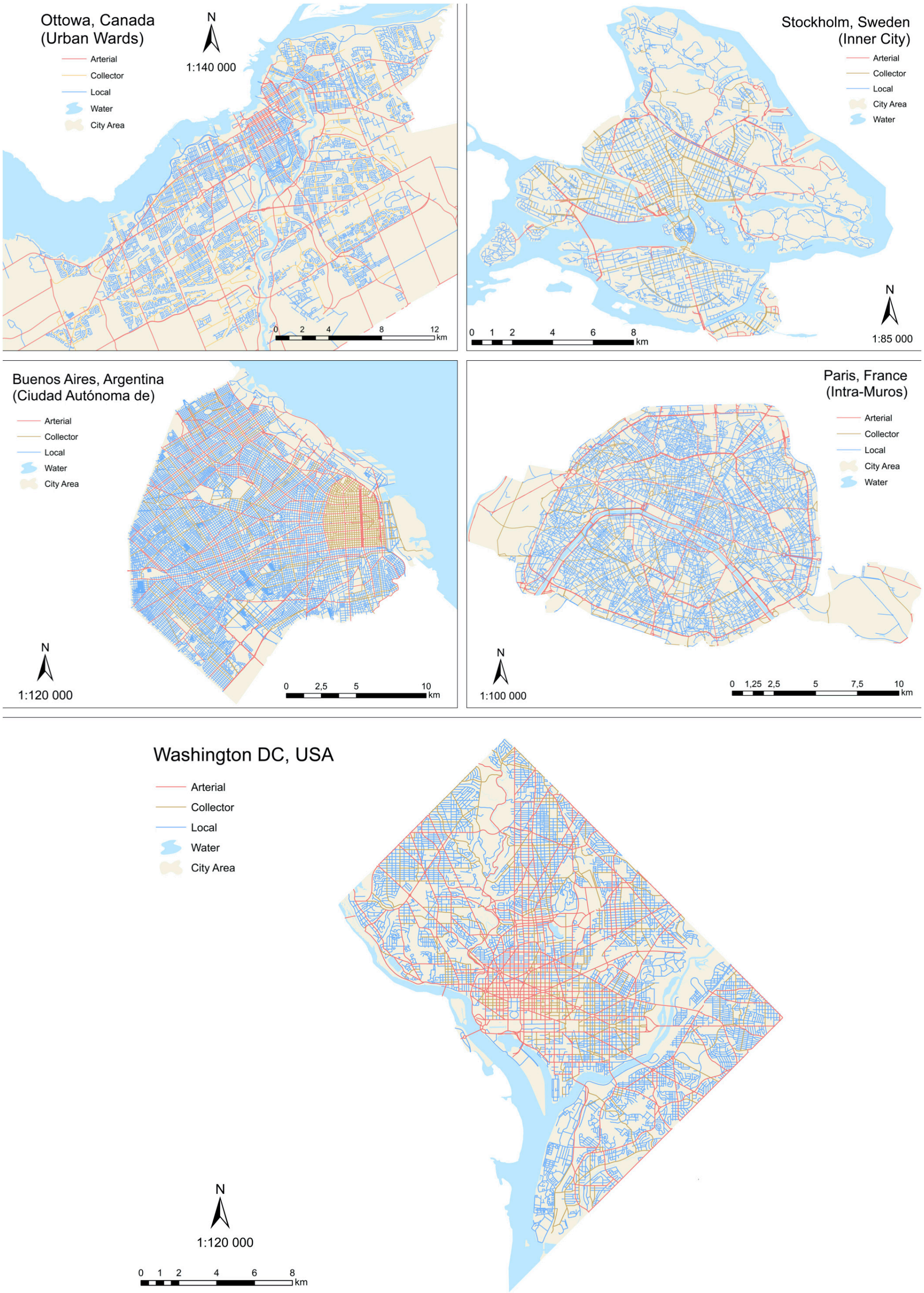

FIGURE 1 | Maps of the central zones and street hierarchies for the five cities in the study.

these functional classes were provided by local authorities. In Paris, OpenStreetMap data was used because the local authority does not use functional classes. The determination of the classification of street segments in each city was based on the cities' respective GIS attribute tables that most closely corresponded to a hierarchical classification of local, collector, and arterial streets. For example, in the local inventory created in Ottawa, streets were separated into 14 subclasses; in Stockholm, streets were graded numerically into 10 classes; in Buenos Aires, five hierarchical network categories were used; in Washington, DC, seven functional labels were used; and in the open-source data used in Paris, 32 roadway type labels were used. After

TABLE 1 | Sources of spatial data for the five cities.

City	Trees	Streets	Jurisdictional area	Bodies of water
Ottawa	City of Ottawa	City of Ottawa	City of Ottawa	City of Ottawa
Stockholm	City of Stockholm	The Swedish Transport Administration	City of Stockholm	City of Stockholm
Buenos Aires	Jefatura de Gabinete de Ministros	Ministerio de Educación e Innovación	Instituto Geográfico Nacional	Instituto Geográfico Nacional
Paris	Ville de Paris	OpenStreetMap	Ville de Paris	Institut National de l'Information Géographique et Forestière
Washington, DC	District Department of Transportation	District Department of Transportation	United States Census Bureau	United States Census Bureau

TABLE 2 | Local authority definitions of street trees in each city.

City	Street tree definition
Ottawa	"Any tree located completely on the right-of-way or jointly owned between the city and the property owner (where any part of the tree trunk is growing across properties)"[a]
Stockholm	Groups of trees or solitary trees within the public place boundary on streets, allés, and in squares[b]
Buenos Aires	Those located on urban streets and sidewalks within the jurisdictional bounds of the city[c]
Paris	[No definition identified]
Washington, DC	[No definition identified]

[a]Genevieve Raymond (Section Manager at the City of Ottawa), email to Nicholas Smart, 25 October 2019. [b]Jennifer Gustavsson (Engineer at the City of Stockholm), email to Nicholas Smart, 22 October 2019. [c]City of Buenos Aires, email to Nicholas Smart, 6 November 2019.

reclassifying these heterogenous labels into local, collector, and arterial streets, the data in each city was ground truthed using Google Earth and Google Maps to confirm that the street classes were consistent across the cities. The resulting street classes are included in the maps of each city (**Figure 1**).

Official street tree definitions were identified in three of the five cities from a review of publicly available online information and through inquiries to municipal representatives. The definitions from the three cities are fairly consistent and designate street trees as those trees located in the public rights-of-way on city streets (see **Table 2**). A similar interpretation is assumed for the two cities that do not have an official definition.

To reduce the heterogeneity of data derived from our cities, two strategies were employed. The first of these concerns street datasets, which were cleaned by removing irrelevant street segments representing highways, tunnels, and bridges (which rarely include street trees). Bridges were assumed to be portions of street segments that spanned bodies of water, and were removed by means of geographic overlay, allowing for a more granular quantification of street network length. The second method to reduce heterogeneity across the data concerns street tree data obtained from local authorities, which was cleaned by first removing datapoints categorized as non-street trees, subsequently removing datapoints flagged as dead, non-existent, or duplicate, and finally by applying a buffer to ensure that all trees were located within 30 m of roadway centerlines (see **Figure 2**). The value of 30 m was chosen as a compromise between the need to remove trees that were not in close proximity to streets and to include trees that lined wide streets. It is important to note that the existing street tree datasets in each city were useful to streamline the analysis, but such data is not readily available in all cities. Where street tree datasets are not available, it would be necessary to develop the dataset by conducting satellite data analysis or a manual visual inventory.

In each city, geospatial analysis was used to calculate the street tree density for each street segment as well as to identify the coordinate points of each segment. The coordinate points corresponding to typical segments were then used in Google Street View to obtain visual representations with minimal seasonal variation for each hierarchical class across all cities. The visual representations were used to illustrate the mean street tree density for each class in each city as experienced by the "peripatetic subject" (Hillier et al., 1993).

Data were graphed for comparative analysis using Excel and MATLAB, and subsequently tabulated with imagery from Google Street View. Street tree point data were spatially joined to roadway segment polyline data in ArcMap to produce a segment-based density metric. The length of a given segment was used to produce a weighted mean density for each street class within a city as well as a city-wide weighted mean across all street classes. To assess the distribution of street trees across a given city, these weighted mean density values were plotted in an Excel bar chart and tabulated with its typical street image to provide an *in situ* visual depiction to allow for comparison of tree density by street class.

RESULTS

Comparison Across Climate

The street tree density for the three street classes in each city as well as a weighted mean for the entire city are summarized in **Figure 3**. Buenos Aires had the highest street tree density, while Stockholm had the lowest street tree density. The difference

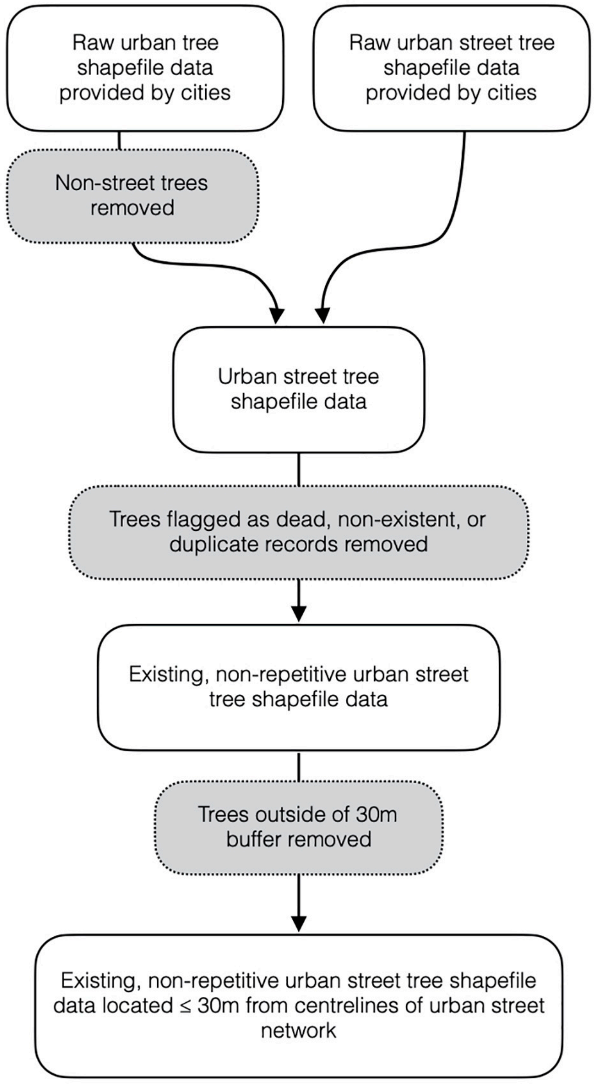

FIGURE 2 | Street tree data filtration process.

in city-wide weighted mean street tree density between these two cities is almost 10-fold. Results in **Figure 3** show that cities in the warmer climate zones (Cfa/Cfb) in this study generally exhibit greater street tree density city-wide as well as across most street classes. Buenos Aires, Paris, and Washington, DC averaged 9.9, 4.9, and 7.3 trees/100 m, respectively. In the cooler climate zone (Dfb), Ottawa and Stockholm averaged 3.5 and 1.0 trees/100 m, respectively.

In the cooler climate zone, the city-wide density of Ottawa (3.5 trees/100 m) was over three times greater than that of Stockholm (1.0 trees/100 m). Much of the difference in the city-wide density of the cities can be attributed to the local street class in the two cities (Ottawa averaged 4.3 trees/100 m and Stockholm averaged 0.8 trees/100 m), in contrast to the similar densities of the arterial street class in each city (1.1 and 1.0 trees/100 m, respectively). In the warmer climate zone, Buenos Aires, Paris, and Washington, DC exhibited marked differences in street tree density on local streets (10.6, 3.6, and 7.5 trees/100 m, respectively) but similar densities on collector streets (8.7, 9.2, and 7.9 trees/100 m, respectively) and arterial streets (7.3, 7.5, and 6.6 trees/100 m, respectively).

Statistical Distribution of Street Tree Density by Segment

Figure 4 includes box and whisker plots to compare and contrast the statistical distribution of street tree density for each street segment. Across the five cities, the street segments in Buenos Aires had the highest density, with 50% of its segments having 4.9–13.7 trees/100 m. This was followed closely by Washington, DC with 50% of street segments having 4.1–10.6 trees/100 m. Ottawa and Paris had similar distributions and the lower 25% of the segments studied in each of these cities had no street trees. The middle 50% of street segments in Ottawa had 0.0–6.5 trees/100 m as compared to Paris with 0.0–5.9 trees/100 m. Stockholm was an anomaly among the five cities with the majority of street segments (76.4%) having no trees.

Comparison Across Street Class

Google Street View images representing the street tree density for each street class are presented in **Table 3**. This is the same data as included in **Figure 3** with the addition of images to provide a visualization of street tree density values for each street class. They serve as a visual representation of what a person might experience while standing in the middle of the street with their gaze aligned with its axis. The images show distinct differences in visual experience depending on the street tree density and street class.

The data reveal distinct differences in street tree density across the street classes, though no conclusive patterns can be defined for a given street class. Collector streets exhibited the highest street tree density in three out of five cities. The arterial street class in Paris exhibited the highest street tree density (7.5 trees/100 m) of all cities, and this is illustrated in the typical westward-looking view of Boulevard Lefebvre. Likewise, the collector class streets in Paris exhibited a higher street tree density (9.2 trees/100 m) than the collector street class of the other cities, as shown in the southwest-looking image of Boulevard Henri IV. In contrast, the local street class of Buenos Aires had a higher street tree density (10.6 trees/100 m) than the local street class of the other cities, as shown in the northeast-looking image of Camarones.

Arterial streets in Ottawa had a lower street tree density than the collector and local streets. The local streets comprise 66% of the total street network and the average of 4.3 trees/100 m is almost four times greater than the arterial streets (1.1 trees/100 m). Stockholm's collector streets (2.1 trees/100 m) had a higher street tree density than local and arterial streets (0.8 and 1.0 trees/100 m, respectively). Reflecting the city's large number of treeless street segments, the *median* value of all street classes in Stockholm is 0 trees/100 m. Buenos Aires' local streets had a slightly higher street tree density (10.6 trees/100 m) than its collector (8.7) and arterial (7.3) streets. The street network of Buenos Aires is largely composed of local streets (74% by

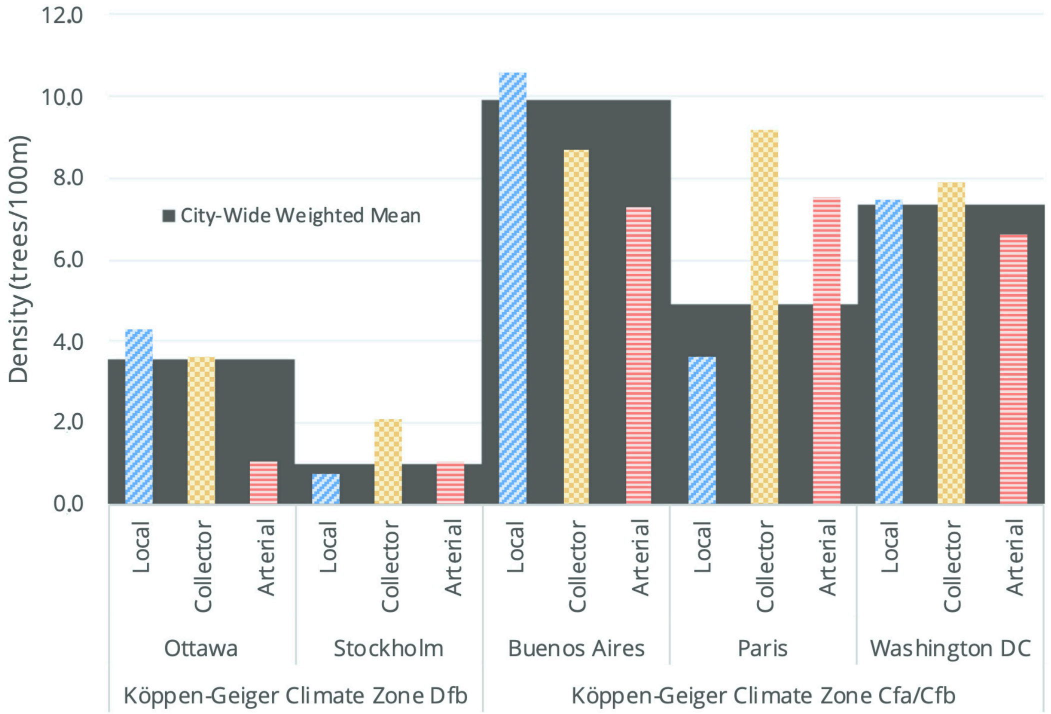

FIGURE 3 | Weighted mean street tree density by street class, city, and climate zone.

network length) and has a street tree density of 10.6 trees/100 m. Collector streets in Paris had a higher street tree density (9.2 trees/100 m) than local and arterial streets, although local streets account for a larger proportion (73%) of the city's total street network, and have a much lower weighted mean (3.6 trees/100 m). Outliers in the Paris dataset are abundant and thus, there is a wide range of street tree densities on individual street segments (**Figure 4**). In Washington, DC, local, collector, and arterial streets exhibited comparable linear street tree densities (7.5, 7.9, and 6.6 trees/100 m, respectively), exhibiting the most consistent street tree distribution of all of the cities studied. As with the other cities, the local streets accounted for the largest proportion (62%) of the city's street network by length and have the highest weighted mean (7.5 trees/100 m).

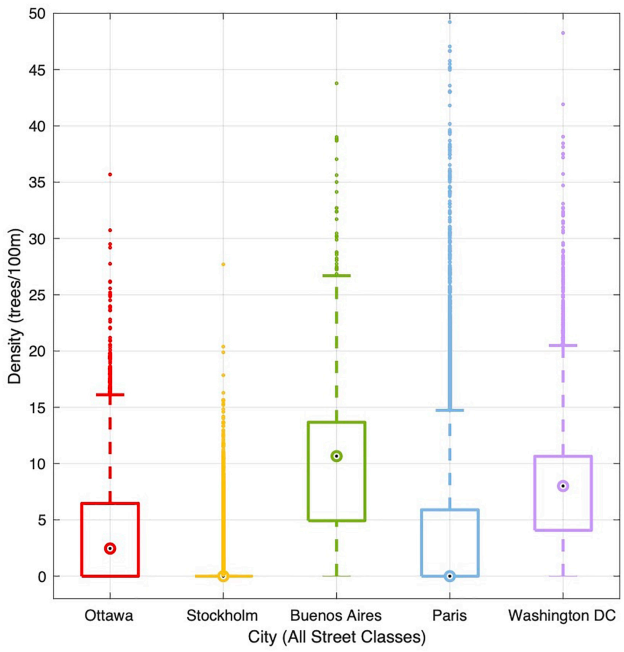

FIGURE 4 | Statistical distribution of street tree density by street segment.

DISCUSSION

This study illustrates clear differences in urban street tree density (the number of trees per 100 m of street) between the two climate zones, as well as within the same climate zone. The street tree density in the three warmer climate zone cities (Cfa/Cfb) ranged from 4.9 to 9.9 trees/100 m, while in the two cooler climate zone cities (Dfb) it ranged from 1.0 to 3.5 trees/100 m—a notable difference between and within climate zones. Within the cooler climate zone, the streets of Ottawa were on average 3.5 times as tree-dense as the streets of Stockholm, and in the warmer climate zone, the streets of Buenos Aires were on average two times as tree-dense as Paris.

The findings also show clear differences in the distribution (the number of trees per 100 m across hierarchical street classes) of street trees in cities located in the same climate zone. In Ottawa,

TABLE 3 | Typical street view depiction for street tree density (trees per 100m) by street class.

Köppen–Geiger	City (mean)	Local	Collector	Arterial
Dfb	*Ottawa* (3.5)			
	Stockholm (1.0)			
Cfa/Cfb	*Buenos Aires* (9.9)			
	Paris (4.9)			
	Washington, DC (7.3)			

street tree density decreased moving up the hierarchy of street classes (from 4.3 trees/100 m on local streets to 1.1 trees/100 m on arterial streets), while Stockholm exhibited a higher street tree density in its collector streets (2.1 trees/100 m) compared to its local (0.8 trees/100 m) and arterial (1.0 trees/100 m) streets. Street tree density in Buenos Aires decreased for larger streets (from 10.6 trees/100 m along local streets to 7.3 trees/100 m along arterial streets) in the same way as Ottawa, while the collector street class of Paris had a higher tree density (9.2 trees/100 m) than its local (3.6 trees/100 m) and arterial (7.5 trees/100 m) streets, similar to Stockholm. Washington, DC, on the other hand, exhibited comparable street tree densities across its street hierarchy (7.5, 7.9, and 6.6 trees/100 m).

While this study comprises a small sample size and caution should be exercised in generalizing findings beyond the cases covered, the differences in street tree density and distribution in these five cities is noteworthy. This points to the strong influence of place-specific legacies. Legacy effects can include a broad range of environmental and cultural drivers (Roman et al., 2018), and to understand the scope of these longitudinal forces requires in-depth historical analysis (see for example Campanella, 2003; Lawrence, 2006; Roman et al., 2018; Laurian, 2019). Such an assessment is beyond the scope of this paper. Having established that substantial differences in street tree density and distribution do exist in the cities investigated in this study, we now offer a preliminary discussion of potential legacies that may explain these findings, with the understanding that this initial inquiry is exploratory and by no means comprehensive.

Legacy Effects

As noted in **Figure 3** and **Table 3, Ottawa** and **Stockholm** have the lowest street tree densities of the five study cities. At first glance, it may seem that environmental factors explain why these cities in the Dfb zone (characterized by a snowy climate, fully humid conditions, and warm summer temperatures) have fewer street trees than the three cities located in the Cfa/Cfb zone (characterized by a warm temperate climate, fully humid conditions, and hot/warm summers). But this interpretation may not be so straight-forward, as Ottawa and Stockholm both have cultural legacies that played a role in the relative dearth of street trees. In Ottawa, for example, street trees were viewed as a "veritable nuisance" in the early to mid-twentieth century, with over 4,000 being cut down between 1921 and 1945 (Dean, 2005, p. 46). This decline is attributed to several intersecting forces: the maturation of large street trees planted in the late nineteenth century, the concomitant rise of professional arboriculture, and increased competition for space on city streets with sidewalks, paving, and utility infrastructure. The lack of trees along most of Stockholm's streets is also noteworthy, and here too, it may be tempting to associate this with the city's climate, in which shaded streets are not advantageous in the cold and dark winter months (Mcbride, 2017). But as described by Lawrence (2006), Scandinavian cities relied upon architecture and open squares rather than trees to enhance the public realm in the eighteenth century during a time when their European counterparts were planting grand avenues and boulevards. The lack of street trees in Stockholm may also be attributed to a longstanding tradition of centralized urban planning (Andersson and Bedoire, 1988). This contrasts with the decentralized approach to street tree planting that has been commonplace in North America. For example, Dean (2005) notes that most street trees in Ottawa were primarily planted by residents in the nineteenth century. And in the United States, street tree planting has historically been undertaken by a diverse network of actors including civic improvement associations, entrepreneurial individuals, non-profit organizations, private property owners, volunteers, and public sector employees (Summit and Sommer, 1998; Campanella, 2003; Lawrence, 2006; Campbell, 2014; Konijnendijk van den Bosch, 2014). In other words, it is possible that more decentralized governance regimes yield greater density and distribution of urban street trees.

Underlying urban form may also be a factor. In Buenos Aires, street trees are abundant and hierarchically distributed, with local streets displaying the highest tree density. Laid out in 1580 on an orthogonal street grid typical of the "new world" in North and South America (Contreras, 2014), Buenos Aires in the nineteenth century was inspired by Parisian planning—including tree-lined boulevards—to become the "Paris of America" (Gutiérrez, 2002). It was not until the earlier part of the nineteenth century, however, after the immigration of technical experts from Europe, that planning of the city's green infrastructure was realized (Gutiérrez, 2002; Benito et al., 2018). As noted in **Figure 1**, the historic core is dominated by collector and arterial streets, which generally have fewer trees than their local counterparts. Importantly, when many street trees were planted in the city in the nineteenth century, the streets of the historic core were deemed too narrow to accommodate street trees, which is why the modern business district lacks trees today (Márquez and Fiorentino, 2007).

In Paris, the study findings show that collector and arterial streets are between two and three times as tree-dense as local streets (see **Figure 3** and **Table 3**). This may be due to changes in urban form and street design driven by shifting priorities and technological changes spanning two millennia, as Paris has roots traceable to 52 B.C.E. (Bournon, 1888). Here, the first urban trees were planted in the fourteenth century in public spaces such as churchyards and marketplaces because the streets were intended for pedestrians and were too narrow to accommodate trees. In the seventeenth century, however, boulevards and avenues were developed on the edge of town to provide open spaces for the upper class to socialize and to facilitate the movement of troops. In the nineteenth century, efforts to improve vistas and public health led to the replacement of large swaths of the city with linear, tree-lined boulevards. These tree-lined streets continue to be an important historical legacy of the city today (Laurian, 2019).

In Washington, DC, by contrast, street tree distribution is fairly even across all street types and the city has on average 2.4 more trees per 100 m street segment than Paris (see **Figure 3** and **Table 3**). This is noteworthy because the city's physical plan was developed by the French architect Pierre L'Enfant (Bednar, 2006), and the French capital city is often portrayed as a European model for the U.S. counterpart (Bednar, 2006; Dümpelmann, 2019). Yet, Washington, DC exhibits a density and distribution of street trees that is markedly different from that of Paris, where trees are unevenly distributed across street types. These distinctions are a likely outcome of the two cities' dramatically different histories and their influence on urban form. Moreover, the distinctions may also be influenced by cultural norms related to national identity and aesthetic preferences, including a nineteenth century aspiration in the United States to create the "pastoral city." According to Campanella (2003, p. 128), this uniquely American urban aesthetic was rooted in antipathy to urbanism and veneration of rural life. This perspective fueled an urban tree planting movement that began in New England with the American elm (*Ulmus americana*) and was exported to nearly every region of the young republic. Echoing this sentiment, Rutkow (2012, p. 8) describes "trees as one of the great drivers of national development ... that helped to forge American identity." These factors may have informed L'Enfant's 1791 plan for Washington, DC, which reserved space in the public right-of-way for trees; an 1870 Parking Act that characterized public right-of-ways as linear parks (Government of the District of Columbia, 2011); and 60,000 street tree plantings in the late nineteenth century (Bednar, 2006). This massive urban tree planting initiative—described as the first of its kind in the United States—pushed the city to the brink of bankruptcy, but resulted in over 450 km of streets lined on both sides with trees (Dümpelmann, 2019; District of Columbia Department of Transportation [DDOT], 2020). Today, the nation's capital is affectionately referred to as the City of Trees (Choukas-Bradley and Alexander, 2008).

Limitations and Future Research

This study is limited to five cites, and for this reason caution should be exercised in generalizing findings beyond the cases covered. An additional limitation of this study is the variation in local data collection methods. This is a fundamental challenge of doing international comparative research, and to reduce the degree to which this variability influenced the findings, protocols were applied as described in the section on "Materials and Methods."

By showing clear differences in the density and distribution of street trees in cities within and between climate zones, this study illustrates the importance of local legacy effects. As noted by Roman et al. (2018), environmental context sets certain bounds on the trajectory of urban forests, but city trees are embedded in an inherited cultural landscape that is heavily influenced by local sociopolitical history. Yet, these legacies are often lacking when seeking to understand the structure and composition of urban forests today. Moreover, cultural factors including but not limited to urban form, aesthetic norms, and governance regimes—all of which can shift across time—may play a pivotal role in the distribution and density of street trees. These factors are, in turn, substantially influenced by national history and identity (Campanella, 2003; Lawrence, 2006), and likely extend to urban flora writ large. Notwithstanding environmental forces such as pests and extreme weather events, which can both substantially reduce urban tree populations (Campanella, 2003; Duryea et al., 2007; Palmer et al., 2014; Conway and Yip, 2016), this study illustrates that cultural dimensions should be foregrounded in scholarship on street tree density and distribution.

The heterogeneous findings of this study illuminate the need for more comparative analysis of urban greening research and practice across national and cultural settings (Keller and Konijnendijk, 2012), where vernacular norms and aesthetic preferences may differ (Hussain, 2017). This has implications for urban greening practice as well as the stewardship and sustainability of urban flora. As noted by Nassauer (1997), landscapes that people admire may be more likely to survive than those that do not attract care or admiration. This, in turn, argues for greater attention to *in situ* research methods based on how people actually experience streets in different places. For example, street-level views account for formal and experiential dimensions of trees that aerial perspectives do not.

Because this study relied upon a small sample of cities and only addressed two climate zones, it would be useful to expand this research to a wider range of climates and cultural contexts. This study also highlights opportunities to advance new geospatial research methods. For example, the original approach to the study involved the use of Google Earth satellite imagery to manually count trees on a 100 m segment of city street closest to the centroids of 100 cells of a grid overlaying the municipal boundary of each study city. But through the course of research, up-to-date geospatial street tree and street network datasets were identified and acquired for each city. This type of data may not be available in many cities, in which case the aforementioned method may be appropriate.

CONCLUSION

This comparative assessment of street tree density and distribution reveals substantial variation across five capital cities spanning two climate zones, and these differences can be attributed to place-specific legacy effects. The environmental legacy of a city was observed to inform differences in street tree density: Ottawa and Stockholm, located in the cooler climate zone, generally exhibited lower street tree density than Buenos Aires, Paris, and Washington, DC, which are located in warmer climate zones. However, the findings also suggest that street tree density and distribution cannot be explained by environmental factors alone. The tree density on local streets in Buenos Aires and Washington, DC was more than double that of Paris, while tree density on local streets in Ottawa was more than four times that of Stockholm. Moreover, the distribution of trees across a three-tiered street classification showed no consistent pattern. These findings reinforce the importance of place-specific legacies as determinants of citywide street tree density and distribution. Substantial differences within climate zones further suggest that cultural factors including but not limited to urban form, aesthetic norms, and governance regimes may play a pivotal role in the distribution and density of street trees, and these dimensions should be foregrounded in urban greening scholarship.

DATA AVAILABILITY STATEMENT

Publicly available datasets were analyzed in this study. This data can be found here: Ottawa tree inventory: Open Ottawa https://open.ottawa.ca/datasets/tree-inventory; Ottawa street network: Open Ottawa https://open.ottawa.ca/datasets/road-centrelines; Ottawa jurisdictional area: Open Ottawa https://open.ottawa.ca/datasets/wards; Ottawa bodies of water: Open Ottawa https://open.ottawa.ca/datasets/water; Stockholm tree inventory: Open Stockholm https://dataportalen.stockholm.se; Stockholm street network: Open Stockholm (Baskarta) https://dataportalen.stockholm.se; Stockholm jurisdictional area & bodies of water: Open Stockholm (Tätortskartan Stockholms län) https://dataportalen.stockholm.se; Buenos Aires Tree Inventory: Buenos Aires Data https://data.buenosaires.gob.ar/dataset/arbolado-publico-lineal; Buenos Aires street network: Buenos Aires Data https://data.buenosaires.gob.ar/dataset/calles; Buenos Aires jurisdictional area and bodies of water: Base de Datos Geoespacial Institucional https://www.ign.gob.ar; Paris tree inventory: Open Data Paris (Les arbres) https://opendata.paris.fr; Paris street network: Open Street Map via data.gouv.fr (Carte des Départements) https://www.data.gouv.fr; Paris jurisdictional area: Open Data Paris (Arrondissements) https://opendata.paris.fr; Paris bodies of water: Institut National de l'Information Géographique et Forestière (BD TOPO®Hydrographie) https://www.data.gouv.fr; Washington, D.C. tree inventory: Open Data DC (Urban Forestry Street Trees) https://opendata.dc.gov; Washington, D.C. street network: Open Data DC (Street Centerlines) https://opendata.dc.gov; Washington, D.C. jurisdictional area: United States Census Bureau (District of Columbia Tiger/Line Shapefile) https://catalog.data.gov; Washington, D.C. bodies of water: United States Census Bureau (District of Columbia Hydrography Shapefile) https://catalog.data.gov.

ETHICS STATEMENT

Written informed consent was obtained from the individual(s) for the publication of any potentially identifiable images or data included in this article.

AUTHOR CONTRIBUTIONS

TSE conceptualized the study. NS developed the methodology, acquired and curated the data, and led data analysis. AK and TSE supervised the analysis and writing process. NS created the original draft. All authors edited and revised subsequent drafts of the manuscript, approved the final version, and agree to be held accountable for the work.

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Dog Urine Has Acute Impacts on Soil Chemistry in Urban Greenspaces

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Urban residents and their pets utilize urban greenspaces daily. As urban dog ownership rates increase globally, urban greenspaces are under mounting pressure even as the benefits and services they provide become more important. The urine of dogs is high in nitrogen (N) and may represent a significant portion of the annual urban N load. We examined the spatial distribution and impact of N deposition from dog urine on soils in three urban greenspace typologies in Finland: Parks, Tree Alleys, and Remnant Forests. We analyzed soil from around trees, lampposts and lawn areas near walking paths, and compared these to soils from lawn areas 8 m away from pathways. Soil nitrate, ammonium, total N concentrations, and electrical conductivity were significantly higher and soil pH significantly lower near path-side trees and poles relative to the 8 m lawn plots. Also, stable isotope analysis indicates that the primary source of path-side N are distinct from those of the 8 m lawn plots, supporting our hypothesis that dogs are a significant source of N in urban greenspaces, but that this deposition occurs in a restricted zone associated with walking paths. Additionally, we found that Remnant Forests were the least impacted of the three typologies analyzed. We recommend that landscape planners acknowledge this impact, and design parks to reduce or isolate this source of N from the wider environment.

Keywords: urban, parks, greenspace, dogs (*Canis familiaris*), nitrogen, soil, urine, city

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INTRODUCTION

As cities grow (Ritchie and Roser, 2019), so do their impacts on and interactions with urban greenspaces. Urbanization has widely recognized impacts on plant, animal, and microbial communities as well as on local and regional nutrient cycles (McKinney, 2008; Churkina, 2016; Decina et al., 2019). Along with increasing urbanization, dog (*Canis familiaris*) ownership is also on the rise. While common in western countries such as the United States – where as many as 49% of households own at least one dog (The Insurance Information Institute, 2019) – dog ownership is rapidly becoming more common globally (GfK, 2016).

Many urban residents visit greenspaces daily, and these areas provide residents with many valuable ecosystem services such as stormwater retention and treatment, the sequestration of excess nutrients and metals, and opportunities for recreation and nature-connections for city residents (Bolund and Hunhammar, 1999; McCormack et al., 2010; Irvine et al., 2013; Bertram and Rehdanz, 2015; Setälä et al., 2017). As dog-walking is a common activity in urban greenspaces

(Brown and Rhodes, 2006; Goliènik and Ward Thompson, 2010; Iojă et al., 2011), dog feces are a recognized problem in these areas (Mallin et al., 2000; Whitlock et al., 2002; McCormack et al., 2010; Cinquepalmi et al., 2013; Rock et al., 2016). However, the impact of dog urine has received little attention. Recent work by Hobbie et al. (2017) showed that nutrient inputs from pet waste (both feces and urine) contributed up to 28% of total N to urban watersheds, second only to residential lawn fertilizer. Paradeis et al. (2013) examined the distribution and concentrations of soil nutrients and salts within enclosed, off-leash dog parks and found that these variables were distributed along gradients and at hotspots within those parks. While N is a vital nutrient for plant growth, its excess application has negative effects on soil functions and the quality of both ground and surface waters. The urine of dogs is rich in urea, which breaks down to available N in the form of ammonium in the soil through the process of hydrolysis. A recent laboratory study by Lee et al. (2019) showed that even short-term applications of dog urine has significant effects on soil biogeochemistry in urban green infrastructures and negatively impacted the ability of these structures to retain and treat stormwater.

Cities are both sources and sinks for N and other nutrients (Lorenz and Kandeler, 2005; Lorenz and Lal, 2009). The need to better understand the N contribution of dogs (hereafter referred to as dog-deposition N) is becoming more pressing as cities and dog ownership rates grow globally. Based on the average dog-ownership rate for Finland (∼22%), we estimate that dogs living in Helsinki, the capital of Finland, may produce as much as 15 kg N ha^{-1} annually. This is comparable to atmospheric deposition, which is estimated to be 2.1–25 kg total dissolved inorganic nitrogen ha^{-1} yr^{-1} (Manninen, 2018). However, dog-deposition N is unlikely to be homogeneous across the urban area, and likely represents an even more significant impact within a particular area of greenspace (see Paradeis et al., 2013).

In this study, we examined soils from different types of urban greenspace, and from different areas within them to better understand the spatial distribution of dog-deposition N. Due to leash requirements in Finland (Järjestyslaki, 2003), we hypothesized that:

1. Dog-deposition is not evenly distributed, but objects located near pathways, e.g., trees and utility poles, receive higher inputs than lawn areas adjacent to the same path. Furthermore, dog-deposition effects will be higher close to these objects than further away, due to the preference of dogs to countermark the urine of other dogs (Lisberg and Snowdon, 2011).
2. The magnitude of dog-deposition along pathways will vary by greenspace type, with Remnant Forests being more heavily impacted than Tree Alleys and Parks the least. We hypothesize that Remnant Forests will show the highest impact due to leash requirements and the presence of understory vegetation and closely spaced trees bounding the paths. This would make excursions away from the pathway more difficult, and so dogs are bound to spend more time on the paths relative to the Tree Alleys and Parks. We expect Tree Alley paths to be more impacted than Park paths due to their linear nature, while the open lawns and widely spaced trees of Parks offer dogs and their owners ample opportunities to deviate from the pathways.

MATERIALS AND METHODS

Study Area

Two cities in Finland were included in this study: (1) Helsinki (60°10′15′′N 24°56′15′′E), population ca. 650 000; and (2) Lahti (60°59′N 025°39′E), population 120 000 (see Setälä et al., 2016 for additional details regarding these localities). Thirty-four sites were selected (Helsinki n = 18, Lahti n = 16), grouped into three typologies: Parks (n = 11): public spaces with maintained pathways, lawns, trees, etc.; Tree alleys (n = 11): linear features consisting of a tree lined path bounded by, e.g., buildings, roads or fencing; and Remnant forests (n = 12): relatively unmanaged areas with a dense tree canopy and networks of maintained and informal pathways. A site location map can be found in the **Supplementary Material** (**Figure S1**), and an interactive version of the field site map can be found at https://bit.ly/3lQcrNq.

Sample Collection

We collected soil samples from 22 August to 13 September 2018. Next to a main pathway at each site, we collected composite samples of eight sub-samples from the top 10 cm of soil using a stainless steel push corer (3 cm ø) at: (1) a deciduous tree (*Acer, Tilia, Ulmus, Betula* or *Quercus* sp.), (2) a utility or lamppost, and (3) a lawn area. Lawn areas were selected to be >5 m away from any objects (e.g., benches, trash bins, lampposts), and outside of the tree canopy where possible. At trees and poles, one composite sample was taken from within 30 cm around the item and a second one from within an area of 1 m^2 centered at 1 m from the edge of the item opposite the pathway. From the lawn, one composite sample was taken from within a 0.5 m^2 area immediately adjacent to the path, and the second from within 1 m^2 centered at 1 m from the path edge. In Parks we also collected soil samples from lawn areas >8 m away from pathways (n = 30), and from around trees inaccessible to dogs or >8 m from a main pathway (n = 6). Schematics for the typical layout of each typology are given in the **Supplementary Material** (**Figure S2**).

Sample Handling, Processing and Analyses

Samples were stored in a freezer at −20°C at the end of each field day to limit the loss of N due to the continued metabolic action of soil bacteria. Prior to analysis, batches of ∼25 samples were removed from the freezer and thawed overnight at +4°C, sieved (2 mm mesh) and homogenized by hand in a 10 L plastic bucket. The sieve and bucket where thoroughly cleaned between each sample using a brush and warm tap water, then dried using paper towels. Disposable nitrile gloves were worn while sieving and homogenizing the soil and were changed between samples. Approximately 0.5 dL of the sieved samples were set aside for soil dry mass determination after drying overnight at 110°C, and organic matter content (%OM) by the standard loss-on-ignition method (Finnish Standard Association, 1990). Soil electrical conductivity (EC) and pH were measured from a 1:2 volume mix of air-dried soil and ultra-pure water, 4 h after mixing.

For the nitrogen analyses, soil samples were extracted following Decina et al. (2018), using a 2M solution of KCl. Laboratory blank samples of filtered 2M KCl were created at least once per day and for each batch of 2M KCl solution. The filtered sample extracts and blanks were stored in 100 mL plastic bottles and frozen at −20°C to await analysis.

Soil extracts were analyzed colometrically for nitrate (NO_3^-), ammonium (NH_4^+), and total nitrogen (TN) at the University of Helsinki's Environmental Laboratory at the Lahti Campus. Briefly, samples were pipetted into 96-well microplates with one standard curve per plate at the beginning with a series of external quality control solutions. Procedures for making and adding reagents to the microplates, as well as their analysis followed Sims et al. (1995) and Doane and Horwáth (2003) for NH_4^+ and NO_3^-, and Miranda et al. (2001) for TN. Limit of quantification (LoQ) values for each analysis were established by analyzing multiple blank samples with added reagents. Manufacturer and batch information for the materials, reagents, standards, and equipment used are given in the **Supplementary Material**.

Soil freezing has been shown to impact the amount of extractable N measured from soils, which may show a marked increase after thawing (see Edwards and Cresser, 1992). While this is a concern, the soils from our study are typically frozen for several months during winter, thus freezing of the soils prior to analyses is unlikely to introduce a bias to our results.

Stable Isotope Analysis

To determine if nitrogen deposited within urban greenspaces originate from similar sources, soil samples from path-side Poles and Trees (0 m) ($n = 6$) and lawn areas 8 m from the path ($n = 4$) were analyzed at the Finnish Museum of Natural History's Laboratory of Chronology in Helsinki to determine their δ15N values. The raw isotope data were normalized with a multi-point calibration using certified isotopic reference materials (USGS-40, USGS-41, IAEA-N1, and IAEA-N2). The mean measured δ15N values for calibration references were −4.32‰ for USGS-40, +46.66‰ for USGS-41, +0.62‰ for IAEA-N1, and +20.13‰ for IAEA-N2. Replicate analyses of quality control reference materials (soil, corn leaf) analyzed alongside the unknowns indicate a 1(σ internal precision of ≤0.10.

Statistical Analyses

All statistical analyses were performed in R (v 3.6.3) (R Core Team, 2020) for each of the response variables: EC, pH, NO_3^-, NH_4^+, and TN. Normality of these variables was determined by inspecting histograms and performing Shapiro–Wilks Normality tests. Appropriate power transformations for non-normal variables were determined using the *transformTukey* function from the rcompanion package (Mangiafico, 2020).

Generalized linear mixed models (GLMM) (Bates et al., 2015) were used to test the effects of dog urine on the soil parameters listed above. First, we examined the spatial extent of dog-deposition at 0, 1, and 8 m from the path-side treatments using data from the Park typology only. These models include (i) *treatment* (a factor with three levels; Lawn, Tree, Pole), (ii) *distance* (a factor with three levels; 0, 1, and 8 m), their two-way interaction, and percentage organic matter (OM) and soil moisture. We included *site*, nested within *city* as a random term in the models. Model selection was performed by removing OM and/or soil moisture when these variables were not statistically significant (p-values > 0.1).

Second, to test if dog-deposition magnitude varies by type of greenspace we again used GLMMs and tested the response variables against (i) *typology* (a factor with three levels; Tree Alley, Remnant Forests, and Parks, (ii) *treatment* (a factor with three levels; Lawn, Tree, Pole), (iii) *distance* (a factor with two levels; 0 and 1 m), the *treatment* × *distance* interaction, and percentage OM and soil moisture. The random term was structured as above, and model selection was performed in the same manner.

RESULTS

Soil chemistry varied greatly depending on proximity to path-side trees and poles (**Figures 1**, **2** and **Tables 1**, **2**). Soil EC, NO_3^-, NH_4^+, and TN levels were several times higher, and pH considerably lower within the 30 cm area around path-side trees and poles compared to soils 1 m away (in all three typologies) and 8 m away (in Parks). However, path-side lawn areas were largely indistinguishable from the lawn areas 1 and 8 m away from the path. We also found slight differences between greenspace typologies (**Figure 2** and **Table 2**).

Stable isotope analysis of a subset of samples showed the soils around path-side trees and poles ($n = 6$) to have a mean (δ15N value of 8.3, while samples taken from 8 m away ($n = 4$) had a mean [δ15N value of 3.5 (Welch two sample t test, $t = 3.556$, $p = 0.008$) (**Supplementary Figure S3**)].

DISCUSSION

We have shown that dog-deposition is localized and impacts soil chemistry in urban greenspaces significantly. Supporting the first hypothesis, soil chemical characteristics and δ15N values around path-side trees and poles were significantly different from those located further from the paths and from lawn area soils next to the same pathway. Differences we observed in the δ15N values of soil samples taken from path-side trees and poles at 0 m and 8 m away suggest that the primary N inputs to these areas are derived from different sources. However, contrary to expectations, we found no difference in the measured variables between the path-side (0 m), 1 and 8 m lawn samples, indicating that path-side trees and poles act as focal points for dog-deposition, while lawn areas do not. This is likely a function of gender-specific differences in dogs' urinating and scent-marking behaviors (countermarking), with male dogs preferring to urinate directly on trees and poles (overmarking) while females generally do not, instead preferring to urinate near, but not at the same locations as other dogs (adjacent-marking) (Pal, 2003; Lisberg and Snowdon, 2011).

Our data do suggest that dog-deposition impacts vary with greenspace typology, but not in the way we expected. Remnant

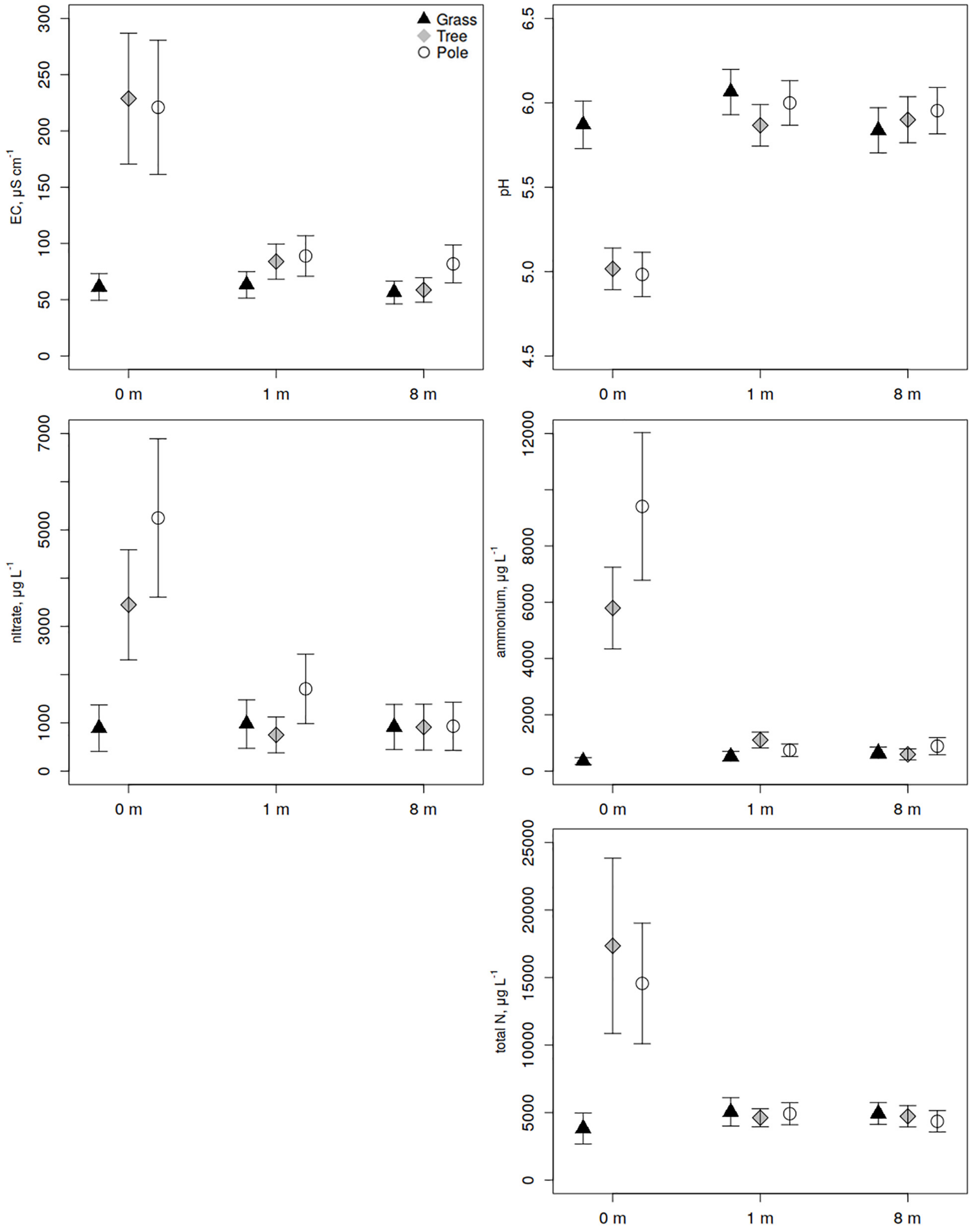

FIGURE 1 | Back-transformed model predicted mean ± SE values for Electrical Conductivity (EC), pH, nitrate, ammonium, and total nitrogen at 0, 1, and 8 m away from path-side grass plots, trees, and poles in Parks only.

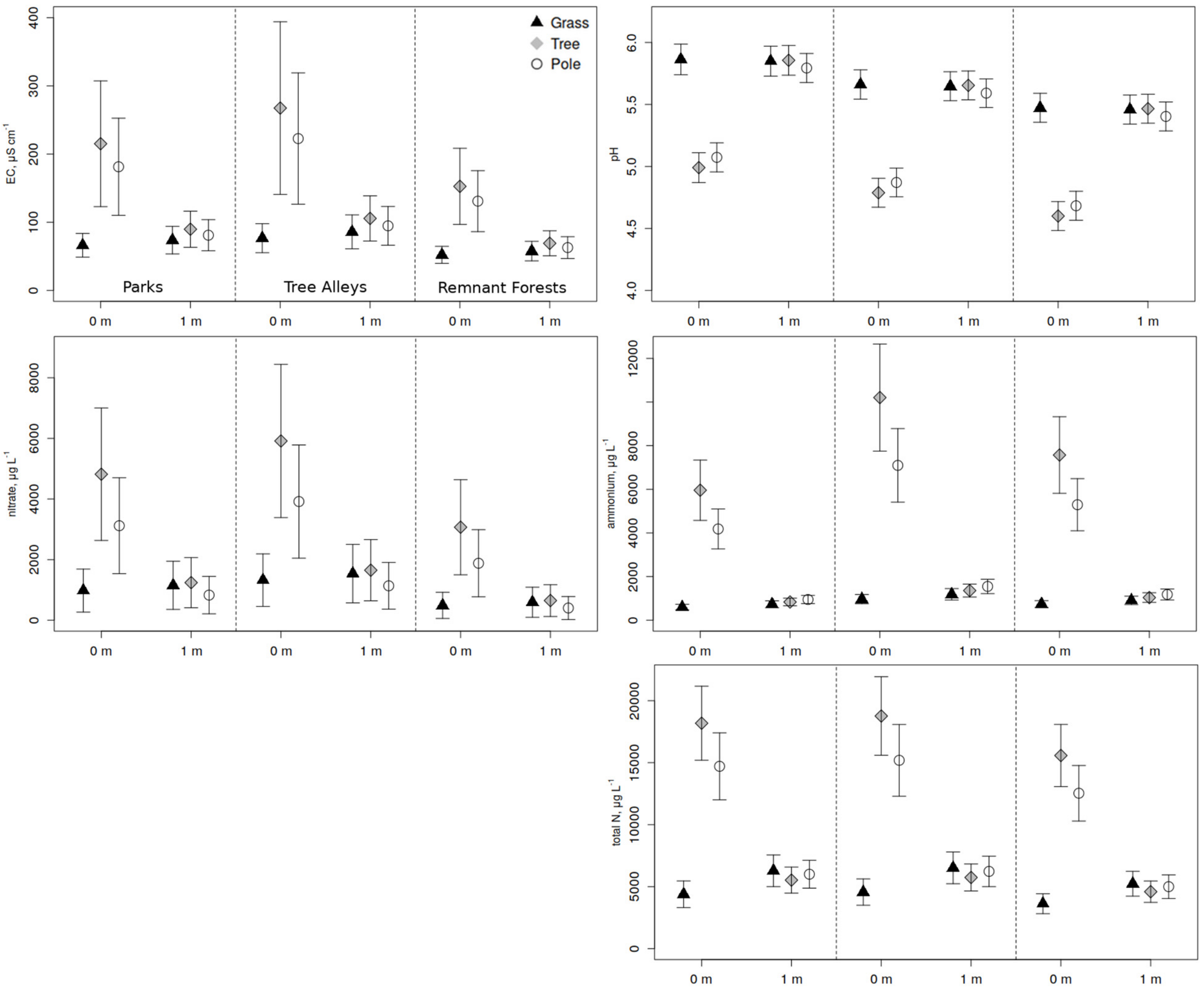

FIGURE 2 | Comparison of back-transformed model predicted mean ± SE values for Electrical Conductivity (EC), pH, nitrate, ammonium, and total nitrogen at 0 and 1 m away from path-side grass plots, trees, and poles in three greenspace typologies: Parks, Tree Alleys and Remnant Forests.

Forests, rather than being the most impacted, were found to be the least affected, while Tree Alleys were found to be the most heavily impacted, followed by Parks. The lower values observed in Remnant Forests could be due to a lower number of dog walkers in these areas, while Tree Alleys may experience higher volumes of traffic and may also be the first area of greenspace that a dog encounters when being taken for a walk. The open design of Parks may allow dogs and their owners more opportunities to deviate from established pathways, thereby spreading their impacts more widely. Another factor that is likely to affect the magnitude of dog-deposition in these areas is residential population density of the surrounding areas. While we did not directly examine this potential correlation, we did select our study sites to be located within the urban core or ≤500 m of multifamily/high-density residential areas.

Our research indicates that dog-deposition is strongly associated with objects near pathways in urban greenspaces and that it is localized. Significant rapid and long-lasting impacts on soil biogeochemistry have been shown to result from even a single application of urine (see, e.g., Haynes and Williams, 1992; Orwin et al., 2009; Lee et al., 2019). The effects on soil chemical properties observed in our study suggest that the impact of dog urine in urban greenspaces is even greater than the impacts observed in these studies. Furthermore, in addition to being highly localized, the input of N from dogs to urban greenspaces is chronic, and it is likely that multiple dogs will urinate in the same location each day. This sustained input of concentrated N in areas frequented by humans for recreation and leisure represents a uniquely urban phenomenon, one whose closest analog may be pastureland urine patches or waste lagoons in the confined animal feeding operations (CAFOs) of industrial agriculture. In fact, the average concentrations of ammonium we measured from soils located around path-side poles in Parks was 103.9 ± 18.4 mg kg^{-1} (mean ± SE), which is more than four times the cleanup standard proposed by Volland et al. (2003) for ammonium (25 mg kg^{-1}) in soil underneath CAFOs and

TABLE 1 | GLMM results, testing the effects of treatment (a factor with three levels; grass, pole, tree), distance (a factor with three levels; 0, 1, and 8 m), and their two-way interaction on five variables (pH, EC, nitrate, ammonium, and tot. nitrogen).

Variable	Intercept	Pole	Tree	Distance 1 m	Distance 8 m	Tree × 1 m	Pole × 1 m	Pole × 8 m	Tree × 8 m	Soil moisture	Soil organic matter
EC	−0.322	**0.096**	**0.098**	0.003	−0.007	**−0.074**	**−0.067**	**−0.064**	**−0.094**		
	(0.017)	**(0.016)**	**(0.015)**	(0.016)	(0.016)	**(0.020)**	**(0.022)**	**(0.022)**	**(0.021)**		
	<0.001	**<0.001**	**<0.001**	0.860	0.642	**<0.001**	**0.002**	**0.004**	**<0.001**		
pH	5.870	**−0.887**	**−0.854**	0.194	−0.033	**0.656**	**0.822**	**1.004**	**0.917**		
	(0.150)	**(0.140)**	**(0.131)**	(0.143)	(0.143)	**(0.180)**	**(0.193)**	**(0.197)**	**(0.191)**		
	<0.001	**<0.001**	**<0.001**	0.174	0.818	**<0.001**	**<0.001**	**<0.001**	**<0.001**		
Nitrate	5.460	**3.052**	**2.202**	0.126	0.041	**−2.557**	**−2.213**	**−3.030**	**−2.210**		
	(0.738)	**(0.671)**	**(0.635)**	(0.698)	(0.684)	**(0.876)**	**(0.933)**	**(0.959)**	**(0.919)**		
	<0.001	**<0.001**	**<0.001**	0.857	0.953	**0.004**	**0.018**	**0.002**	**0.016**		
Ammonium	1.136	**0.099**	**0.084**	0.012	0.018	**−0.062**	**−0.089**	**−0.090**	**−0.087**	**0.134**	
	(0.012)	**(0.012)**	**(0.012)**	(0.013)	(0.013)	**(0.016)**	**(0.017)**	**(0.018)**	**(0.017)**	**(0.045)**	
	<0.001	**<0.001**	**<0.001**	0.368	0.155	**<0.001**	**<0.001**	**<0.001**	**<0.001**	**0.003**	
Tot. N	−0.031	**0.011**	**0.012**	0.003	0.003	**−0.013**	**−0.011**	**−0.012**	**−0.012**		**0.001**
	(0.003)	**(0.003)**	**(0.003)**	(0.003)	(0.003)	**(0.004)**	**(0.004)**	**(0.004)**	**(0.004)**		**(0.000)**
	<0.001	**<0.001**	**<0.001**	0.410	0.421	**0.002**	**0.005**	**0.002**	**0.002**		**0.005**

The Grass and 0 m distance treatment levels are in the intercept. The values presented are the coefficient (with the Standard Error in parentheses), and the p-value. Significant effects ($p < 0.05$) are highlighted in bold. All variables except pH were power transformed using transformTukey function in the rcompanion package.

TABLE 2 | GLMM results, testing the effects of typology (a factor with three levels: Parks, Tree Alleys, and Remnant Forests), treatment (a factor with three levels: grass, pole, tree), distance (a factor with two levels: 0 and 1 m), and their two-way interaction on five variables (pH, EC, nitrate, ammonium, and tot. nitrogen).

Variable	Intercept	Remnant forests	Tree Alleys	Poles	Trees	Distance 1 m	Pole × 1 m	Tree × 1 m	Soil organic matter	Soil moisture
EC	−0.270	**−0.021**	0.012	**0.081**	**0.071**	0.009	**−0.066**	**−0.064**		**0.082**
	(0.022)	**(0.010)**	(0.010)	**(0.008)**	**(0.008)**	(0.008)	**(0.012)**	**(0.011)**	–	**(0.033)**
	<0.001	**0.037**	0.227	**<0.001**	**<0.001**	0.293	**<0.001**	**<0.001**		**0.014**
pH	6.151	**−0.390**	−0.203	**−0.873**	**−0.790**	−0.014	**0.880**	**0.735**	**−0.024**	
	(0.132)	**(0.161)**	(0.159)	**(0.090)**	**(0.089)**	(0.091)	**(0.125)**	**(0.122)**	**(0.004)**	
	<0.001	**0.015**	0.204	**<0.001**	**<0.001**	0.877	**<0.001**	**<0.001**	**<0.001**	
Nitrate	5.162	−0.888	0.438	**2.740**	**1.881**	0.232	**−2.629**	**−2.338**	**5.324**	**−0.041**
	(1.004)	(0.549)	(0.540)	**(0.424)**	**(0.425)**	(0.437)	**(0.600)**	**(0.587)**	**(2.065)**	**(0.019)**
	<0.001	0.106	0.417	**<0.001**	**<0.001**	0.595	**<0.001**	**<0.001**	**0.010**	**0.036**
Ammonium	−0.742	0.008	**0.017**	**0.079**	**0.067**	0.007	**−0.074**	**−0.058**		**0.094**
	(0.010)	(0.008)	**(0.008)**	**(0.008)**	**(0.008)**	(0.009)	**(0.012)**	**(0.012)**		**(0.033)**
	<0.001	0.327	**0.030**	**<0.001**	**<0.001**	0.401	**<0.001**	**<0.001**		**0.004**
Tot. N	3.230	−0.100	0.021	**0.836**	**0.700**	0.194	**−0.725**	**−1.359**	**1.692**	**−0.907**
	(0.153)	(0.119)	(0.126)	**(0.129)**	**(0.135)**	(0.130)	**(0.175)**	**(0.405)**	**(0.485)**	**(0.170)**
	<0.001	0.402	0.871	**<0.001**	**<0.001**	0.136	**<0.001**	**<0.001**	**<0.001**	**<0.001**

The Park typology, Grass and 0 m distance treatment levels are in the intercept. The values presented are the coefficient (with the Standard Error in parentheses), and the p-value. Significant effects ($p < 0.05$) are highlighted in bold. All variables except pH were power transformed using transformTukey function in the rcompanion package.

is comparable to values found in soils underneath CAFO waste lagoons (DeSutter et al., 2005). By contrast, the ammonium concentrations we measured in Park lawn soils 8 m away from pathways was only 6.7 ± 0.9 mg kg^{-1} (mean ± SE), which is comparable to the values from urban soils analyzed by Paradeis et al. (2013).

The localized nature of dog-deposition provides urban planners with the opportunity to alleviate this impact by modifying greenspace designs and incorporating structures designed to attract and isolate dog urine from the broader environment. Dog owners could be encouraged, through educational outreach and on-site signage, to direct their dogs toward structures or areas where drains can capture infiltrating urine and stormwater. Such a system would protect ground and surface waters by diverting this nutrient rich flow to sanitary sewers or other treatment systems prior to release. Furthermore, greenspaces can be designed with the likely locations of hotspots already in mind, and so controls can be included in the site plan, rather than retrofitted.

Compared to natural areas, cities are enriched with N, and while environmental quality regulations have led to a decrease in atmospheric N deposition in recent decades (Eshleman et al., 2013), dog ownership rates are increasing. Even now, some countries are seeing a spike in pet adoptions and fostering in response to the COVID-19 crisis, with many pet shelters in the United States being completely emptied during the summer of 2020 (Oppenheim, 2020; Vincent et al., 2020). This spike notwithstanding, if current growth trends in urbanization and dog ownership continue, the localized impacts that we have found will likely increase in severity and possibly in spatial extent, and dog-deposition could become the single largest source of N in urban watersheds.

As cities sprawl and/or density, urban greenspaces are coming under mounting pressure, even while the services they provide are becoming more important to greater numbers of people and their pets (Haaland and van den Bosch, 2015). Dogs have played an important part in human societies for thousands of years and will undoubtedly continue to be valuable partners. However, as our populations continue to grow, so does the need to better understand the role of dogs in urban N deposition and their broader impacts on sustainable urban development and the environment.

DATA AVAILABILITY STATEMENT

The original contributions presented in the study are included in the article/**Supplementary Material**, further inquiries can be directed to the corresponding author.

AUTHOR CONTRIBUTIONS

JA conceived, designed, and wrote the manuscript and was responsible for performing the experiment and data analyses. HS provided guidance and support in all aspects of the research and participated in the writing process. DK provided guidance and support in all aspects of the research and writing process, especially regarding experimental design and data analyses. All authors contributed to the article and approved the submitted version.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fevo.2020.615979/full#supplementary-material

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The Legacy Effects of Colonial and Apartheid Imprints on Urban Greening in South Africa: Spaces, Species, and Suitability

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Colonialism is a significant legacy across most aspects of urban form, the nature and distribution of public green spaces, and tree species composition in many cities of the Global South. However, the legacy effects of colonialism on urban green infrastructure and the uses thereof have only recently come under scrutiny. Here we collate information from South Africa on urban greening and interpret it through a colonial and apartheid legacy lens in relation to the distribution and types of urban nature found and their resonance with contemporary needs as an African country. The analysis indicates marked inequalities in public green space distribution and quality between neighborhoods designated for different race groups during the colonial and apartheid periods, which continues to be reproduced by the post-colonial (and post-apartheid) state. Additionally, in the older, former colonial neighborhoods non-native tree species dominate in parks and streets, with most of the species having been introduced during the colonial period. Such colonial introductions have left a burdensome legacy of invasive species that costs billions of Rands annually to keep in check. Lastly, the forms of nature and activities provided in public urban green spaces remains reminiscent of the colonial norm, with little recognition of African worldviews, identity and needs. We conclude in emphasizing the necessity for urban authorities and planners to address these anachronistic legacies through adopting a more inclusive and co-design approach with respect to the extent, location and types of urban nature provided, as well as the types of cultural symbols and activities permitted and promoted.

Keywords: apartheid, colonial, green space, legacy, urban nature

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INTRODUCTION

Urban greening and especially the planting of trees is increasingly advocated as an important strategy for promoting urban sustainability, liveability and resilience (Wachsmuth and Angelo, 2018; Du and Zhang, 2020). However, to meet these goals the types of green spaces developed and species of trees that are planted need to be appropriate to the biophysical setting and simultaneously accord with the cultures and meet the needs of local residents because urban trees and green spaces represent symbols or expressions of particular cultures, preferences, histories and planning approaches (Stewart et al., 2004; Hunte et al., 2019). The latter is challenging as urban settings around the world become more multicultural, and especially so in the Global South due to the rapid rates of urban growth (Khan, 2014; Zwiers et al., 2018; Hunte et al., 2019). Moreover, because trees and parks are generally long-lived entities, preferences and decisions made in a particular period may persist for decades or even centuries, constituting a legacy of a former period. However, despite the history of a region, city or specific site within a city being a primary filter of what species mix is found (Aronson et al., 2016), it is rarely included as an interpretative lens in attempts to understand current species composition and dynamics (Roman et al., 2018), especially with respect to anthropogenic legacy effects (du Toit et al., 2016).

Examination and understandings of site legacy effects on current biodiversity patterns and processes are not unusual for rural or natural settings, but are still relatively rare in urban studies (Roman et al., 2018) and even less so in developing country contexts (du Toit et al., 2016; Hosek, 2019). The legacy effects of a site relate to how the current nature of the site is partly or wholly a reflection of how the site was used or disturbed previously, ranging from a few years beforehand to centuries ago. In this sense we follow the widely accepted definition of Monger et al. (2015:13) of legacy effects being "the impacts that previous conditions have on current processes or properties". Despite the growing recognition of legacy effects, only one study in South Africa has explicitly examined landscape history as a driver of species composition in urban areas, that being du Toit et al.'s (2016) work on grassland remnants in Potchefstroom, although the engrossing ecosystem services history of Cape Town by Anderson and O'Farrell (2012) has relevance.

A significant legacy across most aspects of urban form and society throughout many regions of the Global South is that of colonialism (Myers, 2003). Colonial administrations influenced all facets of life including urban planning and architecture (Ignativea and Stewart, 2009), alongside language, education, knowledge systems, social norms and cuisine, to name a few. Many facets of indigenous knowledge, beliefs and practices were suppressed, denigrated or outlawed (Johnson and Murton, 2007). They also influenced the location and layout of residential areas (Myers, 2003; Scholz et al., 2015; Titz and Chiotha, 2019), of formal public green spaces (Säumel et al., 2009; Abendroth et al., 2012; Scholz et al., 2015), and the extent and composition of tree species planted in public and private spaces (Stewart et al., 2004; Peckham, 2015; Hosek, 2019).

Much of the formal public green spaces in modern cities are found in the more low-density, affluent residential neighborhoods (Martin et al., 2004; Wolch et al., 2014), with evidence showing that low income and typically high density urban areas are usually the most susceptible to the uneven distribution of green infrastructure, evidenced by the general lack of trees (Li et al., 2015; Lee et al., 2019) and public green spaces (Thaiutsa et al., 2008; Kabisch and Haase, 2014). Roman et al. (2018) associated this distribution to underlying historical processes and causal mechanisms, among other things, as a result of past decisions because urban green spaces and trees are part of an inherited landscape. The extent and composition of tree species planted in public and private spaces included the introduction of species from the colonizing country for sentimental reasons or from other colonial dominions for aesthetic or economic reasons (Stewart et al., 2004; Abendroth et al., 2012; Roman et al., 2018; Hunte et al., 2019). For example, this pattern is evident in the greater Bandung area, Indonesia, a colonial city established by Dutch settlers at the end of the 19th century, where many shade trees and colorful ornamental species were introduced by the Dutch from species that come from Europe, tropical America, Southeast Asia, and tropical Africa (Abendroth et al., 2012), while Moro and Castro (2015) indicate that the *Azadirachta indica*, an exotic and invasive tree from India (connections that can be traced back to Portuguese coloniszation of Brazil), has become an important ornamental plant in Fortaleza, Brazil in the last decade. Similarly, the *Blighia sapida*, an African species abundant in Georgetown, Guyana, was brought to Jamaica in slave ships in the 18th century during the British colonial rule, and has since become an important livelihood source and integral to Jamaican food culture (Hunte et al., 2019). According to Abendroth et al. (2012), this directly impacts on, and induces a loss of local identity among the colonized communities because traditional natural elements are transformed or replaced by western garden culture.

South Africa offers an interesting case for the examination of the legacy effects of colonialism. As with many previously colonized countries, South Africa has a complex history of occupation and development, which continued for several decades even after independence. It was first colonized by the Dutch in 1652 who established a provisioning station for their ships traveling between Holland and their southeast Asian colonies. After approximately 150 years their hold over the still spatially small colony was broken by the British, who took control in 1806. Over the next century, as the now British colony expanded, there were multiple conflicts between the imperialist British and descendants of the original Dutch colonists (many of who had migrated to the north) and multiple indigenous groups, culminating in the Union of South Africa in 1902 under total British control. This lasted until 1961 when the country became a republic and formally independent of Britain. During this colonial period, the British style influenced urban design and planning (Miraftab, 2012; Scholz et al., 2015), including the establishment and design of several notable urban parks and botanical gardens (such as Kirstenbosch in Cape Town and the botanical gardens in Durban), as well as the introduction of over eight thousand plant species from other parts of the world (van Wilgen et al., 2001; Faulkner et al., 2015). For example, *Lantana camara* was introduced in approximately 1858 as an ornamental, whilst *Acacia mearnsii* was brought in from Australia in 1871 for fuelwood and shelter belts (it subsequently became the foundation species for a large tannin industry). Both have subsequently invaded millions of hectares and are regarded as priority species for control (Robertson et al., 2003).

Almost two decades before becoming a republic, and for three decades afterward, South Africa entrenched many existing and new sweeping racially-discriminatory laws that suppressed the livelihoods, wellbeing, cultures and aspirations of all who were then termed 'non-white' people (we do not subscribe to racial categories but use them to reflect the historical reality, which remains entrenched in urban spatial geographies in South Africa to the present day). Every facet of life of black South Africans was restricted and dictated by a whites-only government and brutally enforced under the central policy of 'apartheid.' These laws restricted which black South Africans could work in urban areas, and where they were allowed to live (Horn, 2019). The urban living areas prescribed for black South Africans became known as townships, and were characterized by systemic underdevelopment with respect to housing, electricity, sanitation, social services (such as education and health), recreational spaces and economic opportunities. They became sites of deep poverty and depravation, which still remains evident today although perhaps not to the same degree (Carruthers, 2008; Horn, 2019). The same applied to most black households who were not allowed to live in urban areas, with millions forcibly relocated to ethnically defined bantustans. With the advent of democracy after the fall of apartheid in 1994, the newly elected government instituted a massive program to address the dire situation in the townships (and bantustans) and the lack of urban housing generally (Miraftab, 2012). Millions of housing units were built under what is known as the Reconstruction and Development Program (RDP) (Horn, 2019), although a monumental backlog remains due to the continued high rates of rural to urban migration. Neighborhoods dominated by RDP houses are now termed RDP areas and are characterized by rows upon rows of tiny houses of identical design, with occupancy prioritized for the indigent. Despite providing shelter for millions, they have been criticized for the small size of the dwelling units, the general lack of social infrastructure such as community halls, sports facilities and public green spaces, and overall not meeting internationally accepted criteria for 'adequate' housing (Goebel, 2007; Moolla et al., 2011; Rapelang et al., 2018) and environmental justice (Ernstson, 2013; Venter et al., 2020).

The legacy effects of colonialism on the composition and distribution of urban forests have only recently come under scrutiny (e.g., Pawson, 2008; Ignativea and Stewart, 2009; Säumel et al., 2009; Hosek, 2019; Hunte et al., 2019), and a great deal more is required to better understand the patterns, processes and implications across, local, national and international scales (Pawson, 2008; Anderson et al., 2020). For example, Hunte et al. (2019) show how distance from the colonial center of Georgetown (Guyana) influenced the type of trees found in particular parts of the city, whilst Gwedla and Shackleton (2017) show how the location of a town in the former racially prescribed bantustans in South Africa influenced the current abundance of street trees. Hunte et al. (2019) argue that generally, literature on urban tree species composition in much of the Global South is largely descriptive with comparatively little understanding of patterns and drivers. Within the context of the above the objective of this paper is to collate and re-interpret existing information and data in South Africa on urban greening using an explicit lens of colonial and apartheid legacy effects, which has not been done before. We consider the distribution and types of urban nature found, and its resonance with contemporary needs as an African country. We do so for three components, namely (1) the public urban green spaces, (2) the woody plant species in public spaces, and (3) the suitability of the spaces and species for the diversity of South African cultures. We focused on these three aspects because most of the studies and knowledge in South Africa cover one or more of these three aspects. Whilst the effects of colonialism on urban form and greening could be examined through other domains and measures, there are insufficient studies to allow detailed examination.

SPACES

The current distribution patterns of private and public urban green spaces and trees in South Africa is extensively reminiscent of colonial planning (Shackleton et al., 2018), and continues to be so even with the development of new low-cost housing residential areas under the RDP (McConnachie and Shackleton, 2010). Drawing on results from several studies in South Africa (McConnachie et al., 2008; McConnachie and Shackleton, 2010; Shackleton and Blair, 2013; Gwedla and Shackleton, 2015, 2017; Shackleton et al., 2018; Radebe, 2019; Anderson et al., 2020; Makakavhule and Landman, 2020), the extent of urban green space and tree distribution reveals inequalities that disproportionately favor the affluent and previously 'white' residential areas, with the poor and predominantly 'black' areas having markedly less public green space and trees.

Public Green Spaces (PUGS)

A few studies in South Africa have revealed that urban green spaces constitute a relatively small portion of the total land cover in urban areas. For example, McConnachie et al. (2008) calculated across 10 towns that PUGS coverage was 10.6% of the area, while Radebe (2019) reported that 21.5% of total land cover of the urban core (excluding wasteland) across eight towns was PUGS. Churchyards and cemeteries also constitute a sizeable portion of the public green spaces of urban areas, as reported by De Lacy and Shackleton (2017) that the green space in these institutions accounts for 13.6% of the total public green space area in Grahamstown. A recent, national remote-sensing survey showed that on average white urban households live within 700 m of a public park, whilst black African households are, on average, 1.7 km away from one (Venter et al., 2020).

Radebe (2019) found that the RDP neighborhoods of eight small to medium-sized towns in the Eastern Cape province had only small areas of PUGS compared to both the township and affluent areas; ranging between 0.9% (Queenstown) to 26.0% (Bedford). On the other hand, the affluent areas were found to have relatively larger areas of PUGS, ranging from 34.8% (King William's Town) to 74.1% (Queenstown) (Radebe, 2019). These findings echo those of McConnachie and Shackleton (2010), who found evidence of relatively poor provision of PUGS in terms of both the size and proportion of the spaces in the RDP areas (0.7 ± 0.3; 3.6 ± 1.5) of a different set of nine towns in the province than both the township (1.7 ± 0.1; 12.0 ± 1.9) and affluent (1.8 ± 0.3; 11.8 ± 1.3) areas. In the same light, Shackleton and Blair (2013) concluded that, based on residents' perceptions on the use and their estimates of PUGS in two towns, the RDP and township areas had markedly lower proportions of PUGS than the affluent areas. Moreover, the proportional area under PUGS in black areas has declined during democracy over the last 25 years, but not so in the white neighborhoods, leading Venter et al. (2020:11) to label it as "green apartheid."

Street Trees

In addition to the disparities in the distribution of PUGS, further inequalities in the distribution of street trees are also prevalent across the country, both between towns and between neighborhoods. For example, in one of the first studies, Kuruneri-Chitepo and Shackleton (2011) found that in three towns the affluent suburbs boasted approximately 76% of all street trees, compared to 20% in the CBD and <5% of the trees were in the townships and RDP neighborhoods, despite the larger size of township and RDP neighborhoods. In an assessment of the distribution of street trees between towns, the legacy of colonial planning was more pronounced in other parts of the province at both the town (Gwedla and Shackleton, 2015) and suburb scales (Gwedla and Shackleton, 2017). At town scale, Gwedla and Shackleton (2015) reported a low abundance of street trees per 200 m transect (0.6 ± 0.3) among the former bantustan towns, which are generally poor, compared to those that were not part of the bantustans (5.8 ± 1.6) during apartheid and are relatively wealthier. Synonymous to this was the distribution of trees between suburbs, where mean street tree density in the affluent suburbs across all towns was 7.8 ± 0.9 trees per 200 m transect, 1.0 ± 0.3 trees in the townships and 0.2 ± 0.1 in the RDP neighborhoods (Gwedla and Shackleton, 2017). Elsewhere in the country, Schäffler and Swilling (2013) concluded that in Johannesburg approximately 24% of the city's historically wealthy northern suburbs had tree cover, whilst the poorer southern region, dominated by townships, it was approximately 7%.

SPECIES

With respect to the tree species present in public spaces, there have been a few studies in South Africa (mostly in the Eastern Cape province in the southeast) reporting on the composition of street trees in several towns. In some instances we drew from published works and theses, and in a few instances we did analyses or counts from the existing datasets collected during these various studies. All the studies revealed significant inequities in the abundance of street trees between the different neighborhoods, with many streets in RDP and townships having no street trees at all (Kuruneri-Chitepo and Shackleton, 2011; Dotwana-Zona, 2012; Gwedla and Shackleton, 2017). Thus, the former colonial core of the CBD and more affluent areas are generally characterized by higher densities and species diversity of street trees than the townships and RDP areas. Anderson et al. (2020) shows that in Cape Town this pattern translates into a greater diversity of functional traits, conferring greater ecosystem resilience and productivity.

This legacy is further evidenced by the usually higher proportion of non-native species in the CBD and older, more affluent areas than the townships (**Table 1**), and a tendency for a greater proportion of native species in the most recent areas characterized by the RDP neighborhoods (if there are any street trees). This was echoed in parking lots where 83% of trees in parking lots of 10 years or older were non-native, compared to 38% in lots younger than 10 years (O'Donoghue and Shackleton, 2013).

The colonial legacy of species introduction goes beyond just what tree species are found in urban streets and parks to include the effects of those species introductions in terms of the ecosystem services and disservices that they provide. The disservices aspects in relation to some introduced species becoming invasive has received significant attention in South Africa as a mega-diversity country (van Wilgen et al., 2020), albeit only recently in urban settings

TABLE 1 | The proportion (%) of non-native street trees species in 14 towns and neighborhoods in the Eastern Cape, South Africa (nd, no data) (towns were selected by the original studies, and all studies reporting street tree composition in the province have been included).

Site	Town as a whole	CBD	Affluent	Township	RDP	Source
Burgersdorp	89	nd	91	100	No trees	Gwedla (2016) unpubl data
Cradock	81	nd	75	83	No trees	Gwedla (2016) unpubl data
Graaff-Reinet	81	nd	81	77	No trees	Gwedla (2016) unpubl data
Grahamstown	59	69	59	100	Nd	Kuruneri-Chitepo and Shackleton (2011) unpubl data
Libode	100	nd	100	nd	No trees	Gwedla (2016) unpubl data
Matatiele	81	nd	63	25	0	Gwedla (2016) unpubl data
Peddie	50	nd	100	nd	0	Gwedla (2016) unpubl data
Port Alfred	58	69	61	45	Nd	Kuruneri-Chitepo and Shackleton (2011) unpubl data
Port St Johns	100	nd	80	nd	No trees	Gwedla (2016) unpubl data
Queenstown	90	nd	84	50	No trees	Gwedla (2016) unpubl data
Somerset East	74	80	72	57	Nd	Kuruneri-Chitepo and Shackleton (2011) unpubl data
Tsolo	100	nd	100	nd	No trees	Gwedla (2016) unpubl data
Umtatha	55	nd	57	50	44	Dotwana-Zona (2012) unpubl data
Willowmore	75	nd	100	0	No trees	Gwedla (2016) unpubl data
Mean	78 ± 18	73 ± 6	80 ± 16	65 ± 26	15 ± 25	

(Shackleton and Shackleton, 2016; Potgieter et al., 2017; Mclean et al., 2018; Mabusela, 2019; Potgieter et al., 2019; van Wilgen et al., 2020). For example, Mclean et al. (2018) recorded 298 non-native species in a street drive-by survey of the small town of Riebeek-Kasteel (population of 1,150 people) of which 105 were declared invasive species, most of which were introduced during the colonial period. Domestic gardens harbored the greatest number of species. Similarly, Lubbe et al. (2010) reported a higher number of non-native species in domestic gardens than other landuse types in Tlokwe. At a town scale, McConnachie et al. (2008) revealed that the PUGS in former apartheid bantustan towns had significantly greater proportions of invasive alien woody species and numbers of individuals than non-bantustan towns. Of the top 20 invasive alien species in South Africa ranked by Robertson et al. (2003), the approximate dates of introduction are known for 14. All were within the colonial period (one in the 18th century, eight in the 19th century and five in the first half of the 20th century), representing a significant burden from the colonial period, which costs the fiscus billions of Rands annually in control efforts and billions in lost ecosystem services (such as water yield, agricultural production and biodiversity) and major ecosystem disservices such as increased wild fires and stream sedimentation (van Wilgen et al., 2012; van Wilgen and Wannenburgh, 2016).

SUITABILITY

The 'suitability' of a particular green space is in the eye of the beholder or aspirant user. Additionally, attitudes toward and needs for green spaces and urban nature are complex. This is because they are a function of at least three, if not more, overlapping domains including (1) availability and accessibility, (2) the quality, condition and amenities within a particular green space, and (3) the expectations or needs for a particular type of urban nature experience as shaped by user attributes such as age, gender, education, culture and belief systems. The first has been covered in Section "Spaces" of this paper.

The second considers the amenities and maintenance of PUGS and whether they meet local residents' needs. There are several studies from South Africa that mention or report residents' perceptions of local PUGS (Shackleton and Blair, 2013; Kaoma and Shackleton, 2014; Shackleton et al., 2018; Adegun, 2019; Gwedla and Shackleton, 2019; Manyani, 2019). Most indicate that, on the whole, residents from the poorer towns and neighborhoods, already disadvantaged by relatively poor provision of PUGS and trees, feel that the PUGS are poorly maintained by the responsible urban agency (Shackleton et al., 2018; Gwedla and Shackleton, 2019; Manyani, 2019). There are frequent criticisms pertaining to the absence of litter bins (or if present, not emptied on a regular basis) resulting in accumulations of unsightly and perhaps dangerous litter, rare or irregular maintenance with respect to the mowing of lawns or tendering of flower beds (if any) and upkeep to children's play equipment, vandalism of infrastructure not attended to, and the depredations and dung of livestock. There are even reports of deaths resulting from poorly maintained and hence unsafe play equipment (Maclennan, 2019). These all create an aura of neglect across many urban parks in the township and RDP neighborhoods, which deter some potential users (Manyani, 2019; Makakavhule and Landman, 2020). For example, Walton (2012) reported a significant, positive relationship between the number of visitors over a 2-week period and a participatory-derived 'condition' score for eight PUGS in King Williams Town (now named Qonce). Similarly, Shackleton and Njwaxu (2021) monitored 11 public parks in six towns over 3 years and found that as the condition of a park declined, so did spot counts of the number of users. The qualitative assessment of de Vries and Kotze (2016) found that most of the ten parks they assessed in downtown Johannesburg were in an unsatisfactory condition and that the maintenance cycle for most of them was too infrequent to keep them in a state that would attract users. Makakavhule and Landman (2020) echo similar sentiments for parts of Tshwane, the capital city, whilst also noting some parks that are well used and maintained.

Whilst the same narratives are voiced in the more affluent areas, they are fewer, indicating greater levels of satisfaction with PUGS maintenance in those areas (Shackleton and Blair, 2013). Indeed, the little work on condition rating of PUGS shows that the park condition is generally better in the more affluent towns and neighborhoods (Walton, 2012; Seboko, 2019). For example, using data from Walton (2012) the mean participatory condition score for 11 PUGS in the affluent neighborhoods of King Williams Town (200 ± 70) was 63% greater ($t = 3.08$; $p < 0.01$) than 11 parks in the townships (123 ± 42) of the same town. It might also be a result of them requiring less maintenance as there are fewer people in the affluent areas and hence fewer potential users. Additionally, in most affluent neighborhoods households have access to private green space to provide some of their nature needs (Shackleton et al., 2018). Nevertheless, management agencies could accommodate different levels of demand and use and allocate resources accordingly so to eliminate the stark differential in the conditions of PUGS between the richer and poorer communities.

In turning to the expectations or needs of local users, much has been overlooked by adopting a colonial and subsequently a Eurocentric lens. Roman et al. (2018) discuss how biophysical and human drivers leave legacies with respect to the design and tree species composition of PUGS in the United States and Canada. However, they did not examine whose legacies 'counted' nor how the prevailing legacies facilitated or excluded the use of parks by particular social groups, especially native peoples. In South Africa, there is generally a strong appreciation of and affinity to nature, including trees, in urban spaces. For example, Gwedla and Shackleton (2019) reported that 74% of urban respondents agreed that trees were important for quality of life, and that the considerable majority have one or more trees in their home yards (Kaoma and Shackleton, 2014; Gwedla, 2020), being appreciated for a diversity of provisioning, regulating and cultural services (Shackleton et al., 2015). However, with respect to private space, most of the township and RDP neighborhoods lack sufficient space for residents to have a satisfactory quantity or diversity of trees (Gwedla, 2020), and perhaps other flora that are important to them for provisioning or cultural benefits (Haynes et al., 2018). For example, Gwedla (2020) reported that a lack of space was the most commonly reported barrier to homestead tree planting (58% of respondents) across eight towns. It was also the second-most cited reason against tree planting in public spaces, with the high housing density in poorer neighborhoods resulting in very few PUGS and that many of the streets were too narrow to accommodate street trees (Gwedla, 2020). Municipal officials also thought that a lack of space was a significant constraint against tree planting in public areas of RDP and township neighborhoods (Gwedla, 2020). Similar sentiments were reported by Haynes et al. (2018) regarding urban respondents in RDP and township neighborhoods lamenting that they did not have sufficient space in their small yards to grow specific plants of cultural significance that were deemed crucial to their cultural identity and place-making.

In the public arena, the Eurocentric legacy has shaped not only the distribution and abundance of formal green spaces and species, but also the form that they take. Most follow some variation of the English-style park, with large trees and extensive lawns, interspersed with flowerbeds and perhaps some artworks or memorial plaques (such as to those who died in one of the World Wars, or during the liberation struggle) (Cocks et al., 2020; Makakavhule and Landman, 2020). This applies to formal public parks established during the colonial and apartheid periods, as well as in the post-apartheid period (Cocks et al., 2020). This form is designed to provide recreational services associated with exercise and relatively passive recreation with family or friends, and some parks have play equipment for children. Whilst these are important services appreciated by many urban South Africans, they do not adequately serve the "deeper and more multifaceted relationships" (Cocks et al., 2016, p. 821) and meanings that many black South Africans seek from nature. Indeed, it is not "from" nature, which implies a unidirectional relationship, but rather that some indigenous cultures view themselves indivisible from nature, i.e., nature and humanity are an integral part of one and the same entity (Cocks et al., 2016). Accordingly, some cultures believe that ancestral spirits are present in nature and have an influence over daily happenings, fortunes and misfortunes and psychological wellbeing (Cocks et al., 2016). Thus, access to particular forms of nature is vital in respecting and nurturing the bonds with one's ancestors, as evidenced through Cocks et al. (2016) reporting that 84% of urban Xhosa respondents in a survey in Grahamstown said that it was necessary to have access to nature to communicate with their ancestors, and 92% affirmed that their household performs cultural rituals to communicate with their ancestors. Consequently, English-style parks and the limited suite of cultural ecosystem services they provide and the activities they allow are not suitable for some urban black South Africans, and do not meet the diversity of needs and cultures, and hence do not promote a sense of "ownership and attachment to place" (Makakavhule and Landman, 2020, p. 5). A further example is that collection of plants required for cultural or medicinal rituals is generally not permitted in urban parks in South Africa, nor is the holding of religious or cultural ceremonies or observances, although the latter is tolerated by some municipal authorities. The grazing of livestock, which are an integral part of Xhosa culture and rituals, is also generally prohibited (Davenport and Gambiza, 2009).

One consequence of the unavailability or unsuitability of many formal PUGS, is that some urban citizens seek specific nature experiences in informal green spaces (Adegun, 2019; Manyani, 2019) or they delay them until they visit relatives or 'home' in rural settings (Njwambe et al., 2019). Informal green spaces provide the opportunities for more deeply held cultural or religious activities and rituals away from constant scrutiny and urban distractions. For example, informal urban green spaces in southern Africa are often used as places to 'escape' and to communicate with one's ancestral spirits (Cocks et al., 2016). Similarly, such informal spaces may be used as places for outdoor religious practices and worship (Ngulani and Shackleton, 2019), with some urban residents regularly traveling considerable distances to gather and pray at specific sites (Ngulani and Shackleton, 2019). Informal green spaces may also be sites for collection of particular plant species used in traditional or spiritual rituals (Davenport et al., 2011; Cocks et al., 2012). Amongst some ethnic groups in South Africa informal green spaces in or on the periphery of urban settlements are the main sites for coming-of-age male initiation ceremonies (Kepe et al., 2015). The informality of these green spaces also fosters some grazing of livestock, although most urban authorities attempt to exclude livestock from the urban core, and seek to restrict them to the urban periphery, not always successfully (Shackleton et al., 2017). However, use of informal green spaces is not an option available to all because of fears of crime or the poor state of many informal green spaces (Adegun, 2019; Manyani, 2019).

A second consequence is that the PUGS do not contribute to a sense of place and belonging (Njwambe et al., 2019), and consequently many suffer neglect and vandalism (Shackleton and Njwaxu, 2021). This has led to the ironic situation of PUGS, supposedly for public use, being gated and locked, including those that offer memorials to liberation struggle heroes (Cocks et al., 2020; Makakavhule and Landman, 2020; Shackleton and Njwaxu, 2021). There has been only limited consideration of sense of place in urban settings in South Africa, but the inklings we have suggest that it is potentially significant for formal and informal green spaces that meet a diversity of local needs and belief systems (Cocks et al., 2016; Njwambe et al., 2019).

DISCUSSION

This paper has collated evidence that the colonial and apartheid legacies have left a marked and continuing imprint on the distribution, tree species composition and suitability of urban green spaces in South Africa that remain very visible and very real in the current day. Whilst the intensity and nature of colonial domination (and subsequently apartheid in South Africa) had particular nuances in different continents and countries, many of the legacy effects are similar across different socio-cultural settings (Ignativea and Stewart, 2009). Thus, the historical patterns of development and the social environment of cities play a significant role in the availability, amount, distribution and 'accepted' uses of urban green spaces and trees (Ignativea and Stewart, 2009; Kendal et al., 2012; Gwedla and Shackleton, 2017; Fan et al., 2019; Titz and Chiotha, 2019).

Green Spaces

The inequitable distribution of PUGS in and between towns in South Africa as presented in this paper is not unique to South Africa. The inequitable distribution has been reported from a number of countries, typically showing that ethnic/racial minorities (Landry and Chakraborty, 2009; Wolch et al., 2014), and those of lower socio-economic means have comparatively worse provision, or quality, than their counterparts in more affluent areas (Vaughan et al., 2013). For example, low density, more affluent neighborhoods of Delhi, India, boast a green index (amount of urban green space) of 0.44 and an urban neighborhood green index of 0.58, compared to 0.29 and 0.47, respectively, in the high density, less affluent neighborhoods (Gupta et al., 2012). Shanahan et al. (2014) present evidence that socio-economic bias between neighborhoods occurs in both public parkland and residential yards in Brisbane, Australia, stating that the more advantaged neighborhoods had slightly more park area and greater lot sizes than the socio-economically disadvantaged ones.

The same pattern is frequently reproduced with respect to street trees. Globally, municipalities and towns with higher populations and relative affluence tend to have a higher density of street trees (Conway and Urbani, 2007; Landry and Chakraborty, 2009). Shams et al. (2020) found that most of the low-income areas of Karachi, Pakistan, had substantially lower street tree densities than the affluent parts, which they attributed to high land value and better infrastructure of the roads and the sidewalks. The socio-economic determinants of street tree cover are also true for suburb types, and neighborhood socio-economic conditions thus play a significant role. Similar patterns can also be found in Brazil, where dos Santos et al. (2010) reported that less than 10% of neighborhoods in Rio de Janeiro had sufficient street trees, and these were generally older, well established neighborhoods hosting households with greater mean income, compared to the poorer, newer, and more peripheral neighborhoods which had low arboreal index values, and rarely had street trees. Such findings were echoed by Szantoi et al. (2012) in Miami-Dade County, Florida.

Despite the prevalence of such disparities they cannot be interpreted solely as a legacy effect of colonialism, because they are also evident to some degree in former colonizing countries of Europe. For example, Wüstemann et al. (2017) revealed that more high income neighborhoods in German cities had significantly more urban green space within a 500 m distance than households in poorer neighborhoods. Similarly, in the United Kingdom, Pauleit et al. (2005) reported that the cover of trees and shrubs in residential areas in Merseyside increased with increasing affluence, concluding that tree cover appears to be a good indicator of the socio-economic status. However, the crux is that in many former colonized countries there is commonly a strong overlap between wealth and race (Gradin, 2014), including in South Africa (Gradin, 2014; Cheteni et al., 2019), such that a greater proportion of households of indigenous communities are poorer than households that are descendants of colonizing groups. This inevitably raises environmental equity concerns (Tooke et al., 2010; Wolch et al., 2014) where some residents have access to the benefits of urban green spaces and trees, and the ecosystem services they provide, while others do not or to a far lower degree (Li et al., 2015; Nyelele and Kroll, 2020). Thus, the legacy effects are felt not only in the lower provision of urban green infrastructure, but also in lower economic opportunities.

Species

South African towns exhibit significant variability in the composition and diversity of urban trees in public spaces. This is a consequence of the interplay of biophysical (Kirkpatrick et al., 2007) and socio-economic factors (Kendal et al., 2012) and, as we argue here, historical factors too. Non-native trees account for the majority of trees in PUGS and streets across various towns and suburb types in the Eastern Cape province of South Africa. According to Moran et al. (2013), this is a widespread pattern in the southern Africa region as a whole, and can largely be attributed to the region's colonial history. In essence, tree species in cities located in areas colonized by Europeans have traditionally been chosen from a European species pool (Ignativea and Stewart, 2009; Nitoslawski et al., 2016), as well as transfers between colonies (Kemp et al., 2020). A substantial proportion of the non-native urban flora in South Africa was introduced during the colonial period to provide, augment or restore specific ecosystem services (Bennett and Van Sittert, 2019; Potgieter et al., 2019; Shackleton et al., 2020). Ordóñez and Duinker (2013) also attributed the abundance of non-native trees across cities in countries like Canada to their respective colonial histories, with similar interpretations in Christchurch (New Zealand) where more than 80% of street and parkland tree species are exotics (Stewart et al., 2009). The high and even majority proportions of non-native species, introduced mostly during the colonial period, is common across many former colonized countries, such as Niger (Moussa et al., 2020), Brazil (Moro and Castro, 2015), India (Nagendra and Gopal, 2011) and the Caribbean islands (Kemp et al., 2020). It can even be more than just adding to the local species diversity, as specific non-native trees can be the dominant species in many cities, such as in Christchurch, New Zealand (Stewart et al., 2009). The two most dominant species (*Azadirachta indica* and *Mangifera indica*) in two Nigerian cities were both introduced from Asia (Dangulla et al., 2020), while the most dominant species (*Terminalia catappa*) in Rio de Janeiro, Brazil, originates from the tropical regions of Asia, Africa, and Australia (dos Santos et al., 2010). There is some evidence that in public spaces at least, the proportion of native species is increasing in newer neighborhoods as some countries adopt more pro-native species and conservation policies (Stewart et al., 2004; O'Donoghue and Shackleton, 2013; Sjöman et al., 2016; Hernández and Villaseñor, 2018). Non-native species are also common, although rarely dominant, in public spaces of former colonizing countries of Europe, but this was not an external imposition and is driven by a desire to augment the low number of native species available (Sjöman et al., 2016), in contrast to many colonized countries which have far greater native species diversity to select from.

Of particular concern with respect to non-native urban tree species is the potential of some to become invasive and thereby pose threats to native biodiversity and ecosystem services in towns and cities, as well as surrounding landscapes (Säumel et al., 2009; Sjöman et al., 2016; Ward and Amatangelo, 2018). Whilst there is increasing concern over the effects of invasive species, most of the research and control efforts are in rural and protected areas, rather than urban ones, and policies and control efforts are highly variable between countries and regions within countries. For example, *A. indica*, originally from India, is classified as an invasive alien species in Ethiopia (Witt and Luke, 2017), Ghana (Vietmeyer, 1992), and South Africa (Henderson, 2001), but not in Nigeria (despite many similar bioclimatic similarities to the other countries), where Dangulla et al. (2020) found it to be a common and in places, dominant species. Moreover, permitting invasive species in urban settings in a specific region where it might be regarded as safe, as argued by Sjöman et al. (2016), ignores the fact that many species are transferred between regions in countries by individual citizens swapping or sharing planting materials, and also that many current day invasive species were first introduced in urban settings and invade surrounding landscapes by various long-distance dispersal mechanisms (Richardson and Rejmánek, 2011).

Suitability

Considering the suitability of the PUGS in South Africa for urban residents, the results echo those reported from other post-colonial settings w.r.t. the colonial design and legacy of permitted activities and relationships with nature, excluding many citizens from practicing indigenous beliefs and 'relationships' in and with nature. Cocks and Shackleton (2020) describe these as "severed biocultural links," whilst Gobster (2007, p. 100) refers to it as the "museumification of nature." This dislocation between the nature needs of many indigenous peoples in urban settings was instigated during colonial rule (and apartheid later on in South Africa) that systematically suppressed, denigrated and sought to transform indigenous cosmologies, cultures and practices associated with nature (Rozzi, 2012; Mashford-Pringle, 2015). This occurred through the combined onslaught of religious and educational conversion, through which western knowledge systems, religions and worldviews were advanced. This dominance of one specific scientific and cultural representation over others, that alienated those with different views and needs, continues to the current day. For example, Low et al. (2002) describe how long-standing immigrant communities to the United States were alienated from PUGS in Philadelphia via various means, such as cultural symbols with which they had no affinity, codes of dress and behavior, or signage that they cannot understand, or neglect of their history and contributions to the neighborhood and perhaps even development of the park. Interestingly, this feeling was greatest amongst the African-Americans, with similar sentiments reported by Byrne (2012) for Spanish–Americans in Los Angeles. Elands et al. (2019) argue for greater recognition of the need to restore and nurture the diversity of biocultural relationships in PUGS as cities globally become more multicultural. Ignativea and Stewart (2009) describe the ubiquity of the English style parks (and city planning generally) in former colonial cities across the Antipodes, including the introduction of species typical of such parks in the United Kingdom to create the necessary effect, which was only questioned toward the last decades of the twentieth century.

CONCLUSION

This paper has adopted a colonial legacy lens in summarizing and interpreting current information on the distribution, species composition and suitability of PUGS in South Africa. It shows that there are marked and seemingly still indelible colonial legacies indicated by all three of these measures, whereby indigenous South Africans were, and continue to be, disadvantaged, first during the colonial period, then the immediate post-colonial apartheid period, which modern urban planning and delivery consciously or unconsciously continues to reproduce to this very day. Colonial authorities and institutions deliberately undersupplied urban green spaces and street greening to neighborhoods designated for indigenous black South Africans, which was continued by the overtly racist apartheid regime. Simultaneously, the introduction of thousands of species from other continents during the colonial period has left the country with a staggering hangover of biological invasions, which threaten urban and rural biodiversity, ecosystem services and human wellbeing, and which cost billions of Rands annually in control efforts. Yet, the stark inequities in urban greening inherited from the colonial and apartheid periods have not been addressed during the democratic period (since 1994). There has been further alienation due to the largely Eurocentric types of urban nature catered for and promoted, even to this day. Current green space planning and delivery is consciously or inadvertently myopic to the different worldviews held by some black South Africans and consequently, the diverse needs of and experiences in urban nature that need to be satisfied. Consequently, it is imperative that urban authorities and planners address these anachronistic legacies through adopting a more inclusive and co-design approach with respect to the extent, location and types of urban nature provided in South African towns and cities, as well as the types of cultural symbols and activities permitted and promoted in urban nature. With the increased pressure to provide housing to a growing population and thus an expectation for more RDP housing developments to be established, opportunities exists for urban authorities and planners to do this in the quest for sustainable human settlements. This will require concerted effort from municipalities and community leaders, urban authorities, and planners to lobby for the inclusion of urban trees and green space planning to national land use or development plans in line with Afrocentric needs and preferences for urban nature. Furthermore, a focus on indigenous species with contributions to residents' livelihoods could enhance efforts to introduce urban nature in residential areas while increasing the abundance of native species at the same time.

DATA AVAILABILITY STATEMENT

The original contributions generated for this study are included in the article/supplementary material, further inquiries can be directed to the corresponding author.

AUTHOR CONTRIBUTIONS

CS conceived the manuscript. CS and NG wrote it together. Both authors contributed to the article and approved the submitted version.

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Three Histories of Greening and Whiteness in American Cities

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How has urban greening related to the degree of whiteness in neighborhoods? The answer to this question provides an essential "historical diagnostic" that can be used to develop an approach to urban ecology which integrates racial and ethnic change into the planning for proposed interventions. In this paper we employ state sequence analysis to analyze the historical trend of greening (including the implementation of new parks, greenways, community gardens, green recreation areas, and nature preserves) between 1975 and 2014 in a sample of nine cities in the United States relative to concentrations of white and non-white residents. We divide the nine cities into three common growth trajectories and separately examine the trends for each growth trajectory. We further illustrate these trends by mobilizing qualitative data from field work in selected neighborhoods to help explain the processes that generate certain key findings in the quantitative data. We find that the relationship between greening and race/ethnicity differs according to city-level growth trajectory. Cities with continuous high and rapid levels of growth in the postwar period have the strongest link between increased greening and whiter populations. Meanwhile, in cities that contracted or had a punctuated growth pattern, non-white areas had a uniformly low level of greening that occurred mostly in recent years. In all, we show how urban growth, greening, and whiteness are inextricably associated qualities of American cities. We argue that understanding this association is essential for development of a race-conscious model for enhancing urban ecosystems.

Keywords: race and greening, green justice, urban greening, green gentrification, social-ecological conflict, urban ecology and sustainability

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INTRODUCTION

Efforts to enhance urban ecosystems, including greening, nature-based solutions, and environmental sustainability initiatives, have been shown to have substantial shortcomings with regard to lack of recognition of the racist and white supremacist histories upon which they are built (Pulido, 2000, 2017; Safransky, 2014). These critiques reflect a deeper concern for the extent to which social injustices persist in the context of an increasingly green urban planning orthodoxy (Connolly, 2019; Anguelovski et al., 2020). According to this critique, a great deal of the modern environmental movement (Curnow and Helferty, 2018), urban sustainability and resilience planning (Goodling et al., 2015; Walsh, 2018), and green space initiatives (Anguelovski et al., 2018) proceed as if the racial and social context in which they operate is immaterial to the urban ecological initiatives enacted. Some even find greening efforts to be part of a conscious plan for gentrifying areas, causing displacement, and social exclusion along racial/ethnic lines (Gould and Lewis, 2016).

In recognition of this critique, cities have already begun to rethink the approach to greening. Philadelphia's Green Works Equity Index (City of Philadelphia, 2016) and resource allocation strategies around Los Angeles's Measure A (Hingorani, 2019) have recently proposed equity-centered measures or indicators for green planning. Though these examples remain exceptions in the American landscape, they point toward a situation where cities are seeing more and more reasons to avoid uncritical approaches to greening, and these reasons are not confined to a vague concern for social justice. Rather, as some scholars have already argued, to the extent that urban greening initiatives help retrace racist and white supremacist tendencies onto present urban development (Safransky, 2018), social justice goals are undermined, but so too are goals related to generating robust urban ecosystems (Ernstson, 2013; Langemeyer and Connolly, 2020; Schell et al., 2020).

Safransky (2018) points toward the need to understand the historical trend of greening and racialized change in cities as a key step toward addressing the drawbacks of de-politicized green planning. She describes the "historical diagnostic" tools needed for the field of urban ecology to have the capacity to account for racist and ethnocentric effects. These tools, she argues, lead toward modes of enhancing urban ecosystems that are "attuned to the moral claims that historical dispossessions make on the present (Bird Rose, 2004, as cited in Safransky, 2018). In its pursuit of justice, an historical diagnostic is concerned with the task of illuminating hidden histories that point toward alternatives, decolonization, and the challenge of recuperation." Thus, Safransky pushes beyond a theoretical framing of the problem and toward a call for development of empirical tools and information that reshape the goals of urban ecological planning agendas toward a more consciously just model. These tools, we agree, are essential for cities to connect greening and social justice goals.

A racially or ethnically focused historical diagnostic useful for developing a more just set of interventions in urban ecology would provide a thorough understanding of the general patterns of association between urban greening and racial change in cities (for some examples of work in this direction see Boone et al., 2009; Grove et al., 2018; Schell et al., 2020; see also work on legacy effects of urban greening, e.g., Roman et al., 2018). More specifically, it would identify the extent, periodicity, and pattern over time of planned changes in urban ecosystems and changes in the racial makeup across a wide variety of urban settings. With regard to urban greening, the development of such an historical diagnostic remains hindered by a lack of overall understanding of how commonly greening is associated with racial and ethnic change; the type of greening and racial/ethnic change that generates an association; and the conditions under which greening is *not* associated with racial/ethnic change. That is, we lack generalized knowledge of the overall race and ethnicity trends that coincide with urban ecological interventions.

In this paper, we address this lack of generalized knowledge by further developing the historical diagnostic needed for a more race-conscious urban ecological practice to take hold. We employ state sequence analysis to understand the historical pattern of increases in new green spaces and racial change between 1975 and 2014 in nine American cities. The nine cities were chosen to be representative of three predominant growth trajectories (described below). We specifically ask: What are the common sequences across all nine cities with regard for changes in green space and changes in the racial makeup of surrounding areas? Does this pattern vary according to the historical growth trajectory of the city? Can a generalized relationship be determined with regard to the relationship between greening and racial change in these cities? In the sections that follow, we describe the data and methods used to understand the relationship between green spaces and racial change in the nine cities that we study. Next, we present our results of the state sequence analysis and illustrate three key trends in our data through case examples drawn from qualitative field work. Finally, we discuss the implications of the findings for the development of a race-conscious model for enhancing justice-centered urban ecologies and summarize the key conclusions relative to our research questions.

MATERIALS AND METHODS

This paper is a sub-study of a large international European Research Council funded project (GREENLULUS, 2016–2021) focused on the conditions under which urban re-naturing projects in distressed neighborhoods redistribute (or not) access to green/blue spaces for historically marginalized groups in mid-sized cities in the United States, Canada, and Western Europe. The data and methods for the overall project from which this paper draws were chosen with an eye toward developing a more generalized notion of greening and social change over time than is currently available in the literature. For the quantitative analysis of this paper, we selected cities based on a balance between data availability and representativeness with regard for growth trajectories. For our qualitative case examples, we chose a further sub-set of three neighborhoods that represent theoretically and practically important trends shown in the quantitative data and that were included in our qualitative data collection based on in-person fieldwork in 2018, 2019, and 2020.

City Selection

We hypothesize that the relationship between urban greening and racial change differs according to the overall dynamics of urbanization and growth within the city of interest. In order to test this hypothesis, we selected nine cities representative of three common trajectories of growth in the United States during the post-war period (covering the transition to post-industrial American urbanization that began in the 1950s, but was clearly evident by the 1970s). We use data from the United States Census on changes in population, race, and city-level Gross Domestic Product (GDP) in order to differentiate cities according to general trajectories of growth (Manyika et al., 2012). We acknowledge that these are simplified indicators of deeper cultural, social, and political dynamics that have played out in the context of a complex shift from "Fordist" production models rooted in the political economy of nation states to "post-Fordist" models rooted in globalized systems (Marcuse, 1997; Walks, 2001) and embodied in the skilled creative, high technology, biotechnology, education, or finance sectors (Glaeser and Saiz, 2003; Peck, 2005).

Primarily, we relied on population changes as the key indicator of a city's trajectory (see also Park and LaFrombois, 2019), with race and GDP data as secondary indicators. The GDP data trends largely follow changes in population, with some deviations described in the text below, but for simplicity's sake we visualize only the population data. The nine cities selected are shown in **Figure 1** with their population trends between 1950 and 2018 (the most recent available data). The trend lines are color coded to denote cities with similar growth trends. Note that New York City is shown with the data adjusted down to 25% of the population in order to be on a similar scale as the other cities.

Since the second world war, these nine cities took one of three growth trajectories. They either experienced a more or less steady process of expansion; a more or less continuous process of contraction, or a punctuated process wherein population and GDP declined heavily but then saw a resurgence to new highs. As a shorthand, we call these three trajectories "steady growth" cities (Austin, Portland, Seattle); "continuous contraction" cities (Cleveland, Detroit, Baltimore); and "punctuated growth" cities (New York, San Francisco, Philadelphia). With distinct population growth, economic growth, and racial change trajectories, we expect that the different levels of resources and cultural conditions in these three categories of cities led to different relationships with regard to race/ethnicity and greening. The generalized trend line of population growth for these categories of cities is shown in **Figure 2** and the three trajectories are explained in greater detail below.

Steady growth cities generally were not the most prominent urban centers during the Industrial Era in the United States, but became prototypical attractor cities during the latter part of the twentieth century. Often, these cities were home to emergent technology and retail firms (e.g., Microsoft and Starbucks in Seattle; Dell and Whole Foods in Austin; Intel and Nike in Portland) and gained numerous associated economic spin-off benefits that generated rapid and relatively high levels of growth in population and GDP by the early 2000s. Overall, these cities have been mostly white throughout their histories, with a long history of housing discrimination and urban renewal programs. They have seen increases in non-white population since 1950, but remain majority white and are among the whitest of the large American cities. GDP since 1950, and especially since 1990, has risen at a relatively fast rate in steady growth cities compared to other American cities. The cities representative of this category that were analyzed here include Seattle (Washington), Portland (Oregon), and Austin (Texas).

Continuous contraction cities were prominent Industrial Era cities that saw a steady decline in population and GDP since 1950, without any period of substantial resurgence. Generally, these cities are characterized, like most "rustbelt" cities, by disinvestment from industrial urban centers and white flight to the suburbs, except in a few neighborhoods. In Detroit, for example, the 2008 bankruptcy of General Motors collapsed the regional economy and contributed to the subsequent further estimated loss of 40,000 residents between 2009 and 2019 (US Census Bureau, 2019). These cities have seen the lowest rates of increase in GDP since 1950 relative to other large American cities. These cities are also characterized by the highest rates of increase in non-white population since 1950. The cities in this study representing those that experienced continuous contraction include Detroit (Michigan), Baltimore (Maryland), and Cleveland (Ohio).

Punctuated growth cities gained prominence during the early Industrial Era in the United States, and then had declining rates of population and economic growth in the decades following 1950, but saw a resurgence between 1980 and 2010. Generally, by the 1980s, large sections of these cities were substantially disinvested after a process commonly known as white flight to the suburbs. However, between the 1980s and early 2000s, rapid and high reinvestment and economic conversion, with a new prominence gained by the high technology, financial, and/or pharmaceutical industries, resulted in GDP increase and population comeback, including increased white population after decades of decreases in this area. For example, companies such as Merck or Johnson and Johnson contributed to Philadelphia's resurgence while San Francisco's new economy was driven by Silicon Valley tech giants Google and Apple as well as newer firms, like Airbnb and Uber. Meanwhile, punctuated growth cities saw continuous increases in non-white population and currently have a moderate non-white percentage relative to other US cities. GDP also increased at a moderate rate relative to other American cities between 2000 and 2018. The cities considered to be representative of the punctuated growth category that were analyzed in this study include San Francisco (California), New York (New York), and Philadelphia (Pennsylvania). While San Francisco and New York had a clear resurgence by the 1990s, Philadelphia was slower in terms of population and GDP growth, but its trajectory by the mid-2000s put it in this category.

Tract-Level Population and Green Space Data

The population data used to develop the census tract-level analysis in this study is derived from the Longitudinal Tract Database (Logan et al., 2014). This database provides a standardized estimate of numerous demographic variables from the United States Census between 1970 and 2010. It was developed to be the best possible data for analyzing change across time in American urban neighborhoods. We utilize the race and population estimates standardized to 2010 census tract boundaries, which cover the full period of time used in this study.

The green space data for the nine cities analyzed is derived from a specialized dataset we assembled and created that contains the boundaries of all public green spaces in each city as of 2016, along with the type of space (classified in a standardized way by park, greenway, preserve, garden, recreation), the size of the space, and the year the space was acquired for public use as a greenspace (if this year was not available, then the year the space was opened to the public was used instead). Greenspaces in the dataset include all formal and informal public spaces

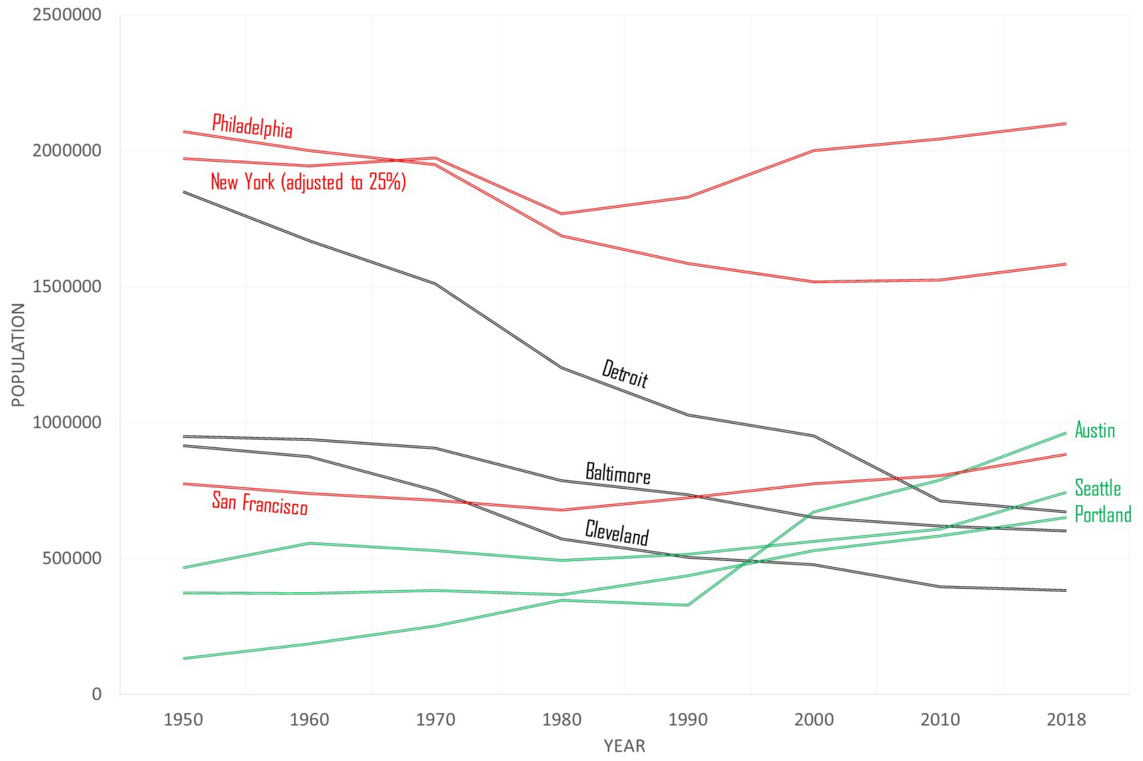

FIGURE 1 | Selected city population trends, 1950–2018. Data source: US Census. For population between 1950 and 1990, see Gibson and Jung (2002); for population between 2000 and 2010, see censusviewer.com; for population 2018, see censusreporter.org.

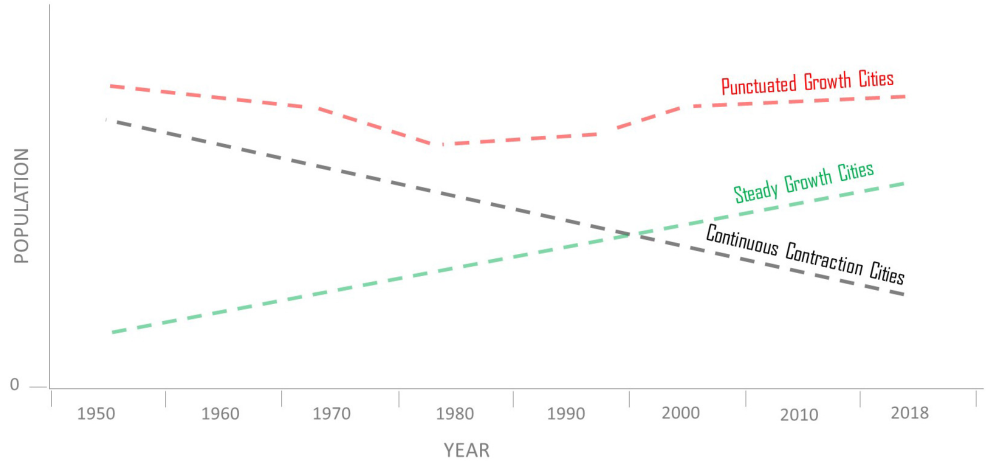

FIGURE 2 | Generalized trend line of population growth in three types of American cities.

for each of the five types analyzed here (i.e., parks, greenways, preserves, gardens, and recreation) and are derived from city files augmented by aerial imagery analysis (see methodological note). For each of the nine cities analyzed in this study, the year the space was acquired/opened is known for at least 90% of all of the qualifying green spaces citywide. Thus, we are reasonably sure that the data we analyze between 1975 and 2014 represents the full set (or very near to the full set) of public green spaces added to these cities in that time. Please see the methodological note in the **Supplementary Material** for a more specific description of the source and procedure used to acquire and develop the green space data.

This analysis does not include private residential yards or landscaped areas, street trees, or most greenspaces built as part of road infrastructure. Thus, the focus here is not on a generalized measure of "greenness," but rather on a specific measure of publicly supported urban green spaces designed for active use or habitat preservation. It is worth acknowledging that a small number of especially informal spaces likely came and went (e.g., a space that started in 1991 but was defunct by 1998) and thus could not be identified when the dataset was developed.

Assigning States of Greening and Racial/Ethnic Change in Tracts

In order to calculate the amount of green space in a tract at a given time, it was first assumed that every green space has a catchment area wherein nearby residents are easily drawn to it. While it is likely that this catchment area is not uniform across green spaces, the widely used measure of 400 m as an easily walkable distance (approximately 10 min) provides a reasonable average baseline estimate of a standard catchment for a greenspace. For this reason, we calculate the number and area of green spaces in a tract by first creating a 400-m buffer around each tract. We then use the formal tract boundary plus the 400-m buffer for all green space calculations by tract. This procedure is important for controlling boundary effects, wherein a tract is directly next to a green space (making it accessible to residents of the tract). Without the inclusion of the buffer area, that green space would not be included in the tract total. Further, in the cases where green spaces comprise most of a tract area but do not reach beyond, this step is required in order to retain that green space in the analysis.

Once the catchment areas were defined, new green spaces were identified for each tract according to 10-year time periods in order to allow for increases in green space provision to be calculated by tract. Only tracts that had at least some greenspace added during the study period (1975–2014) were included in the analysis below. Green spaces were assigned to analysis years according to spaces added in the 5 years before and 4 years after each decennial census year. For example, green spaces added between 1975 and 1984 were assigned to census data from 1980. This approach is meant to capture the "announcement effect" of green spaces (Immergluck and Balan, 2018), so that all new spaces that were either built or likely announced during the 10 years closest to the one being measured in the census data are associated with that year. Time periods to which green spaces were assigned are:

Period 1. *Pre-1975: all green spaces built before 1975*
Period 2. *1980 census data: 1975–1984 green space data*
Period 3. *1990 census data: 1985–1994 green space data*
Period 4. *2000 census data: 1995–2004 green space data*
Period 5. *2010 census data: 2005–2014 green space data*

A categorical grouping based on distance from the median was developed to reflect the "state" of greening for each time period. These categories compare the percent of the tract (plus its 400-m buffer within city limits) occupied by new green space at a given time with the median percent across all populated tracts that added green space in the city at that time. The formula is:

$$\text{area change (ac)} \ = -1\left(\frac{area\ median - area\ tract}{area\ median}\right)$$

The area change (ac) value is the degree of change in green space area in a tract for a given time period relative to the median change in all populated tracts that added green space throughout the city (i.e., a different median value is used for each city and each time period). There is also a zero category for tracts that added no green space during the time period (note that zero tracts are not included in the median calculations). The "area median" value refers to the median percent of area that became green for those tracts that added greenspace (i.e., all non-zero tracts) in the city containing the tract. The "area tract" value refers to the percent of land area within city limits that became green in the tract (plus buffer) for which the ac value is being calculated. For example, if the median new green space area coverage of all populated tracts that added green space in a given city between 1975 and 1984 was 8% and a given tract had 4% new green space coverage during that time, then "ac" for that period is equal to -0.5, indicating the tract was 50% below the median. If the given tract has 12% new green space, then ac is +0.5, indicating the tract was 50% above the median. These ratings were then divided into four "states of greening" described in **Table 1** below.

In order to measure the state of racial and ethnic mix in a tract, we use data developed by Zhang and Logan (2016), which is freely available in the Longitudinal Tract Database (LTDB). For their project looking at racial and ethnic segregation in US cities, Zhang and Logan developed a classification for each tract that designates the presence by tract of whites (w), blacks (b), Hispanics (h), and Asians (a). In order to designate groups as present or absent from a tract, the authors identify a threshold wherein any group with a share in the tract that is less than one-quarter of their average share in all large and multi-ethnic US metros is considered so underrepresented as to be "absent" (for more detail, see Logan and Zhang, 2010; Zhang and Logan, 2016). The minimum levels used as cutoff points vary from decade to decade, reflecting the changing overall composition of the country (minimum cutoffs for whites to be considered "present" are: 1980:20.7%; 1990:20.2%; 2000:18.9%; 2010:17.8%). Crucially, this approach allows researchers to distinguish differences that are inherent to regions' demographic composition from those that simply reveal overall demographic shifts.

TABLE 1 | Greening states: classifications of new green space area in a tract.

Green space area classification	Label	Description
Zero new green space area	Z	There were no new green spaces added to the tract.
Low new green space area	A	The change in percent of tract area occupied by new green space is **notably below the median** (50% or greater) across all cities.
Moderate new green space area	B	The change in percent of tract area occupied by new screen space is **near the median** (less than 50% above or below) across all cities
High new green space area	C	The change in percent of tract area occupied by new green space is **notably above the median** (50% or greater) across all cities.

Bold values refer to the relationship that each category has with the median (bolded for visual clarity).

Using the same threshold as Zhang and Logan (2016), this study employs a simple two-state designation, wherein tracts are assigned "w" if a white population is present above the Zhang-Logan threshold of one-quarter of the average share of whites in all large and multi-ethnic US metros for the given year and "n" if the white population is not present above the threshold. While there is good reason to examine all racial/ethnic categories, this study is primarily focused on the shifting states of whiteness relative to greening. Thus, for the sake of not generating an overly complex number of possible states (which would muddle the state sequence analysis process), the racial/ethnic classifications are simplified to those that, basically, designate which tracts have a sufficient enough presence of whites as to make the white population "present." In other words, we are looking at tracts where the presence of whites is large and visible enough (roughly 20% or higher) as to make them an easily perceived part of the neighborhood makeup.

State Sequence Analysis

Neighborhoods are far from static. Rather, current conditions in a given area of a city are derived from a complex history of demographic and physical change (Grove et al., 2018). We focus on two of these changes—greening of the local environment and racial/ethnic makeup of the local population. Given the above data, we examine the typical sequences followed by the combined state of greening and racial change between 1975 and 2014. We use the eight possible states (four states of greening multiplied by two racial/ethnic states) as the basis upon which to undertake this examination. For example, a tract state for a given time period might be "Aw," meaning it had a relatively low amount of new greening and white population was present above the Zhang-Logan threshold for that year. We examine, as well, if these typical sequences varied by type of city using the three-city growth trajectory classifications described above. Finally, following Borgna and Struffolino (2018) we connected the typical sequences with qualitative data (described below) from a neighborhood representing one cluster in each type of city as a way of enriching our understanding of the general trends found.

In order to determine the sequence pattern of change in greening and racial makeup of tracts, we employ state sequence (STS) analysis. Overall, sequence analysis allows sets of coded categorical data ordered by time to be understood according to general trends, including identification of typical and deviant sequences, and parsing out of difference between groups (such as cities with different growth trajectories) (Ritschard and Studer, 2018). There are three general steps to sequence analysis, each of which requires making methodological choices (Rossignon et al., 2018). The first step is to code data according to categorical sequence states and time periods (this step is described above). The second step is to measure the pairwise dissimilarity between sequences. In this case, that means measuring tract-level differences in the sequence of greening and racial change. The final step is to summarize the dissimilarity between sequences with the aid of a clustering algorithm. This final step allows for typical sequences to be identified.

We developed the state sequence dataset and analysis using the latest version (2.2.0.1) of the "TraMineR" package in R software (version 4.03). For the first step, we built a table in the state sequence (STS) format that included the full demographic data; the city classification; the categorical state of greening (zero, low, moderate, high); and the binary ("white present" or "non-white") state of race/ethnicity for each tract in the nine cities. For the second step, identifying pairwise dissimilarity, we utilized the Dynamic Hammond Distance (DHD) method for calculating how close one race/ethnicity-greening sequence was to another. We selected the DHD method over the commonly used Optimal Matching (OM) approach because DHD is ideal for data where all sequences have the same length and difference across each time period is important from theoretical and analytical perspectives (Lesnard, 2009). For the third step, we employ the Ward's clustering technique with a three-cluster solution in order to identify typical race/ethnicity-greening sequences for the tracts included in our study. The Ward's cluster method is designed to minimize the total within cluster variance, thus generating homogeneous groupings of data. In line with the explorative dimension of this method, we selected the three-cluster solution after analysis of several solutions between two and eight, and subjectively determining that the three-cluster approach produced results that best met the goal of differentiating the key differences among race-greening pathways.

Illustrating Quantitative Trends With Qualitative Data

We selected neighborhoods in Austin (steady growth), San Francisco (punctuated growth), and Cleveland (continuous contraction) as three qualitative case examples to illustrate selected on-the-ground processes that generate one of the three typical sequences of greening and racial change in each city type. Within the scope of this study, we could not undertake qualitative analysis of all of the typical sequences, so we selected neighborhoods that reflect theoretically and practically important trends within the quantitative data. All three qualitative cases shared a municipal commitment to developing new greening interventions since the 1990s.

In each city, we conducted between 25 and 30 semi-structured interviews after pretesting, modifying and selecting a final set of questions for our interview protocol. We spent 1 month in 2018 and 2019 conducting fieldwork in each neighborhood/city, enrolling a variety of participants to maximize the heterogeneity of experiences and expertise on the topics of urban greening; urban development and racialization; and inequalities; covering a range of themes related to re-naturalizing interventions; equity/justice; segregation; and gentrification. We contacted initial respondents through previous research, expert contacts, internet searches, and review of local media articles about each case. We then contacted further respondents through snowball sampling, until reaching theoretical and stakeholder diversity saturation. Respondents included elected officials; urban planning and environmental staff; community-based organizations; neighborhood resident leaders; and other local stakeholders (such as developers, designers, or local experts in the area of urban greening). Although we did not purposely seek older respondents, a variety of respondents (long-term neighborhood activists, environmental non-profits, and/or urban environmental planners) in each city were able to provide historical insights and perspectives about greening changes.

Upon the full transcription of 90% of our interviews, we created a detailed coding scheme based on the main questions addressed by the parent research study. Two primary researchers developed the overarching coding scheme, which was then slightly altered and clarified after extensive team discussions. Coding was completed by one researcher per city, with regular check-ins and verification between all team members to ensure intercoder reliability. Last, we used gray literature to document the greening and urban development trajectory and history of each city, including archival and newspaper articles, non-profit organizations' reports, and media articles.

RESULTS

This study analyzes the relationship between greening and changes in race/ethnicity in tracts that had some greenspace added between 1975 and 2014 across nine American cities classified according to three growth trajectories. For each of the three growth trajectories (and for all cities together), we report below a summary of the overall relationship between changes in race/ethnicity and greening and then visualize the typical trajectories (state sequence clusters) for tracts that greened. We then mobilize the qualitative data to report relevant on-the-ground characteristics of neighborhoods containing tracts that represent a state sequence cluster.

Summary of Quantitative Findings Across All Cities

Overall, by area, tracts that shifted toward a whiter population were overrepresented with new green initiatives beginning mostly in the 1990s–2000s (not prior), but there are differences in this trend by city type. **Table 2** shows the percent of all tract area with population that shifted whiter in two ways. In the top half of the table, we use the Zhang-Logan threshold (which requires that white population is present at or above 25% of the average across all large and multi-ethnic American metropolitan areas at the time—this means the tract had at least 17–21% white population depending on the year), to identify tracts that shifted whiter. Since the threshold (17–21%) needed to meet the Zhang-Logan standard for "present" is somewhat high, the bottom half of the table also reports the tracts that saw any increase in white population in order to provide a more complete picture. The Zhang-Logan approach is focused on identifying neighborhoods where white population has a strong and visible foothold and the simple increase approach is focused on identifying areas where whiteness is increasing to any degree. Both approaches matter for different substantive reasons.

Across all nine cities, 3.17 and 8.47% of the populated tract area (i.e., percent of area covered by tracts—not percent of the number of tracts—was used to control for variation in tract size) shifted whiter according to the Zhang-Logan and simple thresholds, respectively. With all cities considered together, over the entire time period, these tracts were overrepresented with greening according to the simple (lower) but not the Zhang-Logan (higher) threshold. The Zhang-Logan threshold tracts were only overrepresented with greening in the 2000 and 2010 periods. Generally, the overrepresentation of greening in tracts that became whiter between 1980 and 2010 appears to be a phenomenon that was largely driven by initiatives that may have been planned in the 1990s but were rolled out in the 2000s and 2010s. However, importantly, this is not a uniform trend across all three city types.

By type of city, there are unique patterns with regard to the overall relationship between greening and racial/ethnic change. First, regardless of the threshold used, steady growth cities show an overrepresentation of greening in tracts that became whiter for the later periods of the study. Within steady growth cities, when looking at the stricter Zhang-Logan threshold, the tracts that got whiter received new green space area that was roughly twice the percent of area covered by the tracts between 1995 and 2014 (which corresponds to the 2000 and 2010 census time periods in the table). This trend for steady growth cities increases for tracts with any increase in whiteness to an area-based overrepresentation of roughly 2.5.

Next, continuous contraction cities show no overrepresentation of greening in tracts that became whiter for the period studied. This trend holds regardless of the threshold used for designating tracts that became whiter. Two underlying dynamics are important to consider for these cities. First, there were very few tracts that became whiter. These cities generally experienced a decline in white population overall for the time period studied. Second, the area of formal green spaces in the cities was largely stagnant or declining during this time period due to budget restraints but new informal gardens and local greening initiatives grew in non-white tracts. Put differently, there was an emphasis in continuous contraction cities on smaller, informal greening that tended to occur in mostly non-white tracts.

Finally, for punctuated growth cities, greening-racial/ethnic change has a longer and more episodic history. In these cities, the Zhang-Logan threshold shows no overrepresentation of greening in tracts that became whiter, but the simple increase threshold offers a different picture. According to the simple threshold (bottom half of **Table 2**), punctuated growth cities had overrepresentation of greening in tracts that became whiter in the period between 1985 and 1994 (marked as the 1990 census year in the table) and the period between 2005 and 2014 (marked as the 2010 census year in the table). These periods coincide with the efforts of early "frontier" gentrifiers of the 1980s in these cities (Smith, 1996) and the early push toward local sustainability initiatives of the 1990s, many of which involved urban greening efforts in large cities. New greening efforts also took place toward the end of the 2000s, especially so in formerly industrial districts such as Hunters Point-Bayview in San Francisco or north of Center City in Philadelphia. In all, during clearly identifiable periods where punctuated growth cities pushed new greening initiatives, tracts that became whiter were overrepresented by area with greening.

TABLE 2 | Representativeness of new greening in tracts that shifted toward a whiter population (notable shift above the Zhang and Logan threshold and any shift) between 1980 and 2010.

	Year	All cities (%)	Steady growth cities (%)	Punctuated growth cities (%)	Continuous contraction cities (%)
Percent of all tract area that shifted notably white (above Zhang-Logan threshold)		3.17	2.17	4.60	1.18
	1980–2010	2.38	2.77	2.19	0.48
	1980	2.77	1.92	3.79	0.52
	1990	0.13	0.08	0.19	0.00
	2000	**5.87**	**4.94**	0.83	0.31
	2010	**3.39**	**4.16**	1.10	0.64
Percent of all tract area with any shift white		8.47	8.62	16.97	8.78
Percent of all new green in tracts with any shift white	1980–2010	**11.32**	8.02	**17.71**	4.57
	1980	3.23	3.45	2.79	4.87
	1990	**23.10**	2.28	**49.15**	4.17
	2000	6.49	7.34	5.12	3.66
	2010	**18.76**	**21.49**	**17.21**	5.08

Bold values highlight the instances where greening was over represented in tracts that shifted white relative to overall area of tracts that shifted white.

Steady Growth Cities

When considering the data for steady growth cities that had a more or less linear upward growth trend over the period studied, the three most representative clusters of state sequences in **Figure 3** demonstrate a strong link between greening and white populations. For each cluster type, **Figures 3–5** show the two most frequent sequences in black and gray lines (reflecting the modal trend) for that cluster. For a more detailed depiction of the sequences in each cluster see the **Supplementary Material**. The first cluster of tracts (Type 1, *n* = 63), labeled "high green/white," mostly stayed white over the entire period and had intense periods of greening. The most common trends for "high green/white" tracts were for large areas of greening to occur in these tracts during the period between 1985 and 2005, with lower levels outside of that time period. The second cluster of tracts (Type 2, *n* = 205), labeled "low green/white" also mostly stayed white over the entire period, but had lower levels of greening scattered throughout. The third cluster of tracts (Type 3, *n* = 13), labeled "high green/shift white," mostly shifted from non-white to white immediately after or immediately before intensive periods of greening, and thus demonstrate the historical pathway of co-occurring greening and increased whiteness in areas that displace people of color. Interestingly, the "high green/shift white" cluster of tracts shows that intense greening may come before or after the shift toward whiteness in steady growth cities.

In order to demonstrate how these historical patterns in the relationship between greening and whiteness translate into on-the-ground experiences in steady growth cities, we draw on qualitative data gathered in the central East Austin neighborhood of Austin, Texas, which contains core tracts that fall within the "high green/shift white" (Type 3) cluster. In the 1990s, a lot changed for Austin. It transformed from a relatively sleepy college and government town into the fastest growing large US city (Kolko, 2016), labeled the second-most overvalued real estate market in the country and the "most difficult" real estate market in Texas (Gross, 2018). Its growth was propelled by the high technology industry, with firms such as Dell, AMD, and others with secondary offices including Adobe, Amazon, and Apple driving much of the sector, which thrust the city into the heart of the global economy. The growth in technology design and production industries was sparked by strategic investments and a local culture that was increasingly attractive to a highly paid young workforce (Smilor et al., 1989), but the fact that the city had extensively preserved its natural ecosystem and park system, with enviable spaces such as Barton Springs Greenbelt, Walnut Creek Metropolitan Park, Red Bud Isle, or Mayfield Park, was essential to the city's attractiveness. High income technology workers could afford to choose between locations based on such intangibles as the local environment, so cities like Austin with a rich store of green amenities and attractive local culture had an advantage (Florida, 2004).

Austin wasn't designed to accommodate the rapid growth it experienced at the turn of the twentieth century, and so had to plan quickly for where the new development would go. Due to a racist historic land use policy passed in the 1930s that required (through legal mandate) non-white residents to move to the east side of town, which was also where most heavy industry was concentrated, the most underdeveloped area was also where many people of color lived. In the 1980s, community groups in the East Austin neighborhood began organizing to clean up and remove many of the industrial sites near their homes. One organizer said of the time, ". . .we made history by rezoning over 600 properties. . .you had to address what was going on but then you had to address what was allowing all these polluting and industrial facilities in our communities, and it was through land use and zoning that was doing that." As a result of these changes to land use and the greening initiatives that were underway by the mid-1990s, East Austin was seen by many as "ready" to receive the new development, a crucial condition for its formal designation by the city as a "desired development zone," which shifted new growth in that direction.

By the early 2000s, a new development push had taken root in East Austin, which by then was becoming one of the most rapidly gentrifying neighborhoods in the country associated with widespread displacement of low-income people of color (Maciag, 2015). Meanwhile, a new round of greening and associated

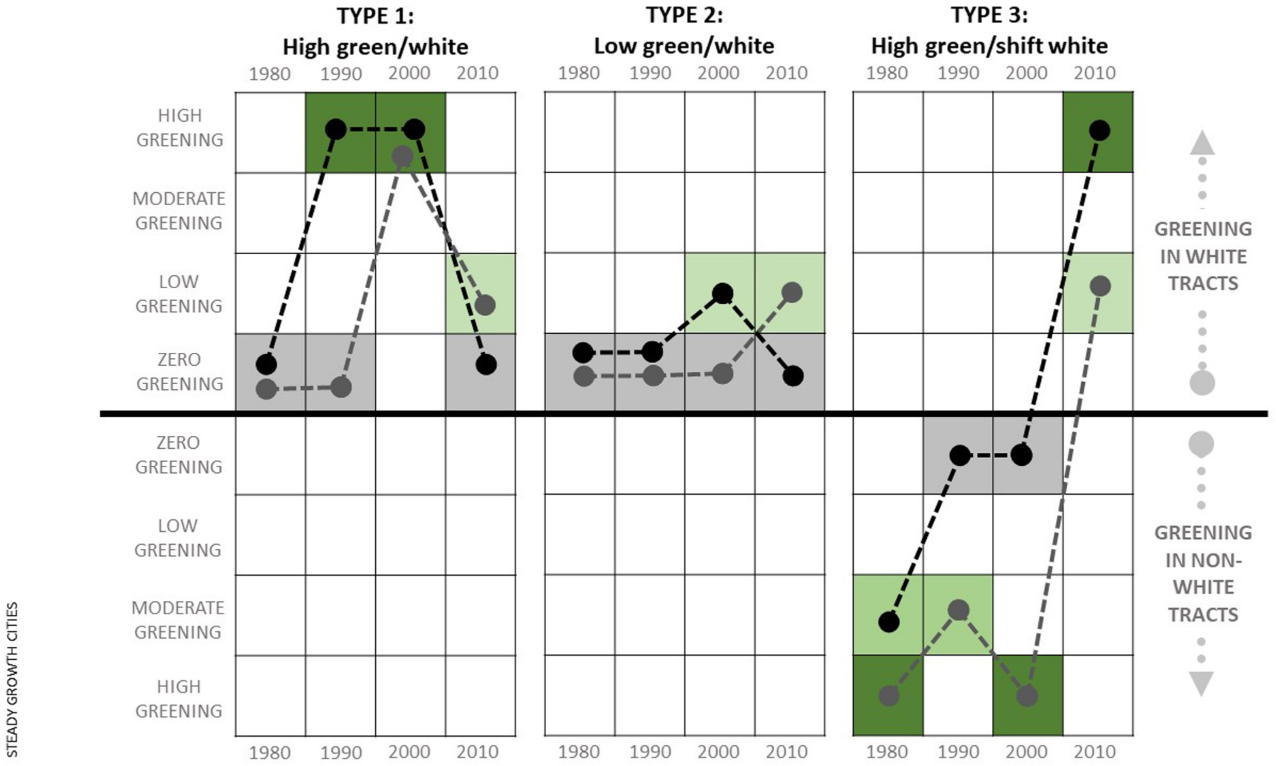

FIGURE 3 | Steady growth cities typical trajectories: three-cluster solution for race/ethnicity-greening trajectories of all tracts that had any greening between 1975 and 2014. The black line represents the most common sequence and the gray line represents the second-most common sequence. Together, these lines reflect the modal sequence, or most common trend, within a cluster. Detailed cluster results can be found in the **Supplementary Material**.

green rhetoric (Garcia-Lamarca et al., 2019) was used as a way to further draw development to East Austin. One city official commented, "the emphasis [became]. . .getting some parkland established out ahead of development actually to help lure development out toward that way [the east side]. . .We were trying to buy land out there ahead of development and hopefully draw development out there with parkland. And it was hugely successful. Now all of the development has followed those tracts of land."

In the end, greening was used to "prepare" East Austin for development in two ways at two different time periods, both of which show up in the Type 3 steady growth cluster. Intensive greening in the early 1990s was focused on cleaning up formerly industrial land in such a way as to make the area ripe for designation as a "desired development zone." Then, intensive greening in the early- to mid-2000s took the form of new parks and greenways as a "lure" for development to the area, with acquisitions for parks such as Roy Guerrero Metro Park and East Boggy creek Greenbelt, and contributing to green gentrification trends (Long, 2016; Garcia-Lamarca et al., 2019). That "high green/shift white" clusters are one of the typical historical trajectories in steady growth cities shows that the intensive greening associated with a shift from non-white to white population seen in East Austin was a defining characteristic across these cities.

Continuous Contraction Cities

When considering the data for continuous contraction cities, the three most representative clusters of state sequences shown in **Figure 4** demonstrate that, while small-area greening has widely occurred in non-white tracts, the most intense levels of greening have mostly occurred in white tracts. The first cluster of tracts (Type 1, *n* = 136), labeled "variable green/white," retained a notable level of white population (above the Zhang-Logan threshold of at least 25% of the nationwide average for whites across all metros) throughout the study period with various levels of greening including high- and low-area greening concentrated in the 1980s and the 2000s. Most importantly, "variable green/white" tracts are the only areas of continuous contraction cities that saw any high-area greening and are the only cluster type that stayed white throughout the study period. The second cluster of tracts (Type 2, *n* = 26), labeled "low green/shift non-white," changed from white to non-white (white population dipped below the threshold to be considered "present") in the 2000s with low-area greening carried out in the non-white period, but not prior. The third cluster of tracts (Type 3, *n* = 134), labeled "low green/non-white," stayed non-white throughout the study period (or, in some cases became non-white very early in the study period) with low-area greening concentrated in the latter years.

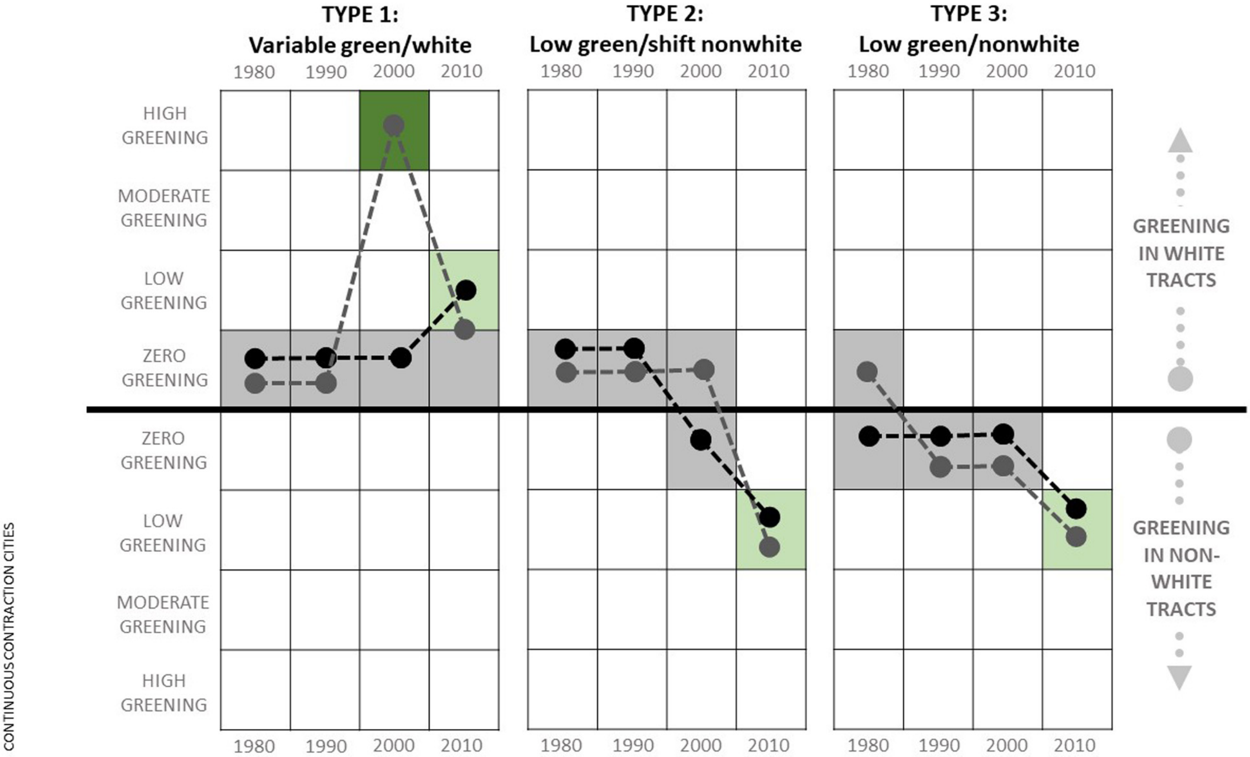

FIGURE 4 | Continuous contraction cities typical trajectories: three-cluster solution for race/ethnicity-greening trajectories of all tracts that had any greening between 1975 and 2014. The black line represents the most common sequence and the gray line represents the second-most common sequence. Together, these lines reflect the modal sequence, or most common trend, within a cluster. Detailed cluster results can be found in the **Supplementary Material**.

There are thus two major, polarized trends in continuous contraction cities: limited but intense new greening in white areas only and widespread but small-area greening in non-white areas.

In order to demonstrate one of these clusters, we focus on qualitative data gathered in the Detroit Shoreway neighborhood of Cleveland, Ohio, an area containing several tracts in the "variable green/white" (Type 1) cluster of continuous contraction cities. Since the 1960s, drastic economic and population decline have defined Cleveland's trajectory, with blue-collar employment plummeting to only 29% in 1986, a population loss of 380,000 inhabitants by 2019 (US Census Bureau, 2019), and thousands of uninhabited homes and vacant lots. From an environmental justice standpoint, the Cuyahoga River and its adjacent black areas have historically been the symbol of environmental degradation and industrial and other types of waste contamination. Since the beginning of the 2010s, however, Cleveland has experienced some cultural, art, and food-based revival, with new real estate development and investment, mostly in racially mixed areas of the city that would fall into the Type 1 cluster (whites are present above the Zhang-Logan threshold of at least 25% of the average across all metros) for continuous contraction urban areas. In contrast, one respondent stated what they saw as an impression held widely by local residents that some other areas, like East Cleveland, are "too poor and too black" to be seen as targets for reinvestment or gentrification by private developers and homeowners.

Since the late 2000s, the restoration of the Cuyahoga River and Lake Erie, as part of the transformation to rebrand Cleveland as "a green city on a blue lake" and depict both spaces as assets (Samanta and Kellogg, 2019), has been triggering other nearby environmental efforts. Neighborhoods like Tremont, Ohio City, and Detroit Shoreway are being revived through new housing developments and greening investments. As of 2018, the Northern section of Detroit Shoreway has seen new luxury housing built next to the restored Edgewater park. Although the park is now an attractive amenity with lakeside beaches, boat docks and fishing areas, and picnic sites, local black families report being unable to find affordable housing in its vicinity, and thus to enjoy greater proximity to the new green spaces. As one resident shared, "In the 5 years since my son bought his house [in the Detroit Shoreway neighborhood], there's no house in this [Detroit Shoreway] neighborhood that they [my daughter and her partner] could find, that's move-in ready that they could afford. And they're not, they're not super low income."

Developers reported during interviews that the new green revival of part of the neighborhood was a major asset, and particularly highlighted their ability to sell high-end homes by marketing Edgewater Park and another recent park called Battery Park as a core asset in Detroit Shoreway. The Detroit Shoreway neighborhood is close to downtown, close to motorized high-speed roads, full of sit-down restaurants and shops, and also hosts a sizable white population (49%), all making up the "attractiveness" of Detroit Shoreway to outsiders. One of them highlights the green value of Battery Park for residents looking for a more dense, proximate-to-downtown "sustainable lifestyle": "From Battery Park you can walk to the Gordon Square area, you can walk to restaurants, you can walk to Capitol Theater, there are theaters, bars, restaurants, retail and people are really attracted to the walkability and accessibility. To walk to Edgewater is a beautiful thing as well because there are the walking paths over here, so you can actually get to Edgewater Beach." Detroit Shoreway shows that "variable green/white" (Type 1) tracts in Continuous Contraction cities, which have sustained white populations above the Zhang-Logan threshold and experienced concentrated periods of high-area greening, are key areas for revitalizing declining economies and population growth, but also for strengthening the link between higher income whiter neighborhoods and greening in cities—a process which undermines racial green justice.

Punctuated Growth Cities

When considering the data for punctuated growth cities that recovered from prior declines, the trends associated with high and low growth in other cities can both be seen. In other words, there are similarities with steady growth and continuous contraction cities. The three most representative clusters of state sequences shown in **Figure 5** demonstrate an historic association between tracts with a high level of greening that sustained white populations and tracts with a low level of greening that sustained or shifted toward non-white populations.

The first cluster of tracts (Type 1, n = 281), labeled "spiky green/white," represents the bulk of the areas that greened. These areas retained white population above the Zhang-Logan threshold of at least 25% of the average across all metros throughout the period studied and had spikes of concentrated greening in two periods—the late 1980s and the early 2000s. Most of the early (1980s) intensive greening in these cities occurred in Type 1 tracts, implying that these actions were associated with sustained whiteness in the following decades. The second cluster of tracts (Type 2, n = 32), labeled "low green/shift non-white," had the lowest occurrence and showed a shift from white to non-white in the 1980s or early 1990s with greening scattered throughout. These tracts represent the tail-end of a long prior period of "white flight" away from these large cities. Philadelphia was especially still experiencing this trend in the 1980s and, thus, has a concentration of Type 2 tracts. Greening in these tracts was mostly relatively low early-on and then non-existent in more recent years. The third cluster of tracts (Type 3, n = 170), labeled "low green/non-white," mostly stayed non-white (with a few exceptions) and had greening concentrated around the early 2000s. Type 3 tracts commonly had low or moderate levels of greening by area, implying that later and lower levels of greening were not associated with substantial racial or ethnic shifts in these cities (however, see qualitative case below for an example of the precarious nature of this trend).

In order to demonstrate how these trends translated into on-the-ground experiences, we draw on qualitative data gathered in the Bayview-Hunters Point neighborhood of San Francisco, which contains core tracts that fall within the Type 3 cluster for Punctuated Growth cities. San Francisco has been a node in the globalized, tech-driven economy since the 1980s, with its green identity being a front-and-center attractor of new high-income and majority white residents. Until recently, some heavily industrial communities such as Bayview-Hunters Point were considered hostile to new developments and to migration by upper income or white residents. Historically the site of intensive municipal, regional, and military industrial facilities, including the former PG&E power plant; the Southeast Sewage Treatment plant; the naval shipyard; waste sites; diesel freight transportation; and two freeway lines, Bayview-Hunters Point residents have continuously been exposed to toxic smells, waste, and acute negative health outcomes (Dillon, 2014; Liu and Shatford, 2019). Local residents and organizers almost universally describe a pattern of municipal neglect of the area as a key factor in exacerbating these conditions.

However, since becoming a federal Superfund site in 1989 and benefiting from remediation, 500 acres of the neighborhood and its waterfront have been or are being redeveloped into 13,000 new homes and more accessible and integrated green and blue spaces (e.g., Hunters Point Shoreline, Candlestick Point, and India Basin). Today, the neighborhood includes one of the few natural areas within the city adjacent to the Bay (India Basin Neighborhood Association, 2019) and it has unique natural and recreational features such as a recreational trail on a segment of the Bay Trail, shoreline access for recreation, one of only a few tidal salt marsh wetlands in San Francisco, and restored habitats for shore birds.

Yet, everyone we interviewed (regardless of background) was acutely aware that the area is seen as one of the last development frontiers in San Francisco. The southeastern waterfront including Bayview-Hunters Point now sits adjacent to and is planned for new luxury development and retail gentrification through new restaurants, shops, and services. For example, the India Basin mixed-use project is envisioned as "a pedestrian-first, human-scaled, bicycle-friendly, residential village where neighborhood amenities are within a short walk," including 15.5 acres of open space. While historically populated by working-class African American families and other ethnic minorities (Chitewere et al., 2017), an increasing number of white and Asian residents with higher incomes have recently purchased homes in the area. As an example of the housing boom characterizing the area, recently homes have gone for sale for more than $800,000 for a 2-bedroom—well above the financial capacity of existing residents. This shift has created a new sense of acute insecurity, wherein the benefits of the new environmental cleanups are almost lost on existing residents of color. In the words of one resident, "everyone is trying to stabilize housing. If you don't have a place to stay, you don't care that the sea level is going to rise, you don't care that your air is bad, you don't care about the quality of the environment."

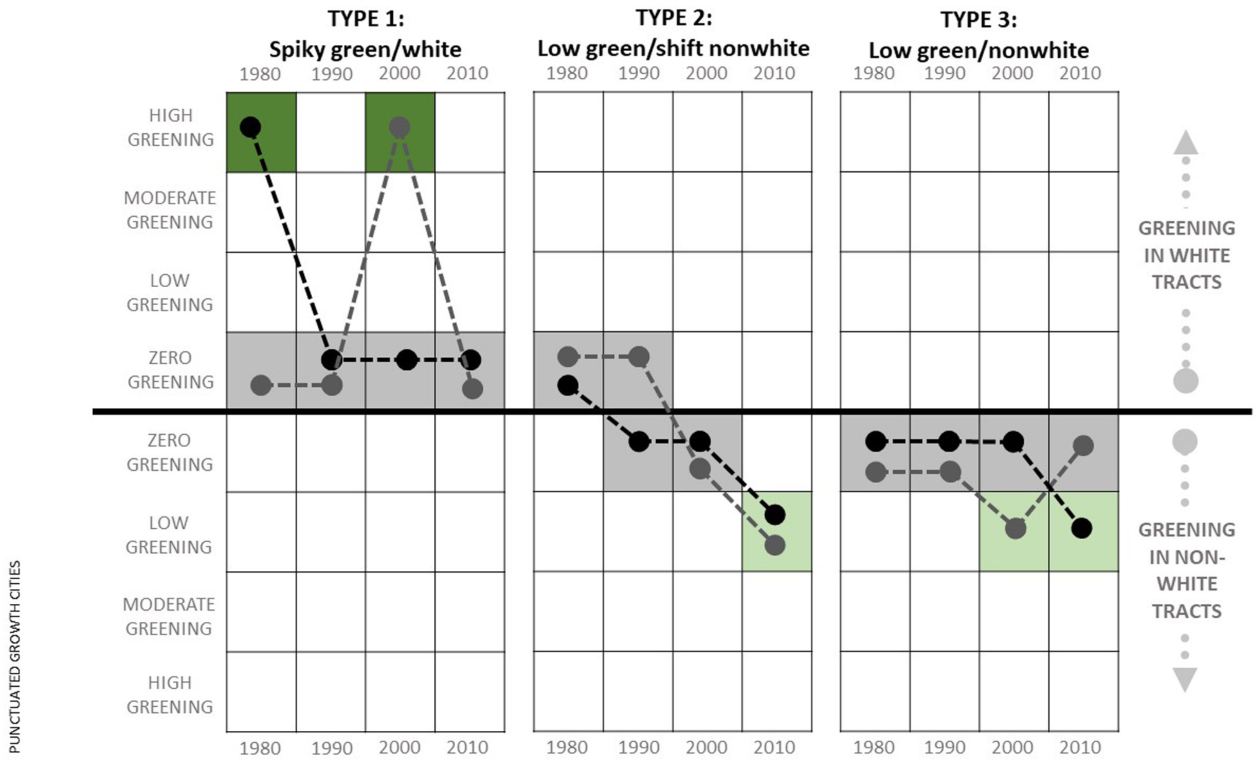

FIGURE 5 | Punctuated growth cities typical trajectories: three-cluster solution for race/ethnicity-greening trajectories of all tracts that had any greening between 1975 and 2014. The black line represents the most common sequence and the gray line represents the second-most common sequence. Together, these lines reflect the modal sequence, or most common trend, within a cluster. Detailed cluster results can be found in the **Supplementary Material**.

The sense that demographic shifts generate a more acute threat than environmental conditions is sensible given recent census data. While in 1980, 72% of residents self-identified as African American, today only 27% identify as black (of any sort). This decline is summarized by the San Francisco Bayview Newspaper as "the smallest black population and the fastest exodus of any big city in America" (Epps, 2020). In contrast, according to the 2009–2013 American Community Survey, 19.5% of all neighborhood residents are now white. This has led to a deep suspicion among Bayview-Hunters Point residents of the motivations behind greening efforts, and they have started to resist greening. As one citywide greening group said, "We have encountered some real resistance to [greening] in this part of town…If we look at a map of where we have been active over the last four years, you know, this is the one area where we have been unsuccessful…and this is primarily about social justice concerns." Bayview-Hunters Point demonstrates that, while many Type 3 tracts have not yet seen a substantial shift toward whiter populations, the people living in these tracts are quite aware of the trends seen in Type 1 (spiky green/white) areas of their city. People of color living in recently greened areas expressed feeling highly vulnerable to displacement in punctuated growth cities, even if those are still contaminated and industrial areas. Our quantitative data analysis supports their suspicion that, as these areas become extensively greened, those who will benefit from new restored and cleaned-up grounds are more likely to be white residents—high profile green projects like those in Bayview-Hunters Point and their associated branding seem to accompany shifts toward a whiter population.

Trends Across All Cities

While each city growth trajectory has a unique register of association between greening and the presence of white population over time, the recurrence of similar cluster characteristics across different trajectories shows that there are some overall commonalities. Steady growth cities have the strongest and most uniform tendency toward a trend expressed across all cities: clusters of tracts that greened and sustained or shifted toward white populations were the only ones that were defined by high-area, concentrated greening. We do also see a cluster of low levels of greening in white tracts in steady growth cities, but this cluster is mostly a consequence of the fact that these cities have a high percentage of white population so nearly all greening occurred in tracts where white population was present. Thus, **Figure 6** shows high and low greening occurred in tracts that sustained or shifted toward a white population for steady growth cities, but not for any other type of city. The key trend for high area greening, though, is that it occurred mostly in white tracts across all three types of cities.

Meanwhile, continuous contraction cities have the strongest and most uniform tendency toward another common trend: all clusters of tracts that sustained or shifted toward non-white

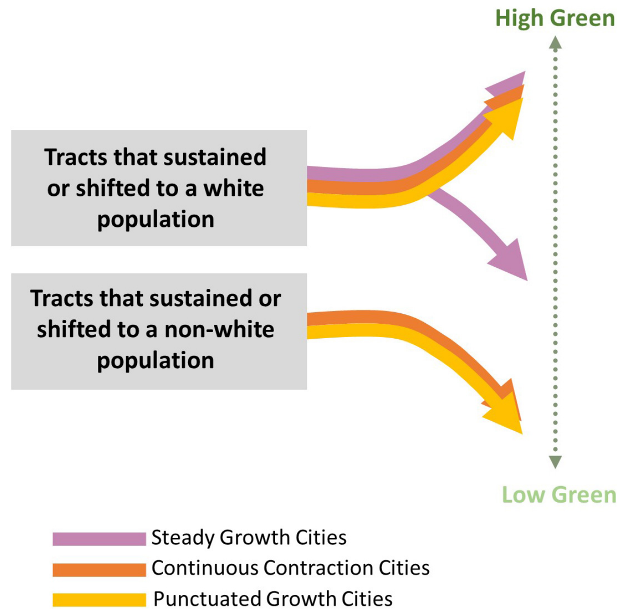

FIGURE 6 | Greening trends in white and non-white tracts.

populations were defined by low-area greening. Overall, combined trends show clusters with an association between high-area greening occurring in tracts with notable white populations and low-area greening concentrated in non-white areas (with the exception of steady growth cities where there was little to no greening in tracts that did not, at some point, have white representation). **Figure 6** visualizes this overall commonality across all growth trajectories.

DISCUSSION

In order to serve as an effective historical diagnostic for urban greening, our findings need to be filtered through an understanding of the underlying processes of urban land use planning that caused them (Heynen et al., 2006; Safransky, 2014; Anguelovski, 2016; Schell et al., 2020). In general, the associations we find make sense when considering the intertwined legacies of housing segregation, gentrification, and urban greening (Grove et al., 2018; Roman et al., 2018). Our findings reflect an underlying process wherein racial difference has been used to generate monetary value through home ownership (Gotham, 2000; Fluri et al., 2020) and been demarcated by policy (Fields, 2018). Urban greening is wrapped up in this underlying process.

In the US, housing finance has facilitated benefits to flow to white homes more readily than non-white (Aaronson et al., 2017; Rothstein, 2017), with an extreme example being what is referred to as "redlining." From the 1930s through several decades past World War II, the federal agency responsible for guaranteeing home loans formally reinforced—by drawing red lines on maps—the widely held racist notion that non-white neighborhoods were "high risk," thus effectively excluding non-white areas from access to low-cost mortgage lending guaranteed by the government (Squires and Woodruff, 2019). Predictably, redlined neighborhoods in cities such as many of those in our sample—Cleveland or Baltimore in particular—became severely disinvested and lost population during a roughly 50-year period of white flight to suburban areas. This mass exodus left many underinvested urban neighborhoods in those cities with under-maintained green spaces.

By the mid-1980s, though, two processes were taking shape that would alter the fate of many redlined neighborhoods and that shapes the findings we report. First, banks developed lucrative and predatory high-interest loan products targeted toward high-risk areas and began making predatory loans widely accessible. Second, a renewed interest on the part of middle-class white Americans and an emerging awareness of "rent gaps" made these neighborhoods a target for gentrification that caused widespread residential displacement of non-white residents. As a result, many neighborhoods that were redlined in the 1930s started to be gentrified by the 1980s (Wyly and Hammel, 1999), and our results for Austin, San Francisco, New York, Philadelphia, Seattle, and Portland especially demonstrate this trend. For example, 87% of San Francisco's gentrified neighborhoods in 2018 were also redlined in prior periods (Urban Displacement and Project, 2020). This effect was common in cities throughout the country, especially those that experienced a post-industrial economic boom around the turn of the twentieth century (Wyly and Hammel, 2004).

Meanwhile, by the 1990s, our analysis illustrates the outcomes of a process by which the selection of which formerly redlined neighborhoods would be targeted by gentrification investments was increasingly driven by actions undertaken within urban sustainability and greening circles. By the early 2000s, this link between gentrification of formerly disinvested neighborhoods and the global push toward greener cities came to be seen by some as a consolidated process of urban eco, green, or environmental gentrification (Dooling, 2009; Checker, 2011; Anguelovski et al., 2019), an argument that has been especially applied to cities such as New York, Seattle or Portland in our sample. As parks, gardens, and greenways started to be seen as investment strategies that would prepare areas for increased property value due to greater attractiveness (Immergluck and Balan, 2018), greening in those cities became linked with the racialized history of gentrification. Urban greening proponents could no longer claim to be innocent bystanders as cities became more unequal and elite (Connolly, 2018; McClintock, 2018; Garcia-Lamarca et al., 2019) and thus centers of green privilege for upper-class white residents.

In booming housing markets such as New York and San Francisco, our results show that especially the push toward presumably more visible concentrated greening was associated with sustained whiteness or a shift from non-white to white—a consequence in these cities of gentrification processes. As one organizer we interviewed in Austin put it, the interests shifted toward a "different shade of green"—referring at once to the green of American money and the continued push toward a whiter cultural norm that came with increased monetary value. Safransky (2014) described this as a process of greening becoming a tool for the settler colonial approach to urban space. Going back to Marcuse (1997), racialized residents in the new green "citadels" of fast growing post-Fordist cities have been increasingly excluded and often have had to move to "outcast [gray] ghettos."

From Historical Diagnostic to Social-Ecological Justice

What are alternatives to the racialized green histories we see in these nine American cities? One response to the racial blinders worn by earlier urban greening agendas was articulated by Heynen (2016) as "abolition ecology." In this alternative, outcomes of recent decades of efforts to re-nature cities point toward a need to rethink the study of urban ecology by examining urban natures first and foremost through the lens of antiracist, postcolonial and indigenous theory. This approach points toward what some scholars have started to call "green reparations" (Draus et al., 2019). The green reparations approach explicitly links urban greening initiatives with efforts to atone for historical violence committed against racial or ethnic groups in a given city. For example, in a recent study comparing the use of greening in Berlin (Germany) and Detroit (United States), the authors found examples of green spaces that served as curative bridges with explicit recognition of historical racial violence embedded in them. Another case is the planned 11th street bridge park in Washington DC, which, despite staying shy of returning land to displaced black residents, aims at allowing local control over development in the neighborhood through a linked land trust (Avni and Fischler, 2019).

Abolition ecology and green reparations represent ways of thinking and acting from an urban ecology perspective that allows the historical diagnostic developed here to be mobilized for what Steil (2018; see also Steil and Delgado, 2019) refers to as "anti-subordination planning." Anti-subordination planning calls for the evaluation of all interventions in urban space from the perspective of effect, rather than intent. If the effect of an action is to perpetuate the subordinate position of a disadvantaged group, then it violates the tenets of anti-subordination planning, regardless of intent. It would be hard to argue against the assertion that urban ecology and urban environmental planning are at times focused only on process and intent, while abolition ecology, green reparations, and anti-subordination planning seek to force an accounting for effect, which is also the primary purpose of the historical diagnostic presented in this paper.

While it is not easy to conceive of exactly what just greening looks like when it embodies abolition ecology principles by enacting green reparations through an anti-subordination planning mode that respects the lived experience of marginalized populations (Anguelovski et al., 2020), history provides a place to start. When we know the pattern of relations between greening and whiteness and the social-ecological outcomes of a city, we can ask how and where that pattern needs to be changed. We can use the patterns alongside nuanced qualitative understanding of the context to develop strategies for creating a new history that does not passively accept the social effect of greening as an unmovable historical artifact. Rather, an historical diagnostic brings into relief the contours of effect that prior actions had by outlining the detailed historic patterns. We can start recognizing how the history of greening—a seemingly apolitical "urban project" sold as a win-win, universal benefit with ample health, economic, and environmental qualities—has deep links to the exclusionary and racist roots of urbanization that manifest at different points in time and space across and within cities.

CONCLUSION

Our goal in this paper was to analyze the co-evolution of greening and whiteness in nine American cities with different growth trajectories. We propose here a novel quantitative, spatial, and qualitative mixed methods analysis of trends and advance existing research at the intersection of urban ecology, green justice, and racialization of urban space. Specifically, we sought to answer three questions. First, we asked: What are the common sequences across all nine cities with regard for changes in green space and changes in the racial makeup of surrounding areas? We find that the combined trend within the state sequence analysis shown in **Table 2** and **Figure 6** demonstrates a basic underlying relationship wherein higher area greening is more associated with tracts that either sustained or shifted toward whiter populations in American cities. There are, though important variations across time and type of city.

Second, we asked: Does this pattern vary according to the historical growth trajectory of the city? While there is an underlying narrative of greening and whiteness, our hypothesis that there are three sub-histories to this narrative was affirmed. The tight interlock between race/ethnicity and socio-economic opportunities in American society appears to be visible within urban greening outcomes and growth seems to shape that relationship to a substantial degree. Each of the three growth trajectory histories point toward different relationships between greening and whiteness. For the most rapidly growing cities—the steady growth cities—greening was either closely associated with white areas (Type 1 and Type 2 clusters) or linked to a process of shifting areas from non-white to white (Type 3 cluster). For the cities that lost population continuously and saw a declining GDP—the continuous contraction cities—there was a trend for high-area greening to be associated with whiter areas (Type 1 cluster) and low-area greening to be associated with non-white areas (Type 2 and 3 clusters). For the largest cities that saw a resurgence during the study period—the punctuated growth cities—intensive greening in the 1980s and early 1990s was associated with a sustained white population (Type 1 cluster), while low or moderate levels of greening in the 2000s was more associated with sustained non-white populations (Type 2 and 3 clusters).

Finally, we asked: Can a generalized relationship be determined with regard to the relationship between greening and racial change in these cities? To answer this question, we look at the subset of clusters where racial change occurred (steady growth Type 3; continuous contraction Type 2; punctuated growth Type 2). The trend of co-occurring greening and a shift from non-white to white was limited for tracts that greened to steady growth cities (Type 3), and this trend was characterized by high levels of greening. This finding does not mean that such a process cannot be found in other cities, but rather that it was only a common trend in cities with intense growth. Meanwhile, the trend of co-occurring greening and a shift from white to non-white only occurred in continuous contraction (Type 2) and punctuated growth (Type 2) cities. In both cases, this trend involved zero early greening and low levels of recent greening, demonstrating the recent push toward small-scale green projects, often led by residents or community activists, in these areas. Overall, the answer to this final question—and the general takeaway of this research—is that greening, growth, and race are tightly interlocked qualities of unequal urban space in American cities. Only in considering this interlock through an integrated historical diagnostic can urban ecological interventions hope to work toward racial justice.

DATA AVAILABILITY STATEMENT

The raw data supporting the conclusions of this article will be made available by the authors, without undue reservation.

ETHICS STATEMENT

The studies involving human participants were reviewed and approved by the Universitat Autonoma de Barcelona. The patients/participants provided their written informed consent to participate in this study.

AUTHOR CONTRIBUTIONS

JC primarily conceived the manuscript, completed all quantitative and spatial analysis, co-developed the qualitative analysis, and primarily drafted the text. IA co-developed the qualitative analysis, contributed to writing sections of the text, and edited the full text. Both authors contributed to the article and approved the submitted version. The authors would also like to acknowledge that some of the greenspace and qualitative data used in this paper are based on datasets developed by several members of the Barcelona Lab for Urban Environmental Justice and Sustainability. In particular, we would like to acknowledge that the qualitative analysis of Cleveland was based on the draft of a book chapter co-authored by Margarita Triguero-Mas and Wendy A. Kellogg, which will be published in full form soon after this article in the forthcoming book published by Routledge and edited by Isabelle Anguelovski and James J. T. Connolly, *The Green City and Social Injustice: 21 tales from North America and Europe*.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fevo.2021.621783/full#supplementary-material

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Valuing the Role of Time in Urban Ecology

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Cities and towns are complex ecosystems with features that can vary dramatically in space and time. Our knowledge of the spatial structure of urban land and ecological systems is expanding. These systems have been investigated across spatial scales, urban to rural gradients, networks of urban macrosystems, and global megalopolises. However, the temporal dimensions of urban ecosystems – such as those related to ecological cycles and historical legacies – are far less understood and investigated. Here, we outline the main dimensions of time that can shape how events in urban ecosystems unfold, which we categorize as: (i) time flows and duration, (ii) synchrony, lags, and delays, (iii) trends and transitions, (iv) cycles and hysteresis, (v) legacies and priming, (vi) temporal hotspots and hot moments, and (vii) stochastic vs. deterministic processes affecting our ability to forecast the future of cities and the species that live in them. First, we demonstrate the roles of these understudied dimensions by discussing exemplary studies. We then propose key future research directions for investigating how processes over time may regulate the structure and functioning of urban land and biodiversity, as well as its effects on and implications for urban ecology. Our analysis and conceptual framework highlights that several temporal dimensions of urban ecosystems – like those related to temporal hotspots/moments and stochastic vs. deterministic processes – are understudied. This offers important research opportunities to further urban ecology and a comprehensive research agenda valuing the "*Urban Chronos*" – the change of urban ecosystems through time.

Keywords: urban dynamics, urban change detection, history, urban development, urban trajectories

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INTRODUCTION

Cities are intrinsic spatial systems, wherein variation in ecological structure and function across small spatial scales is easily observed. As a result, it is no surprise that the spatial nature of cities is the dominant foundation for contemporary urban scholars and practitioners. Cities have been investigated extensively across diverse spatial scales – from local studies of habitat patches, observations across rural to urban developments, to surveys of global urban networks (Gilbert, 1991; McDonnell et al., 1997; Seto et al., 2012; Groffman et al., 2017). Assessments of urban morphologies have a long tradition grounded in ancient cartographic and cadastral records, now translated into modern geographic information systems, with help from big data and remote sensing. With these tools in hand, a large body of research now highlights the extent to which urban ecosystems vary across space, including complex effects of spatial variation on ecological and other facets of cities (Beninde et al., 2015; Ossola et al., 2020).

Most processes within cities – such as urbanization, regeneration, redevelopment, abandonment, densification, and transformation – are spatially structured, but these processes also happen over time. Temporality is often included in conceptual models of the dynamics of urban ecosystems, yet empirical studies rarely quantify changes over time in, e.g., the forces structuring urban biodiversity or land systems over decades to centuries (Marcus and Colding, 2014; Pickett et al., 2017). This general pattern may be due to a lack of robust, standardized data describing events and processes over the period of anthropogenic change. This lack of data has real consequences; in cities, decisions made in one decade can affect future generations of both humans and other organisms. Thus, understanding cities in a temporal context is critical for urban ecological research, practice and management.

Recent efforts toward scenario-based urban planning highlight our limited ability to anticipate the needs of future urban societies (Malekpour et al., 2015). Similarly, strategic urban planning and governance are often little informed by a solid understanding of the urban past (Howlett and Goetz, 2014; Malekpour et al., 2015). How cities are managed remains flawed by implicit assumptions that urban land is a largely static entity, unlikely to change significantly over short time scales, such as that of a single human lifespan or that of a few generations. It is no surprise that space, not time, is still considered the main driver governing cities (Marcus et al., 2019). As put by Davis (2019) in the context of regeneration studies "*this reflects a broader tendency [...] to emphasize the physical and spatial aspects of planned change at the expense of the temporal aspects of transformation.*"

Due to these limitations, the temporal effects of urban change on the ecological facets of cities have been poorly quantified or neglected *tout court*. Like natural systems (Wolkovich et al., 2014; Zelnik et al., 2018), the temporal dimensions and dynamics of urban land are most likely to affect their intrinsic functioning, sustainability, and resilience (Ramalho and Hobbs, 2012). For instance, the signatures of urban and rural past can be found in the very soil where modern cities sit; silent legacies of previous land use and change (Clarke et al., 2015). Time represents an important, yet poorly understood, factor driving all urban events and their dynamics, dictating the pace and direction of change and ultimately affecting the ecology of future urban ecosystems. This is particularly relevant when considering the complex interactions between the temporal and spatial nature of cities and how their interplay can ultimately their ecology (Pickett et al., 2017).

Here, we propose several research directions to guide the investigation of time as a driver regulating structure and functioning of urban ecosystems, as well as its implications for urban ecological research. Selected studies are discussed to highlight critical knowledge gaps and opportunities related to the multiple dimensions of the "*Urban Chronos,*" as we define, the change of urban ecosystems through time. In doing so, we lay out a conceptual framework to move this rich and under-developed aspect of urban exploration forward.

RESEARCH DIRECTIONS TO INVESTIGATE AND PRACTICE URBAN TIME

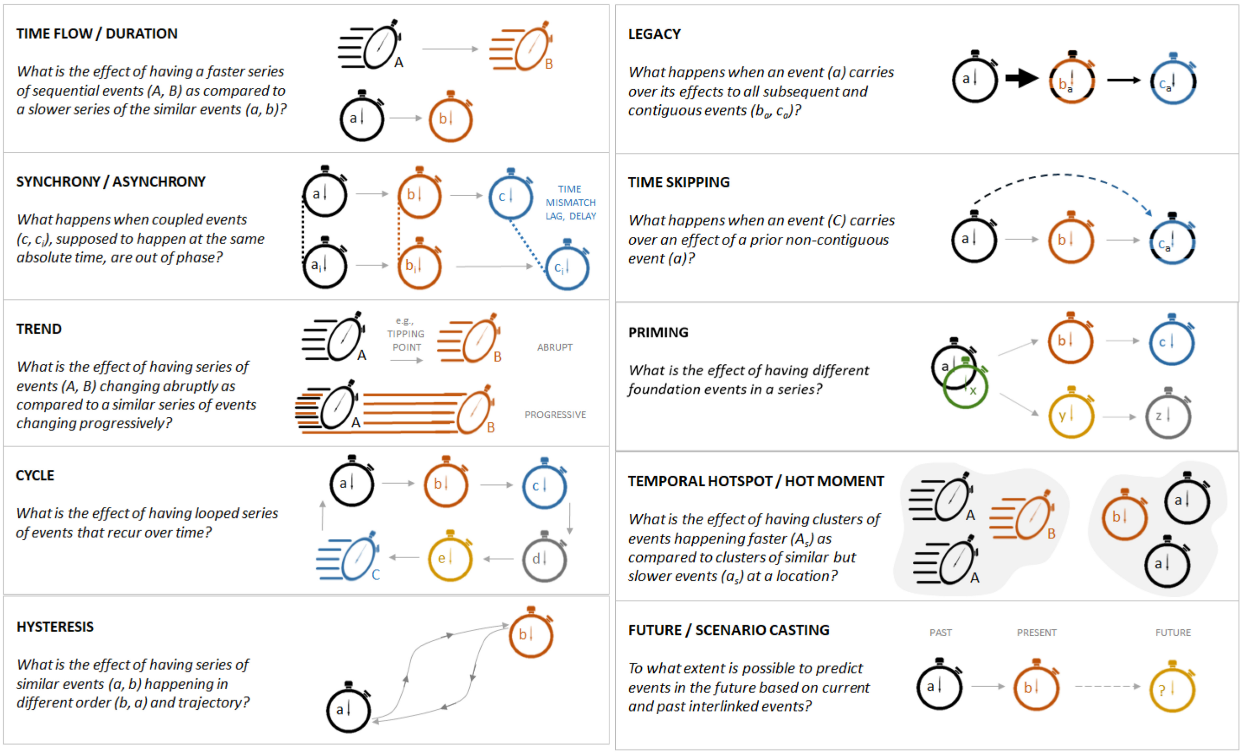
In each section below, we first define and describe the main temporal dimensions of urban ecosystems (**Figure 1**), and key research that exemplifies each dimension. In identifying past research from various fields, we focus on ecological processes and functions – the connections and interactions between biotic and abiotic ecosystem elements that regulates fluxes of energy and matter and the provision of ecosystem services. In doing so, we highlight their interrelationships with other human and technical elements, such as urban planning and management. We then propose key overarching questions related to these temporal dimensions and highlight opportunities for future investigation building upon existing scholarship and practice. For the sake of clarity, we discuss each temporal dimension separately, but we recognize that they can overlap and interact with each other, as well as with the spatial dimensions of urban ecosystems. We avoid proposing a classification of these temporal dimensions, but rather we aim to delineate the main characteristics that can affect structure and functioning of urban ecosystems though the targeted analysis of key ecological literature. In addition, we recognize that some examples could fit more than one temporal dimension or other temporal processes not discussed here.

Time Flow and Duration

Like space, time is affected by significant autocorrelation, whereby a given event at a particular point in time can be affected by other events. Unlike space, however, time has a single directionality, and no change in direction is possible (Wolkovich et al., 2014). Time is a defined dimension where a series of chronological events – or processes – can only develop from past to present to future following a particular series (i.e., time flow). Each of these events develop over a specific length of time (i.e., duration).

Sequences of events might happen in the same order in different places, but the duration of these events may vary with important implications for urban systems. For example, a neighborhood at a specific location could be developed in a mere matter of months, whereas a similar real estate development elsewhere could take years to be built, if not decades. These different time flows can have important consequences on ecological systems and urban organisms, for instance when trying to adapt to long-lasting, chronic vs. short-term, acute anthropogenic disturbances (e.g., noise, pollution) (Grimm et al., 2017). Similarly, relatively small difference in the duration of storms, having similar rainfall intensity and yearly re-occurrence frequency, can determine whether these weather events can generate superficial runoff or not across urban green spaces (Ossola et al., 2015).

Urban change – the flow of homologous events in an urban system – can be ranked from slow to fast (**Figure 1**).



TIME FLOW / DURATION

What is the effect of having a faster series of sequential events (A, B) as compared to a slower series of the similar events (a, b)?

SYNCHRONY / ASYNCHRONY

What happens when coupled events (c, c_i), supposed to happen at the same absolute time, are out of phase?

TREND

What is the effect of having series of events (A, B) changing abruptly as compared to a similar series of events changing progressively?

CYCLE

What is the effect of having looped series of events that recur over time?

HYSTERESIS

What is the effect of having series of similar events (a, b) happening in different order (b, a) and trajectory?

LEGACY

What happens when an event (a) carries over its effects to all subsequent and contiguous events (b_a, c_a)?

TIME SKIPPING

What happens when an event (C) carries over an effect of a prior non-contiguous event (a)?

PRIMING

What is the effect of having different foundation events in a series?

TEMPORAL HOTSPOT / HOT MOMENT

What is the effect of having clusters of events happening faster (A_s) as compared to clusters of similar but slower events (a_s) at a location?

FUTURE / SCENARIO CASTING

To what extent is possible to predict events in the future based on current and past interlinked events?

FIGURE 1 | Exemplification of temporal dimensions, processes and questions related to urban time and its dynamics. Different letters represent different events and are conceptualized as fast (capital letters) and slow-occurring events (lower case letters). Gray arrows between events represent time flows and direction.

For instance, little evidence exists on how past planting of tree species with different growth rates might have contributed to different levels of canopy cover across modern urban landscapes (Roman et al., 2018; Smith et al., 2019). Planting tree species with different ages or lifespans could be linked to greater urban forest turnover (**Figures 2A,B**). Similarly, the use of different tree species with a wide range of safe useful life expectancies [SULEs, i.e., the maximum horticultural duration of a tree stem beyond which maintenance costs exceeds replacement costs, Barrell (1993)] can determine how urban forests change over time under human management.

The different duration of urban events and time flows may affect many aspects of urban ecosystems, raising numerous opportunities for research and practice. One might well ask "*what is the effect of having similar events happening at a slower or faster pace?,*" "*is the rate of urban events related to their overall duration?,*" and "*to what extent events in urban ecosystems differ based on different time frames considered; i.e., what is the effect of temporal scales on urban ecosystems and ecology?*"

Synchrony/Asynchrony, Time Lags and Delays

Urban events can be conceptualized as happening at the same absolute time (i.e., synchronous events) or being separated in time (i.e., asynchronous events). (A)synchrony can affect all urban events regardless of their codependency (e.g., flower-pollinator systems), causation (e.g., urban heat islands and early season flowering) or independency. Perfect synchrony means that the initiation, development and termination of two or more urban events happen at the same time (**Figure 1**). Once historically coupled events are out of synchrony, a temporal mismatch might occur, potentially leading to the disruption of previously coupled events. For example, budburst and flowering are advanced by days to weeks in urban areas compared to surrounding rural areas due to urban areas being warmer (Neil and Wu, 2006). The cascading effects of this asynchrony on plant-animal interactions are largely uncharacterized. In a rare example along these lines, Fisogni et al. (2020) recently observed that urbanization advanced the timing of flowering in France, but not pollinator flight, potentially leading to disrupted pollination in cities. In another example, ecological theory predicts that higher trophic levels may become less synchronized due to the urban heat island effect, disrupting ecosystem services, such as biological pest control. This was observed for a scale insect pest of oak trees and its suite of parasitoid wasps in Raleigh, NC, United States, where urban warming shortened the vulnerable development period for the scale insect, and parasitoid development did not similarly advance (Meineke et al., 2014).

Generally, the potential decoupling of plant relationships with co-evolved species in urban areas (e.g., pollinators, insect herbivores, and soil microorganisms) remains underexplored. Such asynchronies have been investigated in relation to climate

FIGURE 2 | Example of tree planting interventions using trees of different size and age. **(A)** Tree saplings ready to be planted in a residential development in Melbourne, VIC, Australia (Author provided) and **(B)** planting of adult "recycled" palm trees in San Francisco, CA, United States (Courtesy: Bertrand Duperrin, Flicker). Despite streetscapes being redeveloped around the same time, these urban green spaces might have a different vegetation structure due to the different age of the trees planted. Example of an urban forest growth over seven decades. The Tiergarten in Berlin, Germany, in 1945 at the end of WWII **(C)** was used as a large urban farm to sustain the city's starving population, with only a few scattered trees left standing (Courtesy: Landesarchiv Berlin). Today, Tiergarten Park **(D)** is a dense urban forest enjoyed by Berlin's residents for leisure and sport activities with little evidence of its wartime past (Courtesy: Savin A., Wikimedia Commons).

change (Kharouba et al., 2018), but evidence regarding urban change is scant. Similarly, while the effects of pesticide application on urban insects are relatively well-known (Botías et al., 2017), no conclusive evidence exists on how the different timing of chemical (re)application across urban ecosystems might affect insect communities over time. For example, in residential landscapes, timing of application of yard chemicals represents a complex patchwork based on individual choices of residents that, due to limited biological knowledge, might be out-of-sync with targeted phenological phases, or in sync with plant flowering, thus endangering already threatened species.

Time lag analyses on drivers structuring biological communities and their diversity – as regulated by dispersal barriers, trophic interactions, parasitism, life history, etc. – are commonplace in natural ecosystem investigations (Kampichler and van der Jeugd, 2013). In cities, however, these investigations are more limited. For instance, time lags were found when analyzing global urban biodiversity patterns and extinction debts (Hahs et al., 2009), and in South African urban grasslands, du Toit et al. (2016) found that time lags in the compositional change of communities were longer for indigenous woody grassland species than open grassland species. Moreover, evidence on urban time lags and the cascading effects on urban ecosystem function, recovery from stress and perturbations, and resilience is scant. This is an important data gap because evidence from non-urban systems demonstrates ecological time lags can determine the success of ecological management and conservation efforts (Watts et al., 2020). For instance, in road verges, multi-decadal time lags have been identified when investigating community assembly, thus posing new challenges when designing appropriate urban restoration and conservation efforts (Auffret and Lindgren, 2020). More detailed knowledge is needed on the effects of time lags and delays on urban biodiversity and ecological functioning, and how they could underpin effective urban planning and management interventions.

Temporal coupling of urban ecosystems leads to little explored research questions such as "*what happens when interdependent events, that are supposed to occur at the same time, are out of sync?*," "*if the events are out of sync, how far out of sync can they be and still retain the functioning of urban ecosystems?*," "*can asynchrony be compounded and determine cascading effects?*," "*to what extent can time lags and delays change over time affecting structure or functioning of urban ecosystems?*"

Trends, Rates of Event Change and Transitions

In general, urban events may appear static and semi-stationary (i.e., have a null rate of change), particularly if the temporal scale of the event does not align with the temporal scale of observations. The transition between events may be slow and progressive, and thus unnoticeable if only observed during a short window of time during the transition (**Figures 2**, **3**). However, when considering a longer temporal scale, urban properties can noticeably change, revealing clear temporal trends. In other words, urban systems that appear stationary at a particular time scale, might be non-stationary on a different time scale. For instance, neighborhoods across Detroit, MI, United States in the early 20th century appeared stable in population and housing density (**Figure 3**). However, because of a protracted event such as its slow economic decline, these areas saw a progressive downward trend in population and housing density, pushing the urban system into a non-stationary state that ultimately led to a much lower housing and population density, and, concomitantly a higher green space cover today. On the other hand, Joplin, MO, United States saw an abrupt drop in housing and population density because of a tornado that obliterated parts of the city in 2011 (**Figure 3**). While Joplin's urban ecosystem bounced back to a state comparable to that existing prior the tornado disaster, the downward trend in Detroit's status caused the city to reach a new dynamic equilibrium significantly different from the initial one (**Figure 3**). It is worth noting here that some trends can happen at spatial scales much larger than that of a city or a neighborhood and be determined by exogenous forces, such as climate, macro-economic trends, and human drivers, that reach well beyond the single urban area investigated (**Figure 4A**).

Attempts to answer questions related to trends, rates of event change and transitions could employ historical data on urban ecosystems as well as emerging big data (Ossola et al., 2020). For instance, the use of natural history collections – herbaria and animal specimens – could help clarify the effects

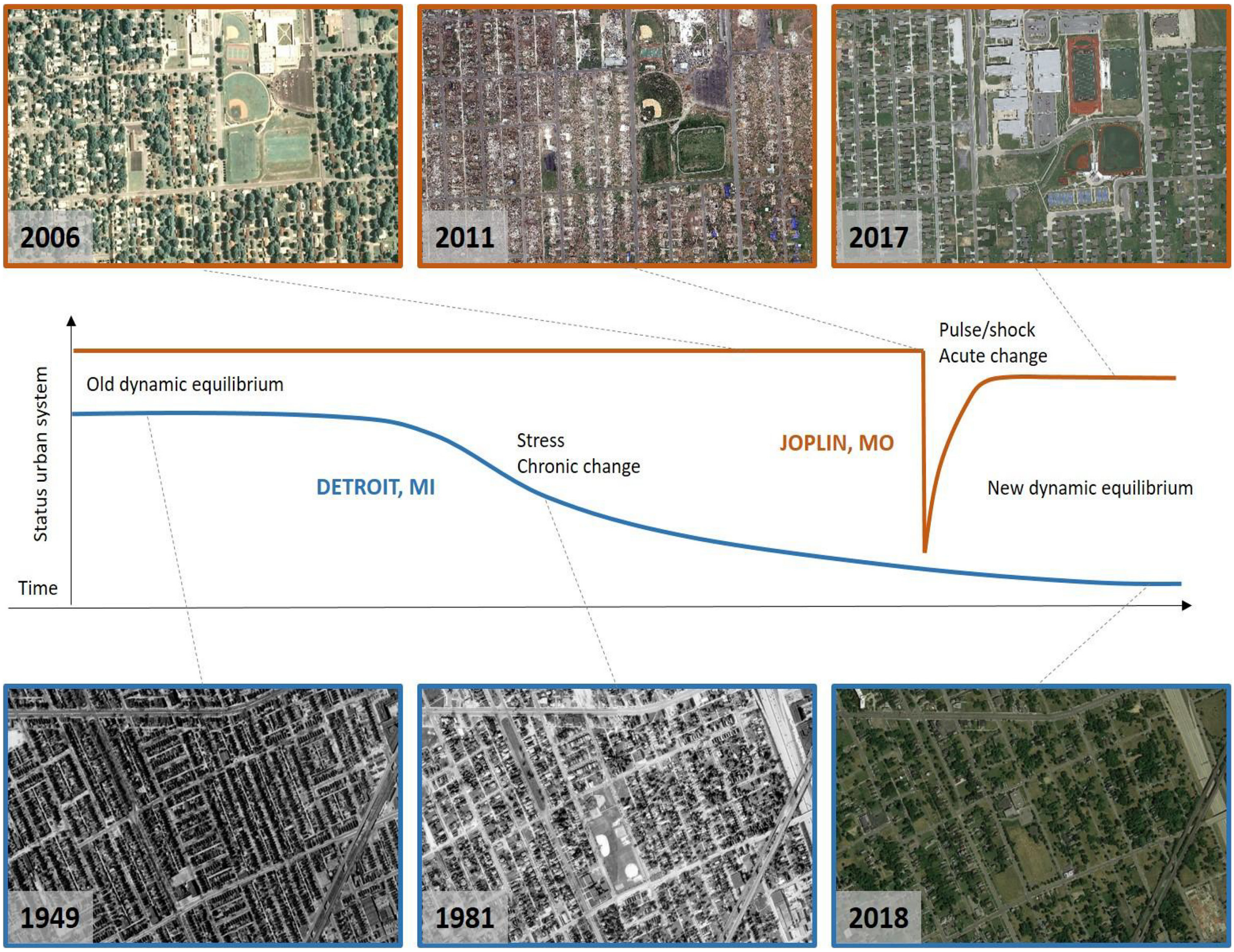


FIGURE 3 | Examples of urban land system change (i.e., trends) following a catastrophic event in Joplin, MO, United States (above orange panel and line), after part of the city was obliterated by a category 5 tornado on May 22nd 2011, and following a chronic and slower change in Detroit, MI, United States caused by the decline of the local automotive industry (below in blue). While Joplin was quickly rebuilt with similar urban form, housing and population density, achieving a new equilibrium similar to that prior to the tornado, Detroit plateaued to a new urban equilibrium significantly different from the original with lower housing and population density, albeit higher green and open space cover. Imagery is courtesy of https://digital.library.wayne.edu (years 1949 and 1981) and www.google.com/earth/ (years, 2006, 2011, 2017, and 2018, accessed on 15 February 2020).

of urbanization on urban biodiversity, its change over time (e.g., species turnover), and the ability of organisms to survive new urban conditions within particular time periods through tolerance, plasticity, and/or adaptation (Shultz et al., 2020), and even evolution of new urban phenotypes and genotypes (Rivkin et al., 2019). Future studies could test whether some organisms may tolerate a relatively slow change to their urban habitat, but not a fast one, likewise some organisms might be able to adjust to the "*press*" of climate trends (i.e., acclimation) but not to the "*pulse*" of climate extremes (i.e., adaptation) (Harris et al., 2018).

Investigating trends in urban systems could be organized by questions like "*what is the effect of having a progressive versus abrupt change to urban ecosystems?,*" and further "*do urban ecosystems return to prior states after a tipping point or become temporarily or permanently shifted to a different state [e.g., legacy lock-ins sensu Ziter et al. (2017)]?*" Ultimately, "*can transitions between events be managed or regulated?*"

Cycles and Hysteresis

Urban cycles capture a series of events that repeatedly lead to comparable results. Hysteresis manifests when two events reoccur following different pathways, thus representing the simplest type of cycle (**Figure 1**). It is important to note that these recurrent points in time might not be the exact manifestation of a particular urban past, but rather the realization of a similar status within a particular urban facet, process or function. For instance, considering the United States real estate market, new residential constructions in the last six decades demonstrated a cyclical pattern. Each year the number of new units fluctuated between 12,000 and 25,000 new units/year. This cyclic pattern

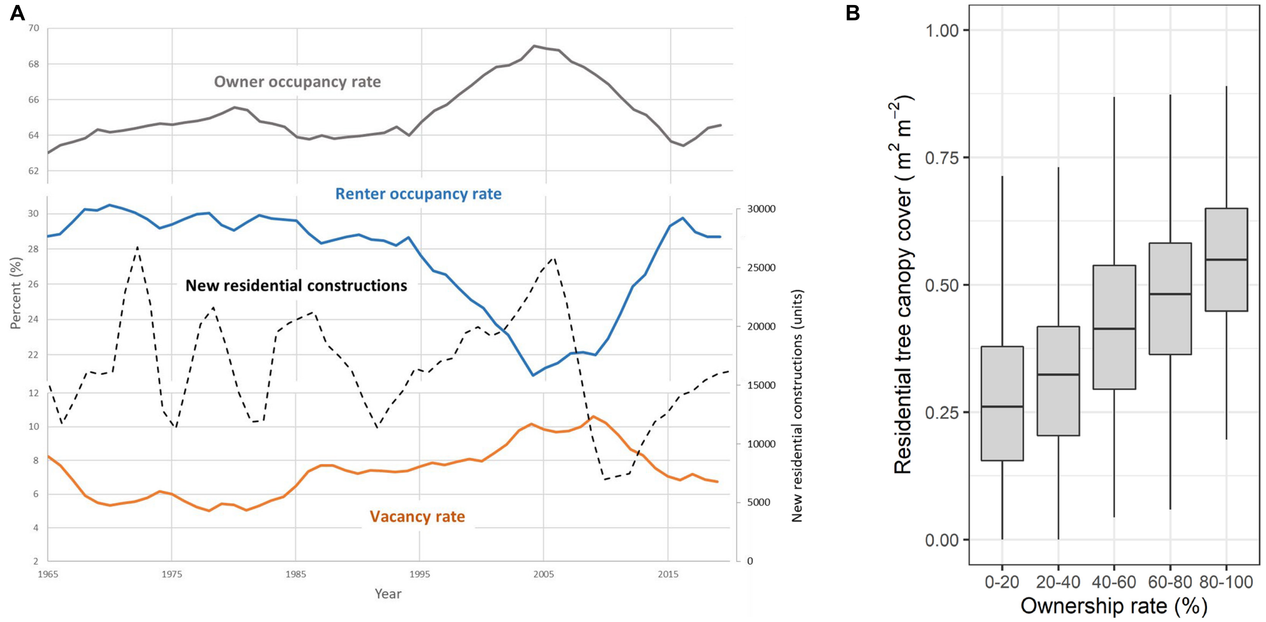


FIGURE 4 | (A) Historical trends (1965–2017) in owner and renter residential occupancy and vacancy rate across the entire United States (in gray, blue, and orange, respectively). Ownership rapidly increased between 1995 and 2005 to then return to the original level in the decade after the global financial crisis in 2007–2008. The trend in new residential constructions in the United States (dashed line), however, had a cyclical regime (drawn with data from https://www.census.gov/econ/currentdata/, accessed on 15 February 2020). **(B)** Relationship between current residential ownership and tree canopy cover measured across 1,709 United States neighborhoods (i.e., census tracts) and calculated based on vegetation data measured in 1,482,206 residential properties in nine cities and 2010 United States Census data (re-drawn from data in Ossola and Hopton, 2018a).

was disrupted by the 2008 financial crisis (**Figure 4A**). The cyclical nature of urbanization trends, land development and real estate markets is a common feature of most urban areas (Riggleman, 1933; Nyström, 1992; Gabrovski and Ortego-Marti, 2019). This reflects macrotrends in dynamics affecting urban land systems (**Figure 4B**) that have been captured in some urban planning theories, such as the "*neighborhood life-cycle theory*" in the first half of the 20th century (Metzger, 2000) and more recently the "*adaptive renewal cycle*" framework (Marcus and Colding, 2014) and the concept of "*rotating land-use*" (Davis, 2019), though their implications and impacts on urban systems are yet to be fully explored. Cyclicity is a recurrent characteristic affecting many facets of urban ecosystems like those related to climate seasonality (e.g., wet/dry seasons), sunlight availability (e.g., day–night year cycle), and recurrence of human actions and disturbance (e.g., bird feeding, mowing, pruning).

Evidence of effects of urban cycles has been documented for some organisms. For instance, altered mowing regimes can shift the composition of plant communities and insect assemblages predictably, such that reduced mowing frequency generally benefits plant and insect diversity (Sehrt et al., 2020; Watson et al., 2020). Human perceptions of outdoors organisms and the urban environment, as measured by people's online search patterns, is strongly affected by seasons (**Figure 5**), as is human perceptions of colors, in mid- to high-latitude cities (Welbourne et al., 2015). This raises opportunities to deepen our knowledge on nature-human interactions and their change over time, seasons and years. Like other temporal dimensions discussed here, it is important to note that cycles in one urban ecosystem might have complex interactions with cycles of other urban systems, often connected through globalized networks of energy and materials, as well as those of natural systems.

Attempts to investigate urban cycles could focus on questions such as "*what happens to an urban ecosystem when an existing cycle is interrupted or a new cycle is formed?*," "*does the number of cyclical events affect properties and functioning of urban ecosystems and to what extent?*," and again "*what is the effect of two looped events happening in a different trajectory (i.e., hysteresis)?*"

Legacies, Priming and Time-Skipping

Temporal legacies are the manifestation of temporal autocorrelation within and across urban ecosystems. Legacies can be thought of as carry-over effects upon subsequent events where urban ecosystems can change while retaining some manifestations of prior events (**Figure 1**). Due to their long-term nature, legacies can often be confounded with urban events having relatively long durations and as such can be identified by examples previously discussed (see section "Time Flow and Duration"). Here, however, we consider as legacies past, fully completed events that affect future consecutive and non-consecutive events (i.e., "time-skipping"). Urban legacies can form within and across urban systems and processes. For instance, planting of a particular species might inhibit other plant species with similar traits from existing in the same system in the future through allelopathic processes (Tabassum et al., 2020).

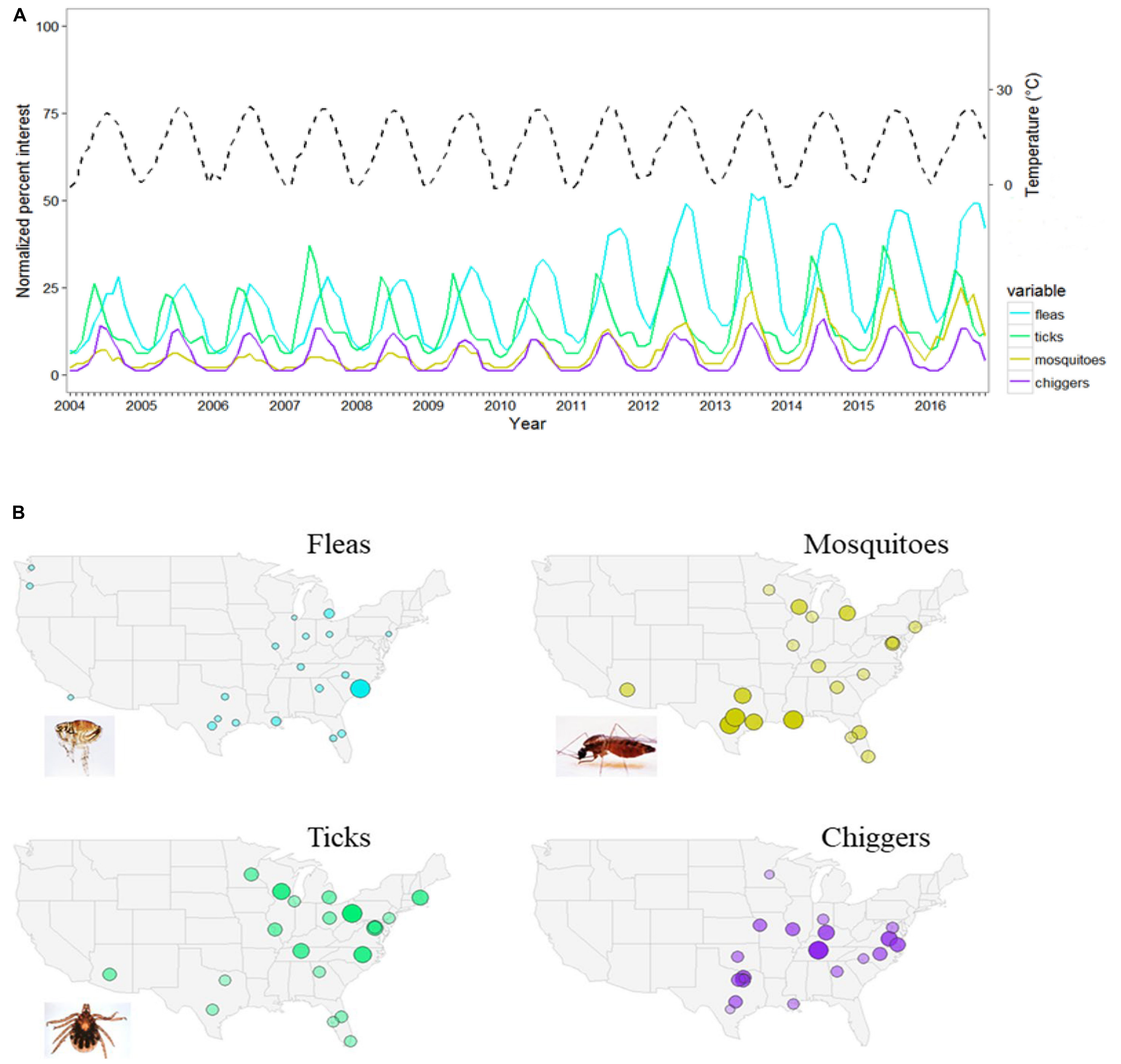


FIGURE 5 | (A) Example of cyclicity in urban ecosystems showing people's online search interest for selected insect groups (solid lines) in relation to seasons (i.e., temperature dotted black line) across the conterminous United States from 2004 to 2016. People's interest is measured through the normalized percent interest of Google searches on the insect name as reported by Google Trends (accessed November 2016). Temperature is the mean monthly temperature for the conterminous United States (accessed from https://www.ncdc.noaa.gov/cag/time-series/us/on November 2016). **(B)** Maps showing the United States cities with the greatest online search interest for each insect group for the 2004–2016 reference period.

In this example, temporal autocorrelation is self-directed within a single ecological system. On the other hand, the selection of tree species to shade urban landscapes might provide tangible benefits to the built environment only years or decades later, thus causing temporal autocorrelation between an ecological and a technical system.

Among the existing evidence on the effects of urban time, legacies are perhaps the most commonly studied though not fully understood or identified. Legacies of former urban land use have been documented by measuring contamination of contemporary soil with pollutants since banned (Nassauer and Raskin, 2014; Clarke et al., 2015), as well as changes in soil physical and chemical properties, and ecosystem services (Raciti et al., 2011; Setälä et al., 2016; Ziter and Turner, 2018). Effects of former urban planning efforts dictate the structure of contemporary urban forests (**Figure 6**), as well the benefits they provide to residents (Boone et al., 2010). Similarly, infrastructure and zoning plans have legacy effects on the urban fabric that can last

FIGURE 6 | Legacy relationship between the historical trend in average residential parcel size (points in blue) measured across 1,503 neighborhoods (i.e., Census blocks) in nine United States cities and average residential vegetation cover (bars in orange) (re-drawn from data in Ossola and Hopton, 2018b). The historical trend in parcel size (points in blue) affects the current amount of vegetated cover that is achieved in residential parcels (i.e., the legacy in light orange). The decade of maximum housing development of a neighborhood is the decade when most buildings were built. Error bars are the respective standard errors of the mean.

for centuries (Ahlfeldt and Wendland, 2013; Twinam, 2018). Legacies related to former "referential housing types," aesthetics and architectural styles (Ledent and Komossa, 2019) can further determine how contemporary urban green spaces and landscapes are structured and function (Ossola et al., 2019).

Complex urban legacies that arise from both the ecological system and the coupled social and technological system and their interactions deserve closer investigation. For example, human factors can influence horticulture and landscaping resulting in preferences of the past influencing current vegetation composition and structure (Ossola et al., 2019; Avolio et al., 2020). In addition, the spatial spread of tree diseases such as Dutch Elm disease through a population of urban trees is influences by the spatial distribution of that species. Future species distributions of trees will, in turn, by influenced by the management choices made in an effort to deal with the spreading disease. Heterogeneity at one time period – distribution of the elm trees – becomes a driver of the heterogeneity in another time period – distribution of diseased trees. The distribution of diseased trees is an outcome but it, in turn, becomes the driver for the heterogeneity in management response. This process has been called dynamic heterogeneity and is a spiral of complex social-ecological interactions (Pickett et al., 2017).

An understudied temporal dimension of urban systems relates to "priming effects" – whereby the occurrence of one event, rather than an alternative one, can lead to a significantly different future status of an urban system (**Figure 1**). For instance, little is known about how prior colonization or planting of particular species might affect future biotic communities. Such "*priority effects*" (Fukami, 2015) have the potential to shape urban communities and have been found in some recent studies. For example, Aloisio et al. (2019) found that established plant communities on green roofs strongly affected which species were able to survive on these same green roofs at a later time. In addition, Johnson et al. (2015) found that after building removal, plant communities within the footprint of the original building diverged compared to those in the yard areas of parcels. If we can learn to facilitate founder species that tend to promote the establishment, survival and success of other desirable species, such as natives and other organisms that provide targeted ecosystem services, we might be able to harness more benefits from urban nature-based solutions (Dallimer et al., 2015).

Several questions arise when trying to better understand the effects of the past on contemporary cities: "*how long do urban legacies last?,*" "*are urban events more* important *if they create long-lasting legacies?,*" "*what are the effects of interactive legacies in* *urban ecosystems?,*" "*what happens when legacies are not manifest in all consecutive events along a time series (i.e., skipping)?,*" or "*would the change of a foundational event determine a significantly different series of events (i.e., priming)?*"

Temporal Hotspots and Hot Moments

When looking at temporal dynamics in relation to the spatial dimension of urban ecosystems, we can further conceptualize places and periods with different temporality (i.e., the unfolding of events and not the absolute time) as compared to nearby locations and periods; *temporal hotspots* and *hot moments*, respectively. These concepts arise from the recognition that cities are intrinsically heterogeneous yet dynamic (Cadenasso et al., 2006) and they have been recently reframed in the context of "*control points*" within rural ecosystems (Bernhardt et al., 2017). A cluster of residential parcels that changed little over the last century could sit next to a group of parcels redeveloped multiple times over the same period. Thus, a temporal hotspot can be defined when urban events happen at a different rate at a location compared to similar events at nearby locations. In contrast, a hot moment refers to when urban events occur more frequently within a particular time period compared to another time period. For instance, urban tree removal is a ubiquitous event across urban landscapes (Ossola and Hopton, 2018b), but its rate could significantly increase from a baseline level after storms and hurricanes (i.e., hot moment), as well as increase when moving from unaffected areas toward those in the midst of wind gusts and tornado paths (i.e., temporal hotspot). Similarly, prolonged illumination at night under a tree can be considered a *hot moment* as it can locally extend the duration of daily photosynthesis in trees around artificial light sources. Considering the entirety of an urban forest, trees around light poles could also be identified as *temporal hotspots* whereby recurrent light pollution determines longer growth periods and seasonal phenology (Ffrench-Constant et al., 2016; **Figure 7**).

The recent COVID-19 pandemic can be conceptualized as a hot moment in itself, whereby sudden curfews, lockdowns, and other restrictions to people's movements likely released anthropogenic pressures on several organisms causing many of them to quickly (re)occupy many urban areas in various regions globally (Bates et al., 2020). While global hot moments like this might be relatively rare, they might have important implications on how cities are envisioned, used and planned for the future. Thus, future studies could further investigate the effects of more localized, city-scale, hot moments, and temporal hotspots, such as those related to climate change (i.e., fires, heatwave, flood, and drought events). In particular, extreme heat events are expected to become more common in the future as a result of climate change and urban heat island effects will exaggerate these events (Li and Bou-Zeid, 2013). The urban heat island effect is more prominent in certain parts of cities due to higher concentrations of impervious surfaces and vegetation removal, such that extreme urban hot spots may arise during heatwaves, with uncertain localized implications for human health, urban environment and biodiversity (Ossola et al., 2021).

This type of thinking could help address questions like: "*can temporal hotspots and hot moments be regulated to promote functions and services or mitigate unwanted effects?*" and "*are there specific locations and periods for events to occur so that urban functions are optimized?*" By attempting to link temporal and spatial dimensions one might ask "*to what extent mapping the development of urban events across landscapes and periods can help us better understand and measure urban ecosystems and their change?*"

Stochasticity, Determinism and Future Casting

The temporal dimensions analyzed in previous sections focus on past and current urban events. These past and current events can serve as a benchmark for future urban events as well as a baseline to measure urban scenarios against, i.e., a rich suite of possible urban futures that can be envisioned and planned for Iwaniec et al. (2020). These events can be used when designing and implementing steps to reach desirable urban futures through backcasting (Bibri, 2018). Urban futures can themselves sometime "*be described in terms of their duration – the time, in other words, in which they are expected to exist, last or persist before yielding to the present or to other futures*" (Davis, 2019). For instance, the development of a new neighborhood in the future might well have a defined start and end date, and duration overall.

Future urban events can be further considered along a *deterministic-stochastic gradient*. Fully deterministic events can be predicted as they are less affected by temporal randomness, at least within the time scale considered (e.g., the recurrence of summertime in the next 50 years, **Figure 5**). For instance, street tree diversity in the United States changed little in the last 40 years (Ma et al., 2020); as such, without a significant and widespread change in nursery production, plant selection or an unanticipated climate disaster, it is reasonable to anticipate that street tree diversity will remain constant in the next decade or so. On the other hand, stochastic events are random by definition, and as such, it is difficult if not impossible to predict their future occurrence, development, and duration (e.g., a tornado destroying a city, **Figure 3**).

Stochasticity has been considered in some ecological studies looking for instance at biological control of invasive species in the wild (Evans et al., 2012) and community ecology (Ning et al., 2019; Shoemaker et al., 2020). To date, however, little effort has been placed toward evaluating how stochastic vs. deterministic processes in cities can affect biological communities and urban ecology (Caruso et al., 2017; Goddard et al., 2021). Recent research shows, however, that specialist insect herbivore populations in cities might be governed more by stochastic events rather than deterministic drivers (Herrmann et al., 2012). In contrast to urban biodiversity research, modeling approaches moved toward acknowledging the importance of stochasticity for urban technological structures (Ellam et al., 2018), stormwater quality (Obropta and Kardos, 2007), urban population growth and morphogenesis (Raimbault, 2018) as a way to improve our ability to make predictions of future urban events and properties.

More research effort could be placed toward looking back at the urban past to test and validate current urban theories and

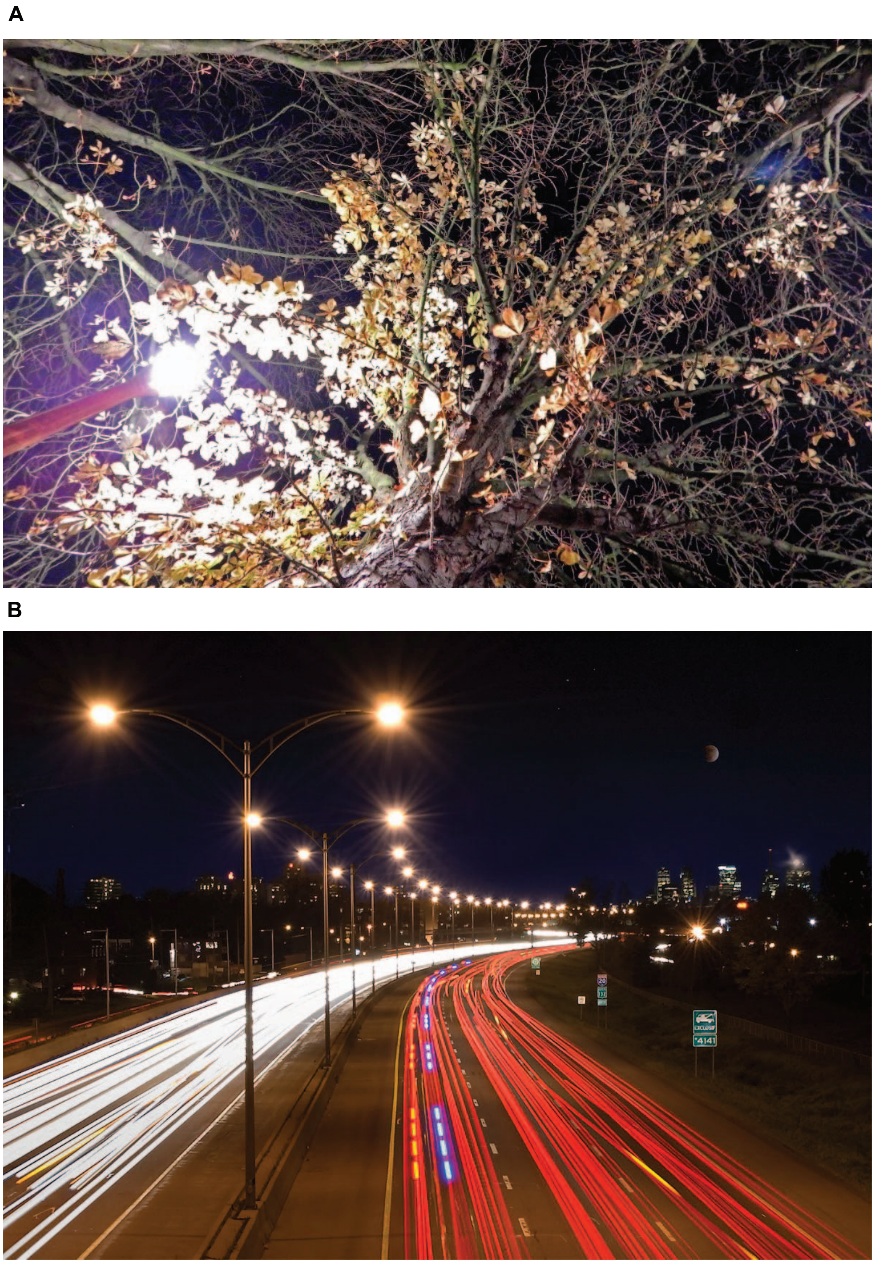


FIGURE 7 | (A) Localized light pollution affecting the urban forest can be thought as a "temporal hotspot" whereby the longer duration of artificial illumination can determine longer plant growth periods and extended seasonal phenology. The image details delayed leaf drop in the most illuminated side of a canopy of *Aesculus hippocastanum* in a public park in Lille, France, in December 2016 (Courtesy: Lamiot, Wikimedia Commons). **(B)** Peak nighttime traffic can be thought as a "hot moment," whereby the higher frequency and intensity of car illumination at determined periods can potentially affect light-sensitive organisms such as insects, plants, and wildlife. The image details car light trails on a highway in Montreal, QC, Canada (Courtesy: Pierre Vignau, Wikimedia Commons).

models, and how these can be projected and extrapolated into the future (**Figures 2C,D**). Particularly promising are studies based on "*time-for-place*" surrogates that could be used to better infer future properties of urban and non-urban systems. For instance, contemporary cities produce microclimatic gradients and conditions similar to those predicted under global and climate change, such as heatwaves and droughts, thus opening up unique experimental conditions to test how organisms, communities and processes might work in the future (Lahr et al., 2018). However, whether urban areas can serve as analogs for rural areas in the future, and for what biological processes [e.g., Youngsteadt et al. (2015); Wohlfahrt et al. (2019)] remains an important area for research.

When considering urban futures, numerous research questions could be framed: "*To what extent can the unfolding of future events be predicted based on current and past ones?,*" "*how can current and past urban events inform the development of scenarios and future-casting efforts*?," or again "*what is the relationship between stochastic and deterministic events and their interplay in affecting urban futures?*"

CONCLUSION

In this contribution we outlined the main dimensions of urban time that can shape how events develop in urban ecosystems. Among these dimension, those related to temporal hotspots/moments and stochastic vs. deterministic processes – are understudied. We believe that – by better valuing urban time – urban ecology can achieve a richer, more nuanced and complete understanding of cities and towns through investigations that utilize the theoretical lens discussed here.

Future research is needed not only to elucidate how these dimensions affects urban ecosystems, but also how they interact with each other, the interplay with spatial dimensions of cities across scales, as well as the connections with other urban and non-urban systems. We believe that a comprehensive evaluation of how the "*Urban Chronos*" affects social, ecological, and technical systems is now needed to significantly advance urban ecology and ultimately its practice and societal impact.

AUTHOR CONTRIBUTIONS

AO initiated and led the manuscript with contributions to writing and editing by EM and MC. All authors contributed to the article and approved the submitted version.

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The Benefits and Limits of Urban Tree Planting for Environmental and Human Health

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Many of the world's major cities have implemented tree planting programs based on assumed environmental and social benefits of urban forests. Recent studies have increasingly tested these assumptions and provide empirical evidence for the contributions of tree planting programs, as well as their feasibility and limits, for solving or mitigating urban environmental and social issues. We propose that current evidence supports local cooling, stormwater absorption, and health benefits of urban trees for local residents. However, the potential for urban trees to appreciably mitigate greenhouse gas emissions and air pollution over a wide array of sites and environmental conditions is limited. Consequently, urban trees appear to be more promising for climate and pollution *adaptation* strategies than mitigation strategies. In large part, this is due to space constraints limiting the extent of urban tree canopies relative to the current magnitude of emissions. The most promising environmental and health impacts of urban trees are those that can be realized with well-stewarded tree planting and localized design interventions at site to municipal scales. Tree planting at these scales has documented benefits on local climate and health, which can be maximized through targeted site design followed by monitoring, adaptive management, and studies of long-term eco-evolutionary dynamics.

Keywords: urban ecology, forestry, sustainability, policy, climate mitigation, climate adaptation, ecosystem services, ecosystem disservices

INTRODUCTION

Urban trees in parks, yards, streets, and remnant parcels have been features of urban design and landscape architecture for centuries (Arnold, 1980), and are still integral components of civic spaces that are well-recognized for their public value. Urban trees are purported to have a number of environmental benefits, such as pollution absorption (Nowak et al., 2006, 2018), stormwater mitigation (Bartens et al., 2009), atmospheric cooling (Shashua-Bar and Hoffman, 2000), reduced energy use (Akbari et al., 1997; Akbari, 2002; Donovan and Butry, 2009; Hsieh et al., 2018), and habitat provision (Burghardt et al., 2009). In addition, studies have indicated that proximity to urban vegetation may increase property values (Sander et al., 2010; Li and Saphores, 2012; Escobedo et al., 2015), facilitate recuperation after stress and illness (Ulrich, 1983; Ulrich et al., 1991; Li and Sullivan, 2016), and reduce mental fatigue (Houlden et al., 2018). There is a substantial literature indicating that trees provide benefits for municipalities and their residents, and this perception, in part, has motivated local, regional and global initiatives that promote the planting of urban trees (McDonald et al., 2016).

However, there is also an increasing empirical understanding of the limits of tree planting as a nature-based solution to climate change and pollution. Urban forest dynamics, species composition, soil dynamics, and the costs of planting and managing designed spaces are important variables in urban forest outcomes, and must inform urban planting practices for successful planning and management (Oldfield et al., 2013). In addition, recent empirical studies highlight that spatial and temporal scales heavily influence the extent of environmental and social impacts of urban trees. Some of the purported benefits of urban forests require that trees are planted on large spatial scales beyond municipal boundaries, and maintained over the long term, to ensure effectiveness (Salmond et al., 2016). However, trees may provide other ecosystem services even when planting is constrained by the relatively limited spaces in dense cities. In this perspective, we refer to *mitigation* as strategies that aim to reduce climate change and pollution, and *adaptation* as strategies that aim to modify cities to help residents cope with climate change and pollution (Laukkonen et al., 2009). Many environmental benefits attributed to urban trees fall into one of these two categories. In addition, we discuss the evidence pertaining to both positive and negative impacts of trees on human health in the context of the extent of urban tree planting.

URBAN TREES ARE MORE EFFECTIVE FOR ADAPTATION THAN MITIGATION STRATEGIES

Early studies of urban tree ecosystem services emphasized rates of carbon (C) sequestration and air pollution reduction (Nowak and Crane, 2000; McPherson et al., 2005; Ray, 2005). Tools such as iTree[1] translated the relatively scant data on urban tree processes available at the time—primarily estimates of biomass and dry deposition rates—into easily understood, municipal-scale metrics, such as tons of pollution absorbed (Tallis et al., 2011), energy savings (McPherson, 1993), C sequestered (Nowak, 1993), and total monetary value (McPherson, 1992; Nowak et al., 2002). However, since these tools were originally developed, additional empirical studies of the influence of trees on pollution concentrations have reported negligible or inconsistent impacts (Setälä et al., 2013; Han et al., 2020), or even increases in the residence time of particulate and NO_2 concentrations in the atmosphere in the presence of tree canopies (Tong et al., 2015; Viippola et al., 2018). When atmospheric mixing is low, pollutants may be concentrated under tree canopies (Salmond et al., 2013), and when atmospheric mixing is high, studies have shown no discernable effect of the presence of trees on urban pollutant concentrations (**Figure 1**). In addition, trees may exacerbate rates of asthma due to the release of allergens and this is seldom accounted for in assessments of the impacts on trees on public health (Lovasi et al., 2013). In recent comprehensive reviews, Eisenman et al. (2019) and Xing and Brimblecombe (2020) both concluded that as a result of the many influences of trees on atmospheric composition besides dry deposition rates, current empirical evidence does not support the assumption that trees significantly and consistently reduce pollution concentrations.

Similarly, for C sequestration, tree inventories coupled with allometric equations have been commonly used to estimate CO_2 stored in urban trees in units of mass (Nowak and Crane, 2002; Nowak et al., 2013). However, to assess the potential of trees to enable climate mitigation, a systems-level approach is needed to compare C sequestration in urban forests to local fossil fuel emissions (Hutyra et al., 2014). Most studies of urban forest C sequestration attempt to estimate either tree biomass (McHale et al., 2007; Hutyra et al., 2011; Strohbach and Haase, 2012; Timilsina et al., 2014) or the rate of change in biomass over time (Net Primary Production, NPP) (Bialecki et al., 2018; Sonti et al., 2019; Trlica et al., 2020). Soil C dynamics should also be assessed as part of urban C balance, as soil C may either contribute to sequestration if organic matter is accumulating, or release C to the atmosphere if heterotrophic respiration reduces organic matter concentrations (Pouyat et al., 2002, 2006; Decina et al., 2016). But in either case, the spatial extent of urban trees and soils is quite limited relative to the magnitude of fossil fuel emissions. Cities are highly heterotrophic and expend orders of magnitude more C than they fix in photosynthesis (Collins et al., 2000). In most modern cities, fossil fuel combustion exceeds NPP per unit land area by at least an order of magnitude (Pataki et al., 2011). Hence, urban tree growth typically offsets municipal C emissions by only 0–3% annually (Pataki et al., 2009; Escobedo et al., 2010; Liu and Li, 2012; Baró et al., 2015; Velasco et al., 2016; Lindén et al., 2020), even before accounting for the energy needed to produce, transport, irrigate, prune, and fertilize urban trees (Roy et al., 2012).

This is fundamentally a problem of scale. Pollution and greenhouse gas (GHG) emissions in modern cities are disproportionately large relative to the extent of urban trees. Globally, forests are an important contributor to the C cycle because they occupy about a third of the land surface (FAO and UNEP, 2020). However, cities occupy less than 1% of the global land surface (Zhou et al., 2015; Liu et al., 2018), and within cities tree cover is highly variable but seldom equivalent to closed canopy forests (Nowak and Greenfield, 2018). Consequently, urban trees are most effective at providing ecosystem services that operate at local scales, such as parcels or urban forest/neighborhood patches.

For example, trees may improve human thermal comfort locally both through evaporatively cooling and humidifying

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[1] https://www.itreetools.org/

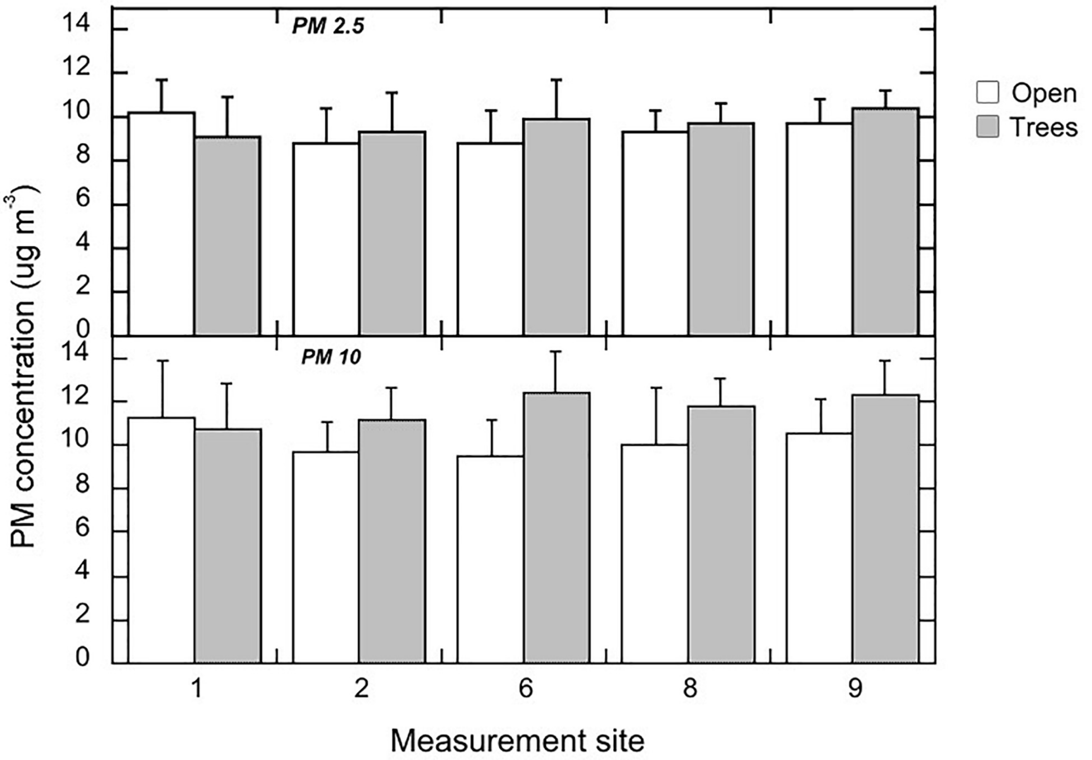

FIGURE 1 | In Lahti, Finland, researchers found that atmospheric particle deposition (the mass of particles deposited on the sampler) was significantly lower at urban sites with trees than in open areas. However, this difference did not result in significant differences in atmospheric PM 2.5 or PM 10 concentrations (particulate matter <2.5 and <10 μm in diameter) at any of their measurement sites, as shown above. This distinction is important because while many studies have inferred that trees will improve human health based on the influence of trees on particle deposition, it is atmospheric concentrations that actually impact human health directly. Error bars show SD; data from Setälä et al. (2013).

urban air, and through shading, i.e., the interception of radiation by large canopies (Shashua-Bar et al., 2011; Chen and Ng, 2012; Rupp et al., 2015; Middel et al., 2016; Wang et al., 2018; Zhao et al., 2018). Wang et al. (2018) utilized the Weather Research and Forecasting Model (WRF) and found that all urban trees in the contiguous United States lower air temperature by about 3°C across urban areas compared to scenarios in which cities contained no trees at all. This is likely an upper limit on municipal-scale cooling effects, as this study evaluated the impact of all existing urban trees. Salmond et al. (2016) pointed out that most studies of urban forest-climate interactions are regional scale modeling that use WRF or other simulation tools, and there is generally less information about cooling effects from empirical measurement at local scales. Nevertheless, cooling has been found in empirical studies as a function of canopy cover. Jung et al. (2021) showed that the capacity of urban trees to mitigate land surface temperatures is non-linear and depends on the specific land cover type and level of development. Santamouris et al. (2017) found that across a wide range of studies, the median reduction in local air temperature by cooling interventions involving urban trees was as high as 1.5°C when including modeling scenarios that largely eliminated built cover, and 0.6°C across more realistic urban conditions. By intensively sampling urban air temperature across land cover gradients, Ziter et al. (2019) estimated urban cooling of up 2°C at spatial scales of 60–90 m in radius at very high canopy cover close to 100%.

Therefore, localized cooling with dense tree planting offers the potential for microclimate adaptation through highly targeted design interventions that focus on parks, bus stops, pathways, school yards, community centers and other pedestrian urban gathering spaces that significantly impact thermal comfort. In addition, the reduction of stormwater runoff from impervious surfaces may also facilitate climate change adaptation. Canopy interception, i.e., rainfall that is intercepted by the tree canopy and evaporates from the leaves, may constitute a significant fraction of water inputs from low intensity, short duration rainfall events, depending on tree species (Xiao et al., 1998; Xiao and McPherson, 2002; Asadian and Weiler, 2009; Nytch et al., 2019). Urban greenspaces typified by pervious, non-sealed soils can also absorb and store considerable amounts of rainwater in the soil. For example, urban catchments in two Finnish cities with 30–40% of permeable soils (parks and other green infrastructure) stored more than 1,000 m^3 of rain water per ha per year, measured as the difference between the amount of water entering the catchment and discharge. The result was a substantial reduction in urban runoff volume and considerable improvement of the quality of runoff water (Valtanen et al., 2014a,b). In these cities, about 70–80% of rainfall occurs in low intensity (<10 mm) events, which are the conditions under which adding pervious areas of soil to cities is impactful for stormwater mitigation. Consequently, urban forests and other greenspaces with trees and permeable soils can help ameliorate the modified urban microclimate and hydrological cycle and mitigate heat and flooding locally.

THE SPATIOTEMPORAL DYNAMICS OF URBAN FORESTS

Cities are limited in their extent globally. Space allocated to urban trees is also constrained. Urban spatial configurations that have been designed for dense human populations, movement, and social interactions leave limited options for accommodating biophysical processes like water uptake and evapotranspiration, nutrient uptake, wet and dry deposition, gas exchange, and C sequestration at the scale required to offset urban pollution. Hence cities are difficult to retrofit to accommodate greening strategies even as we discover their specific value. For example, it would be quite difficult, if not impossible, to reduce stormwater runoff by 50% using green infrastructure in a city that is 90% impervious, particularly if we can only allocate 1–2% of the land surface to greening strategies.

The spatial scale and distribution of urban greenspace reflects a range of socioeconomic dynamics and historical planning decisions made by city governments (Grove et al., 2014; Locke et al., 2020). Recent studies investigating the socioeconomic determinants of urban vegetation have predominantly shown a correlation between wealth and both vegetation cover and diversity, although this relationship is far from universal and may depend on other factors (Kendal et al., 2012; Szantoi et al., 2012; Schwarz et al., 2015; Watkins et al., 2017; Kuras et al., 2020). Trees may also present liabilities like tree falls, root damage to infrastructure, pollen allergies, and maintenance concerns. These disservices will necessarily constrain the location of trees in the built environment (Pataki, 2021). Temporal dynamics, including tree demographics, host-pathogen interactions, extinctions, and other population and evolutionary processes also influence the extent to which tree planting efforts influence urban conditions. A multitude of both anthropogenic and biophysical factors influence tree planting rates, removal, and mortality at the landscape scape, and some cities and neighborhoods are on trajectories of decreasing tree density and canopy cover (Roman et al., 2014; Ossola and Hopton, 2018; Hilbert et al., 2019). Low species diversity, poor site conditions, and planting palettes that are no longer suited to changing climatic conditions and pathogens present additional vulnerabilities for urban tree populations (Laćan and McBride, 2008; Berland and Elliott, 2014; Ordóñez and Duinker, 2014). There is increasing evidence of urban-driven evolutionary change in urban forest dynamics that interacts with ecosystem function over the long term (Johnson and Munshi-South, 2017). For example, evolutionary changes in traits that influence C cycling such as plant root traits, stomatal conductance, leaf nutrient stoichiometry, and soil microbial C cycling traits, have been shown to have significant implications for C sequestration (Monroe et al., 2018). While evolutionary processes may operate at long time scales for long-lived organisms such as trees, they have been shown to be very rapid for the microbial communities that influence plant fitness (Lau and Lennon, 2012). Both demographic and evolutionary processes are highly complex in urban forests that contain varying proportions of planted and naturally dispersed and regenerated trees (Nowak, 2012). Therefore, urban greening strategies must account for both spatial and temporal dynamics of planted urban trees and urban forest patches in defining planting strategies and management targets.

URBAN FORESTS AND HUMAN HEALTH AT THE SCALE OF URBAN TREE PLANTING

Maintaining urban tree populations at small spatial scales appears to significantly influence human health. Recent reviews have utilized the World Health Organization's definition of health to evaluate the effects of either trees specifically, or greenspace more generally, on physical, mental, and social well-being (Nesbitt et al., 2017; van den Bosch and Ode Sang, 2017; Kondo et al., 2018; Wolf et al., 2020). Several aspects of physical health have been shown to be correlated with aspects of urban "greenery," such as mortality, longevity, and heart rates, and weight changes (Nesbitt et al., 2017; Kondo et al., 2018). There are also numerous studies relating aspects of mental health to the prevalence of vegetation (Bratman et al., 2012; Nesbitt et al., 2017; Houlden et al., 2018). Notably, some studies distinguish the effects of trees, or species of trees, from other vegetation and some do not (Nesbitt et al., 2017). As a result, the precise mechanisms linking trees, biodiversity, and the different components of health: physical, mental, and social, remain uncertain (Lee and Maheswaran, 2010; Aerts et al., 2018). However, mechanistic responses of human health to actual and perceived biodiversity have been generally categorized as those that cause or reduce harm, and restore or build capacity for physical and mental health (Marselle et al., 2021). Research has shown that the amount of greenspace, as well as the distance urban dwellers have to travel to that greenspace, can influence the benefits of trees on public health (Annerstedt van den Bosch et al., 2016), but few studies have attempted to understand the type and amount of exposure to trees that confers health benefits (Shanahan et al., 2015; Jiang et al., 2016; Zhang et al., 2017). Finally, the relationship between the prevalence and types of urban trees and social health remains largely unexplored (Dinnie et al., 2013; Nesbitt et al., 2017; Jennings and Bamkole, 2019), but may be impactful at small spatial scales and through particular configurations, such as community gardens and the associated place attachment (Petrovic et al., 2019).

The health effects of trees may be a critically important aspect of urban forest benefits, even in dense cities with relatively limited space to support urban forests. This is because it is very possible that exposure to small plantings, parks, and views of urban forests may be effective in improving human health. That is, unlike pollution mitigation, small-scale plantings may have large impacts on health. Though some urban residents avoid large wooded areas due to safety concerns (Klein and Felson, 2021), many make varied use of small greenspaces (Peschardt et al., 2012). Indeed, some urban residents are highly deprived of virtually any access to nature, such that even modest additions have been shown to have measurable positive effects (Nadkarni et al., 2017). These social and public health functions of small greenspaces have been further highlighted by the COVID-19 pandemic, which in many cities restricted recreation to small local parks (Kleinschroth and Kowarik, 2020). Similar to the call for "dose-response" relationships between exposure to nature and human health (Jiang et al., 2016), there is a great need for additional quantitative and qualitative studies of the health impacts of urban tree plantings at different spatial and temporal scales, applicable to the realistic conditions and constraints of adding and maintaining trees in dense cities.

DISCUSSION

Going forward, a central question across many disciplines and stakeholders is: “What types of urban spaces promote social, economic, and environmental sustainability and prosperity?” As cities struggle to address myriad social, economic, and environmental problems, it is important to identify the specific social and environmental goals that can be achieved by the location, density, and extent of tree planting. Hence, the spatiotemporal dimensions of urban forest dynamics are a critical research uncertainty. For some ecosystem services, such as C sequestration, the ecological dynamics of C uptake by trees are well understood, but the direct impacts on atmospheric CO_2 concentrations and climate depend on the spatial extent of urban trees, their demographic and population dynamics, and the interactions between productivity, heterotrophic respiration, and soil C dynamics at large spatial and temporal scales. Conversely, the reciprocal impacts of the atmosphere, including urban pollution and heat, on urban forest ecological processes are also in need of additional empirical measurements (Meineke et al., 2016). For other ecosystems services, the interactions between tree and soil processes and the built environment will determine the net influence of urban forests. For example, the effects of buildings, street design, and urban morphology on atmospheric dispersion, and their interactions with tree canopies, play a significant role in atmospheric pollutant concentrations (Han et al., 2020). Integrating trees into the built environment

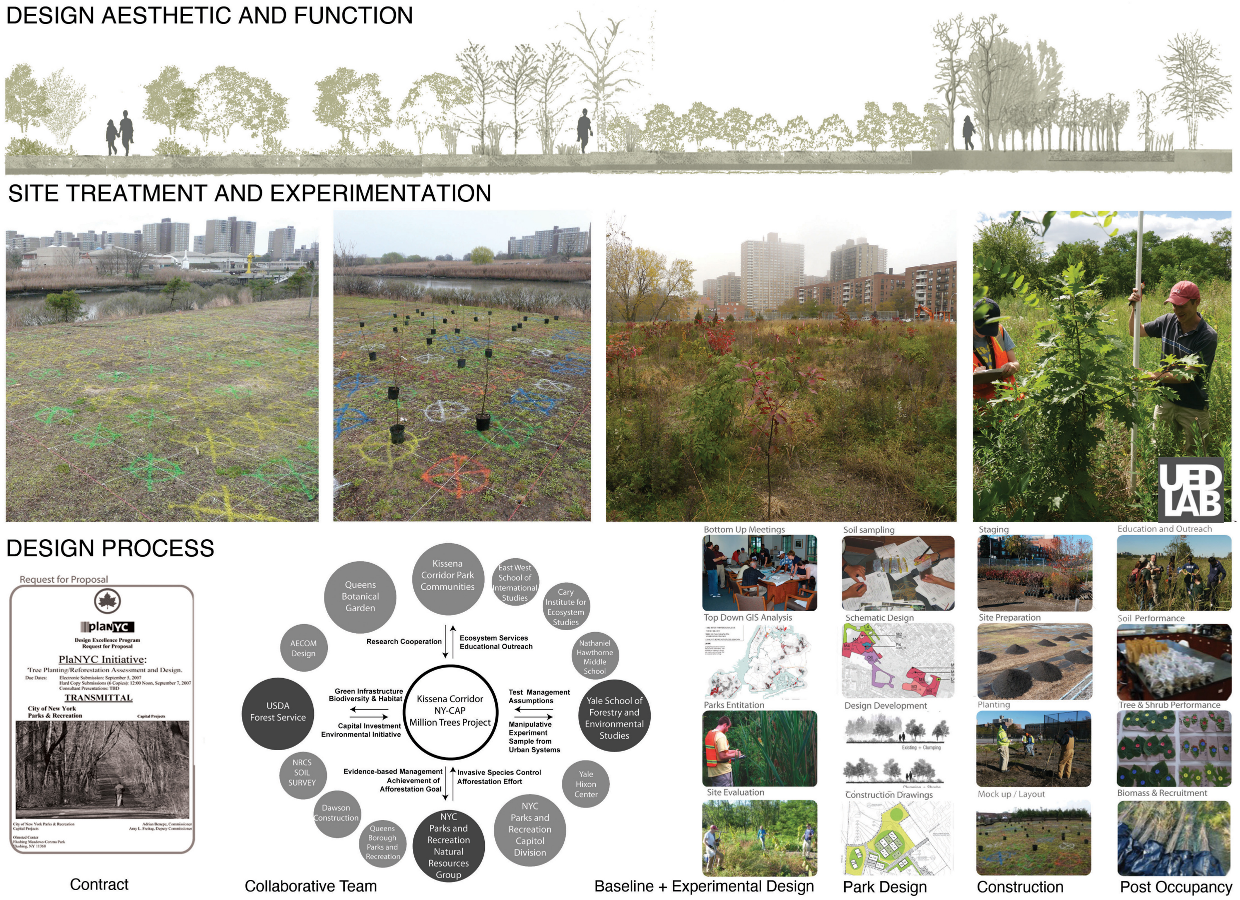

FIGURE 2 | The MillionTrees NYC Initiative uses the model of designed experiments to engage ecologists and designers in the creation of a long term and large-scale urban green infrastructure research program. Adapted from Felson et al. (2013a,b).

may also facilitate interventions to increase active transportation (Tsai et al., 2019; Young et al., 2020). Overall, we conclude that is difficult to significantly offset urban GHG and atmospheric pollution with localized tree planting, given the magnitude of emissions in modern cities. However, trees have localized effects on climate, thermal comfort, human health, and habitat for other species that may be impactful at the site scale.

The public health impacts of urban trees have been particularly difficult to characterize because collaborative, interdisciplinary approaches are needed to ascertain the nature of these impacts. Assessing the effects of planting interventions may require techniques such as virtual and/or real life walks or progressions through different types and scales of landscape designs (Berman et al., 2008). Browning et al. (2020) suggested that a lack of randomized treatments and experimental controls has hampered our understanding of the influence of natural landscapes on cognitive and mental health. There have also been disciplinary gaps and barriers between urban ecological and forestry studies and health scientists that continue to limit progress in linking trees and health (Eisenman et al., 2019). The next phase of planning and managing tree planting programs will require collaborative teams of natural scientists, social scientists, and practitioners from the health sciences including epidemiologists, ecopsychologists, and clinicians to evaluate the specific dimensions—including forest composition, biodiversity, soil health, and spatiotemporal dynamics—that interact with human health.

The study of socioecological interactions in cities provides a framework for generating and organizing place- and site-specific data across the many disciplines involved in planning and designing urban spaces. Furthermore, the growing field of ecological design and urban-focused landscape ecology provides hands-on approaches to planning and managing urban flora and fauna (Beck, 2013; Felson et al., 2013a, **Figure 2**). The COVID-19 pandemic has provided an unparalleled opportunity to re-configure urban landscapes in ways that integrate trees, forest patches, and green corridors into the built environment with evidence-based ecological designs. There may be unique opportunities to expand tree canopies in abandoned or re-zoned urban lands if remote work becomes commonplace on a permanent basis, causing redistributions of commercial and/or residential land uses (Boyd, 2020; Eltarabily and Elghezanwy, 2020; Ferrini and Gori, 2021). There are previous examples of "urban shrinkage" that resulted in reconceptualized urban greenspaces, open space networks, or forest regrowth (Kowarik and Körner, 2005; Nassauer and Raskin, 2014; Frazier and Bagchi-Sen, 2015; Haase et al., 2018). Where this is feasible, cost effective relative to other land use priorities, and implemented with resources to maintain tree planting over decadal timescales, extensive urban forests have observable environmental and social benefits. However, cities have competing demands for space that include pressing human needs for affordable housing, renewable energy generation, and food production, among other uses. Nevertheless, we suggest that even small-scale and temporary tree planting may have specific benefits. To maximize these benefits, it is essential to establish collaborative teams working through the design process to influence the direction of the built environment (Felson et al., 2013b). Focusing on comprehensive and phased planning alongside targeted site design and monitoring for the specific attributes of trees that contribute to climate adaptation and human health may be most effective for integrating urban forests into sustainability strategies.

AUTHOR CONTRIBUTIONS

DP conceived of the manuscript and organized the text and manuscript. AF, MM, HS, and TW contributed images and figures. All authors contributed equally to writing the text. All authors contributed to the article and approved the submitted version.

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Designing (for) Urban Food Webs

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Interest is growing in designing resilient and ecologically rich urban environments that provide social and ecological benefits. Regenerative and biocentric designs fostering urban ecological habitats including food webs that provide ecosystem services for people and wildlife increasingly are being sought. However, the intentional design of urban landscapes for food webs remains in an early stage with few precedents and many challenges. In this paper, we explore the potential to design (for) urban food webs through collaborations between designers and ecologists. We start by examining the ecology and management of Jamaica Bay in New York City as a case study of an anthropogenic landscape where ecosystems are degraded and the integrity of extant food webs are intertwined with human agency. A subsequent design competition focusing on ecological design and management of this large-scale landscape for animal habitat and ecosystem services for people illustrates how designers approach this anthropogenic landscape. This case study reveals that both designing urban landscapes for food webs and directly designing and manipulating urban food webs are complicated and challenging to achieve and maintain, but they have the potential to increase ecological health of, and enhance ecosystem services in, urban environments. We identify opportunities to capitalize on species interactions across trophic structures and to introduce managed niches in biologically engineered urban systems. The design competition reveals an opportunity to approach urban landscapes and ecological systems creatively through a proactive design process that includes a carefully crafted collaborative approach to constructing ecologically functioning landscapes that can integrate societal demands. As designers increasingly seek to build, adapt, and manage urban environments effectively, it will be critical to resolve the contradictions and challenges associated with human needs, ecosystem dynamics, and interacting assemblages of species. Ecologists and designers are still discovering and experimenting with designing (for) urban food webs and fostering species interactions within them. We recommend generating prototypes of urban food webs through a learning-by-doing approach in urban development projects. Design and implementation of urban food webs also can lead to research opportunities involving monitoring and experiments that identify and solve challenges of food-web construction while supporting and encouraging ongoing management.

Keywords: urban ecology, urban food webs, landscape architecture, ecological design, species assemblages, trophic structures, design process

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1. INTRODUCTION

For as long as ecologists have been describing and studying **ecosystems**[1], they have documented the occurrence, structure, interactions, and associated dynamics of species within food webs (**Figure 1**; McCann, 2012). More than a century of observational and experimental studies in natural ecosystems has revealed that food webs assemble themselves from rich pools of dozens, hundreds, or even thousands of species. A range of strong and weak predator-prey interactions and the flow of energy from producers to consumers to decomposers work in concert to exert both bottom-up and top-down control on the structure and dynamics of food webs (e.g., Ulanowicz et al., 2014).

Ecologists disagree about the effects of urbanization on biodiversity—the pool of species from which food webs assemble. Some studies have reported that biodiversity is reduced and homogenized with increasing human occupation and urbanization (McKinney, 2006; Groffman et al., 2014). Others have found that **urban** systems have considerable species diversity and support locally rare or endangered native species. For example, changes in productivity and habitat structures in cities may favor specific species of birds (Marzluff and Ewing, 2008), spiders (Shochat et al., 2008), bees (Mata et al., 2019), or amphibians (Hamer and McDonnell, 2008). This urban biodiversity increasingly is valued and deliberately cultivated in more intentionally designed urban green spaces (Aronson et al., 2017). For example, homeowners may deliberately introduce a single trophic level (e.g., "butterfly plants") to attract herbivores and pollinators (e.g., caterpillars and butterflies), and perhaps unintentionally, their predators (e.g., birds and small mammals) to their yards or gardens (e.g., USDA Natural Resources Conservation Service, 2020). Reconciliation ecology in particular has a unique perspective on fostering biodiversity in human-dominated ecosystems, positing that ecological engineering can shift the abiotic and biotic conditions of more artificial urban analogs to support native and indigenous biodiversity (Rosenzweig, 2003). Through reconciliation, ecologists can make urban environment more suitable for native and indigenous biodiversity and foster wildlife systems that structurally or functionally resemble natural ecosystems and habitats from the region (Lundholm and Richardson, 2010).

A logical next step is to consider entire food webs in urban habitats. We recognize distinctive degrees of structure and composition, utility for people ("**ecosystem services**"), and **intentionality**—all at different spatial scales (**Figure 2**; and see Faeth et al., 2005; McKinney, 2006; Warren et al., 2006)—among **urban food webs**. For example, many urban food webs occur in the seemingly pre-urban conditions found in semi-natural fragmented forest remnants or urban streams (e.g., Faeth et al., 2011; Adler and Tanner, 2013; Forman, 2014). Others occur unintentionally in small patches or successional **landscapes**, such as overgrown median strips or accidental wetlands (Nassauer, 2012; Adler and Tanner, 2013; Beninde et al., 2015; Palta et al., 2017).

Clearly many species interactions and networks are reshaped by urbanization (Forman, 2014). Remnant urban food webs have been shown to support less diverse ecological communities, with specialist species and rare species being particularly vulnerable. Urbanization can be seen as destabilizing, reshaping ecological stability and (dis)assembling species interactions (Start et al., 2020). We note that our intent with **Figure 2** is not to define the only possible sets of food webs in specific combinations of ecological function, ecological intentionality, and space [i.e., the locations of the images in (x,y,z) space in **Figure 2**] but rather to encourage readers to consider the myriad possibilities about how effects of these different variables may play out in "natural" and designed food webs at different scales.

But urban food webs need to be considered differently from those in "natural" ecosystems. Habitat modifications, species introductions, and dispersal across fragmented and heterogeneous urban landscapes are environmental filters influencing composition of species assemblages and structure of urban food webs (Andrade et al., 2020). **Human agency** in the deliberate **design** of urban green spaces and land **management** practices provides a further set of structuring filters (Hagen et al., 2012). Urban food webs interact with human socioeconomic and cultural systems, including designed landscapes, governance structures, and social networks. The uniquely defined ecological communities and food webs found in urban ecosystems have been described using a variety of terms, from "urban assemblages" (Aronson et al., 2017) and "urban metacommunities" (Andrade et al., 2020) to "ecological networks across environmental gradients" (Tylianakis and Morris, 2017), "ecological networks" (or "meta-networks") (Mata et al., 2019), "interaction networks" (Start et al., 2020), living shorelines, and **green infrastructure** (Hostetler et al., 2011), as well as the more concise "urban food webs" (Faeth et al., 2005; Warren et al., 2006; Aronson et al., 2017)[2]. Each of these terms includes concepts from community, ecosystem, landscape, and **urban ecology**. While urban food-web research is expanding, the impact of urbanization and human activities on food-web structure remain poorly understood because cities are challenging spaces for ecologists interested in studying and deciphering food-web structure and dynamics (Andrade et al., 2020).

Although ecologists are moving forward slowly, landscape architects and design professionals are more rapidly exploring and incorporating food webs in urban projects through habitat creation, regenerative design, biocentric design, biophilia, urban agriculture, permaculture, urban restoration projects, and commissioned landscape projects, such as large parks or botanical gardens (see examples in section 5 and Czerniak and Hargreaves, 2007; Beck, 2013). Large infrastructure projects, such as stormwater parks, utility corridors, or management zones that encompass or are adjacent to existing habitats and waterways may require habitat mitigation for particular species or provide environmental conditions in which urban food webs

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[1] Terms defined in the Glossary are set in bold-face type at first mention.

[2] A search run in Web of Science on 22 November 2020 retrieved 31,243 papers published between 1900 and 2020 on "food web" or "food webs" but only seven published on "urban food web" or "urban food webs."

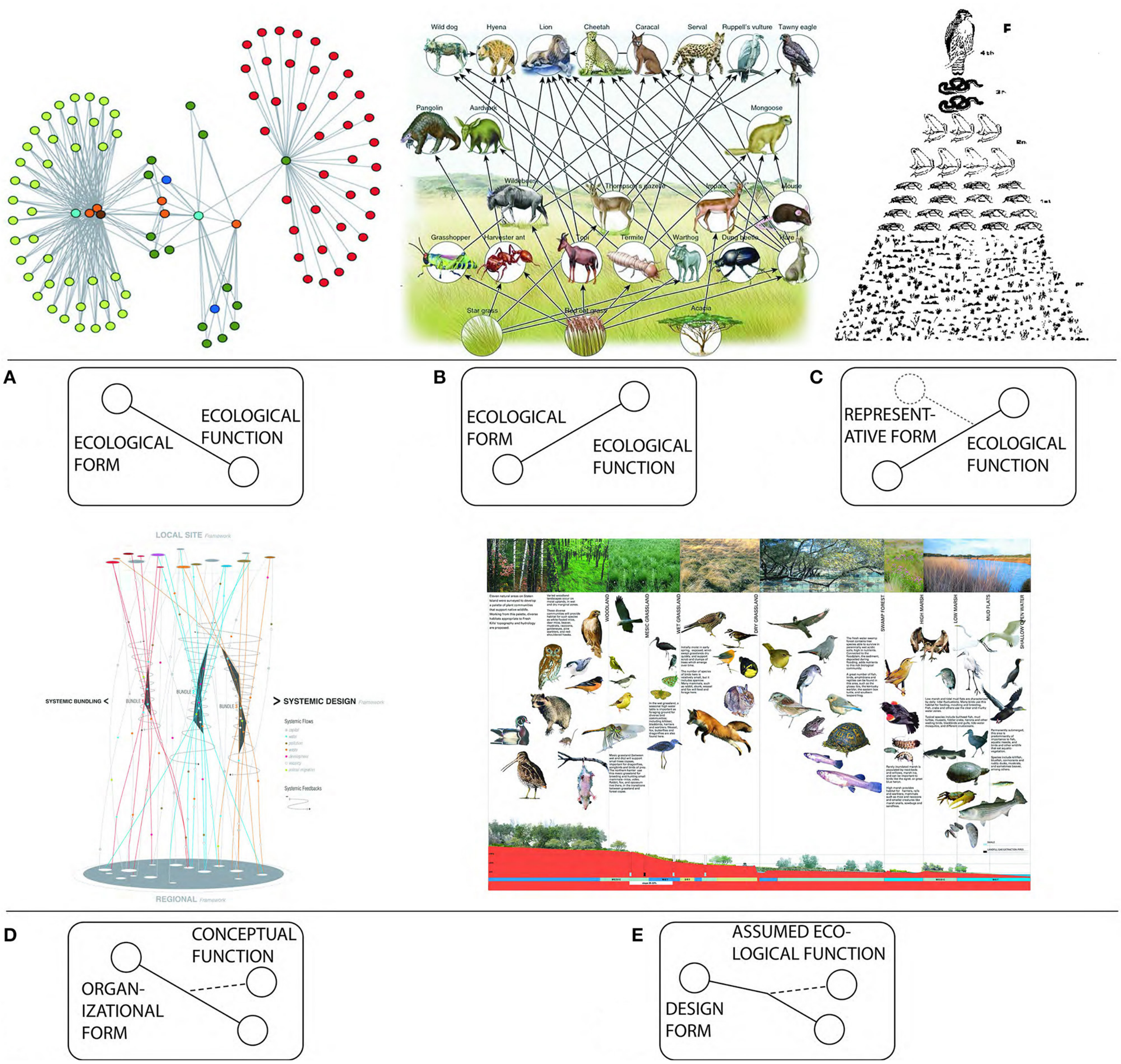

FIGURE 1 | Food webs as envisioned by ecologists and designers place different emphases on their structure (form) and dynamics (function or ecosystem services). The large color panels illustrate portrayal by ecologists of food webs **(A,B)** and their pedagogical representation **(C)**; and their physical embodiment by designers and landscape architects for the built environment **(D,E)**. The "see-saw" icon below each color panel indicates the relative emphasis on form and function in the associated panel. Form (on the left) or function (on the right) may be emphasized (upper left or right) or de-emphasized (lower left or right) and there is usually (but not always, as in **C**) a trade-off (see-sawing) between form and function in emphasis by ecologists or designers. For additional discussion and illustration, see, respectively main text section 6.4 and sliders below photographs in **Figure 10**. **(A)** Is an abstraction of a food web that illustrates its topology. Colored nodes identify functions (e.g., producer, consumer) but there is no identification of who eats whom, directionality, or interactions. **(B)** Is an illustration of a food web that emphasizes functions (as interrelationships across organisms and trophic structures) using directional arrows to indicate who eats whom. **(C)** Is a "food pyramid" like those seen in high-school textbooks that emphasizes trophic transfer and the decreasing abundance of individuals at increasingly higher trophic levels. **(D)** Illustrates a "Systemic Design" framework for managing interactions and information translated across scales in a hierarchy of nested relationships. Information is bundled into clusters ("Systemic Bundles") that tie local site conditions to regional perspectives and expose issues surrounding any given site. **(E)** Is a design-based representation by James Corner Field Operations of an idealized process of ecological succession and food-web development in an urban landfill-turned-public park. Image sources: **(A)** Image produced with `FoodWeb3D`, and provided by the Pacific Ecoinformatics and Computational Ecology Lab (http://www.foodwebs.org/; Yoon et al., 2004); **(B)** Savanna food web from https://savannabiomeassignment.weebly.com/food-web.html; **(C)** Image in the public domain from Brace (1977); **(D)** Alan M. Berger and Casey L. Brown, P-REX, published in Berger (2009) and used with permission; **(E)** James Corner Field Operations, and used with permission.

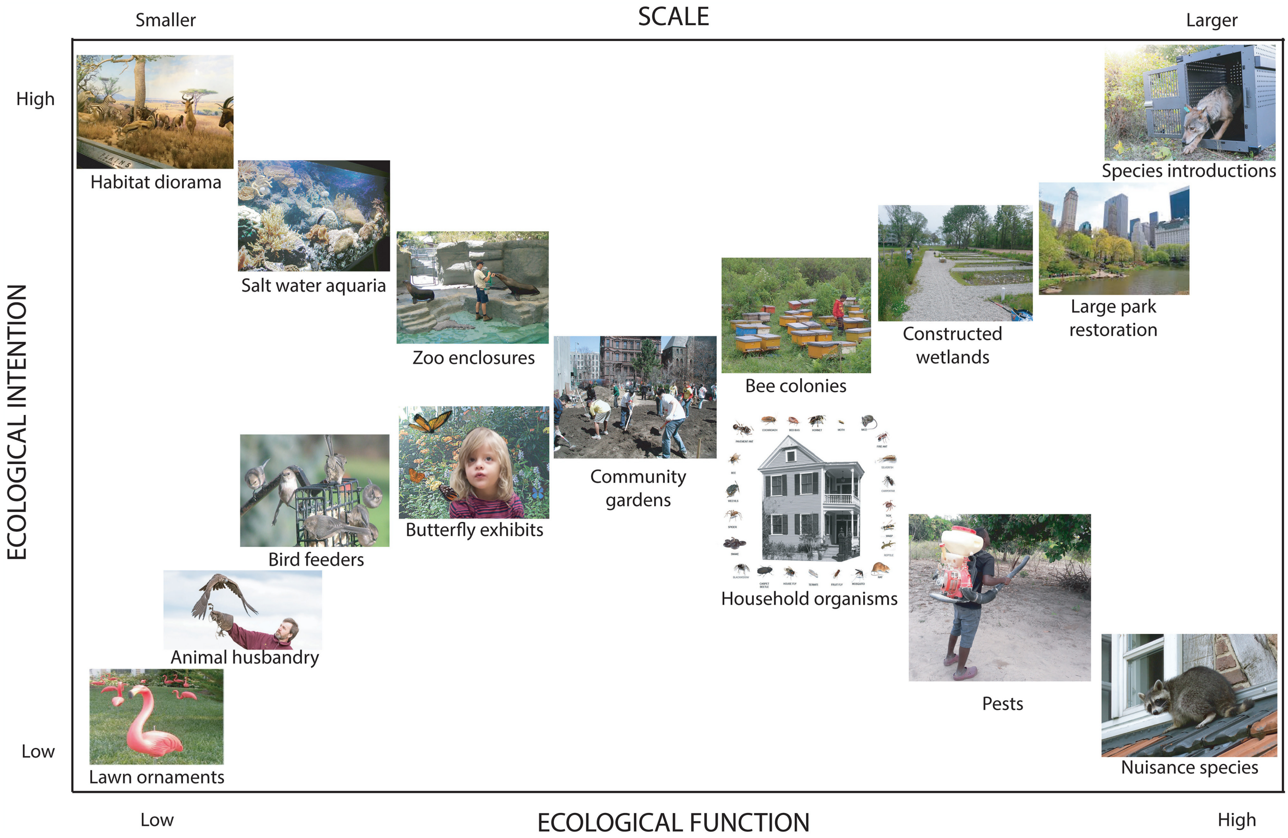

FIGURE 2 | Constructed urban food webs are an outcome of both intentional and unintentional human activities and come in many shapes and sizes. This figure illustrates relationships between ecological functions and human intention to modify the local ecology in real and representational food webs. For designed food webs, we need to combine intentionality with theories of biological succession, invasion, species interactions, and other ecological applications to evaluate and manage the systems over time. Image sources: Lawn ornaments (Bert bert: https://commons.wikimedia.org/wiki/File:Flamingo_1.jpg; CC-BY); Animal husbandry (Government of Victoria: https://www.yarracity.vic.gov.au/events/2018/06/02/hawk-owl-falconry-display; public domain); Bird feeders (Scott Catron: https://commons.wikimedia.org/w/index.php?curid=849195; CC-BY-2.5); Butterfly exhibits (Alexander Felson: photomontage, 2020; used with permission); Community gardens (Alexander Felson: photograph, 2005; used with permission); Habitat diorama (NComparato: https://commons.wikimedia.org/wiki/File:Plains_Diorama.JPG; CC-BY-SA 3.0); Salt water aquaria (Fritz Geller-Grimm; https://commons.wikimedia.org/wiki/File:Aquarium_fg01.jpg; CC-BY-SA-2.5); Zoo enclosures (Fir0002: https://commons.wikimedia.org/wiki/File:Seals@melb_zoo.jpg; CC-BY-SA-3.0); Bee colonies (Okkisafire: http://self.gutenberg.org/articles/eng/File:Beekeeping_at_Kawah_Ijen,_Indonesia.jpg; CC-BY-SA 3.0); Constructed wetlands (Alexander Felson: photograph, 2020; used with permission); Large park restoration (Ingfbruno: https://commons.wikimedia.org/wiki/File:3037-Central_Park-The_Pond.JPG; CC-BY-SA 3.0); Species introductions (US National Park Service: https://www.nps.gov/isro/learn/news/isle-royale-national-park-and-partners-release-two-wolves-on-the-island.htm; public domain); Household organisms (Alexander Felson: photomontage, 2021; used with permission); Pests (Kente I: https://commons.wikimedia.org/wiki/File:Kente_I.jpg; CC BY-SA 4.0); Nuisance species (Carsten Volkwein; https://upload.wikimedia.org/wikipedia/commons/7/7f/Waschbaer_auf_dem_Dach.jpg; CC BY-SA 2.5).

can self-assemble and opportunities for more intentionally-designed and constructed urban food webs. However, whether we can design and manage urban food webs remains an open and challenging question.

2. ROADMAP

In this paper, we articulate a vision for connecting ecological theory (i.e., concepts, ideas, and generalizations) with **landscape architecture** and urban design to guide the cultivation and construction of urban food webs. We begin with short précis that briefly reviews the history of how ecologists have thought about design and how designers have engaged with ecologists (section 3). We include in this section a Glossary of Terms (**Box 1**) used regularly by ecologists and designers but often with very different meanings. We follow this historical overview with a description and analysis of a single case study of the ecology and management of the food web within Jamaica Bay's Gateway National Park within New York City (section 4). This case study illustrates how ideas and issues that will be familiar to ecologists are addressed and reinterpreted by designers. It shows how existing urban food webs are connected to land-use practices and human behaviors through decision making and expressed in **design thinking** and management. Jamaica Bay also has been the locus of two landscape **design competitions** that have foregrounded challenges and opportunities in designing, building, and managing urban food webs. The winning designs illustrate how design professionals working independently of ecologists (section 4.2) or in collaboration with ecologists (section 4.3) use

BOX 1 | **Glossary of Terms** used commonly by designers, that may be unfamiliar to ecologists, and whose use and interpretation differs between ecologists and designers.

- **Aesthetics**: The philosophy of the nature of art, beauty, and taste (Adorno, 1997). Applied in design projects to the creation and appreciation of form, layout, materials, and function. From aisthētós (Greek: "sensible, perceptible"; by extension: "to perceive, take notice of, understand").
- **Design**: The process of inventing and creating proposals for buildings, landscapes, and urban artifacts through rigorous inquiry and individual creativity.
  - **Design competition**: An organized competition in which design teams submit proposals responding to a site-specific or conceptual competition brief; winners may or may not be able to actually build the proposed project.
  - **Design process**: Project activities from its initial analysis and conceptualization through design and construction.
  - **Design thinking**: Process of sketching and conceptualizing ideas into design solutions from inquiry through crystallization (Rowe, 1996).
  - **Parametric design**: Computational process that uses algorithmic-based software to generate responsive and immersive three-dimensional models used in (landscape) architecture.
  - **Working methods of design** include iterative approaches to site analysis, conceptual and schematic design, design detailing, **material selection**, communication with clients and stakeholder, design, and construction (Felson et al., 2013b).
- **Green infrastructure**: Land and water management using vegetation, water, and modified natural process to restore and sustain hydrologic and habitat functions.
- **Ecological infrastructure**: Green infrastructure that prioritizes ecosystem functions over engineering infrastructure and re-adapts landscapes and associated ecological processes into multifunctional open spaces.
- **Ecosystem**: Biotic and abiotic elements and all their interactions in a geographic area.
  - **Designer ecosystems**: ecosystems in which ecological goals have been articulated explicitly, management actions are employed and evaluated to achieve those goals, and where the goals and actions are continually optimized as new information becomes available (Ross et al., 2015).
  - **Ecosystem functions**: Processes that affect biological (or human) communities. Examples include climate control, productivity, energy flow, and nutrient cycling.
  - **Ecosystem services**: Benefits provided by nature to humans, usually economically valued (Millennium Ecosystem Assessment, 2005). Categories include provisioning, regulating, supporting, and cultural ecosystem services.
  - **Novel ecosystems**: Species combinations or environmental configurations that are not thought to have existed previously (Hobbs et al., 2009). For landscape architects, novel ecosystems are common features of design projects.
- **Form**: The shape or structure of an object.
- **Function**: The purpose or performance of a structure or system; the relationship from inputs to possible outputs.
- **Human agency**: The capacity and freedom to make our own choices and to have those choices impact the world around us.
- **Intentional(ity)**: The conscious or subconscious directedness of ones' mind toward a particular task or agenda.
- **Landscape**: Socio-ecological systems in a bounded area consisting of a mosaic and gradient of more natural to human-modified ecosystems. In art, land that can be seen and comprehended in a single view. From *landschap* (Dutch: a collection of farms or fenced fields); *landschaft* (German: land shared by people); also *paesaggio* (Italian), *paysage* (French); *paisaje* (Spanish).
  - **Landscape architecture**: The professional and academic discipline focused on planning, design, management, stewardship, and nurturing of urban, rural, and natural environments, including small-scale trails, streetscapes, plazas, and residences, and regional-scale parklands, urban parks, and campuses.
  - **Landscape urbanism**: A way of thinking about cities through a landscape lens. Includes notions of territory, contemporary city forms, systems; considers landscape, infrastructure, and ecology as equally important components of cities.
- **Management**: The process of using people and resources to achieve targeted goals. In design, the deployment, effective use, and coordination of finances, technology, and people to efficiently manage the built environment and natural resources.
- **Maintenance**: Activities required to forestall deterioration of a landscape. Includes mowing, pruning, tree removal, edging, cultivation, fertilization, and pesticide application.
- **Material selection**: Process of researching and selecting materials—constrained by budget and availability—for distinct areas in a design.
- **Urban**: Characteristics of a town or city, usually defined by density. In the US, a cluster of areas with 2,500–50,000 people or a single area of ≥50,000 people[3].
  - **Urban ecology**: The study of (socio-)ecological processes in cities and urbanized regions.
  - **Urban food web**: Multi-trophic assemblages of interacting species found throughout human-dominated rural, exurban, suburban, and urban built environments. Examples of urban food webs include microbiomes and their hosts in single-family homes, plants, pollinators, and insect herbivores in community gardens, and predators and their prey thriving within planned residential neighborhoods.

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[3] https://www.census.gov/programs-surveys/geography/guidance/geo-areas/urban-rural.html.

a creative process that includes hybrid ecological, recreational, and educational solutions and encourages investment in habitat management for the sake of people and the threatened species.

In section 5, we build on the case study to discuss approaches already being applied by design professionals interested in designing and managing for food webs. We differentiate design *for* food webs from *intentionally designed* food webs in urban environments. We emphasize that designers are already designing (for) food webs even though many ecologists are likely to question the rationale for, or feasibility of, establishing stable urban food webs. We also recognize that designing for food webs through habitat restoration or "rewilding" (e.g., Lorimer et al., 2015; Corlett, 2016; Svenning et al., 2016) is pragmatic and intentional and shares some features with designed food webs proposed by design professionals. The paper concludes with a discussion of the key challenges associated with building bridges between ecologists and design professionals that could help achieve the goal of establishing urban food webs (section 6). We identify roles for ecologists in developing and applying relevant theories needed to intentionally design urban food webs (Pulliam and Johnson, 2002; Grose, 2017) that build on a growing literature establishing ecological principles for landscape design (e.g., Beck, 2013).

3. A BRIEF HISTORY OF INTERSECTIONS BETWEEN ECOLOGISTS AND DESIGNERS

Ecologists are increasingly working with designers and other urban practitioners involved in shaping cities (Felson et al., 2013b; Tanner et al., 2014; Pickett et al., 2020). These interactions can be challenging but also have potential to enrich all the partners (e.g., Johnson et al., 2002; Musacchio, 2009; Pickett et al., 2016).

3.1. Ecologists Increasingly Are Considering Design

Ecologists interested in urban areas initially focused on understanding autecology and interactions of non-human populations and communities *in* cities (e.g., Adams, 2005; Bornkamm, 2007; Sukopp, 2008). These and other early studies relied on a direct translation of classical ecological concepts and theories developed in non-urban systems into cities (Rebele, 1994). Urban ecology has begun focusing on ecology *of* cities to integrate the built environment and human behavior (Alberti et al., 2009; Pickett et al., 2011; Tanner et al., 2014; Groffman et al., 2017). Concurrently, ecologists are actively revisiting their role in urban research, applied science and design (Felson and Pickett, 2005; Palmer, 2009; Pickett et al., 2020). Most recently, ecologists are embracing ecology *for* cities (Childers et al., 2015; Pickett et al., 2016) and working directly with designers through the **design process**.

3.2. The Ebb and Flow of Ecology in Design

Among designers, Ian McHarg heavily influenced the integration of ecology into landscape design, particularly through education (McHarg, 1969). He framed the ecological paradigm around disturbance ecology as a fundamental part of ecosystems and invited ecologists into design schools to teach ecological theory to design students (Spirn, 2000). Many of his students went on to become leaders whose work and that of their students and colleagues influenced the field of landscape architecture worldwide (e.g„ Spirn, 2000; Steiner et al., 2013, 2019; Reed and Lister, 2014; Grose, 2017)[4]. In the last 50 years, the place and influence of ecology in the design professions has waxed and waned with changing priorities and demands on designers (Steiner et al., 2019). These shifting priorities have reflected the responsiveness of urban design to contemporary cultural attitudes, needs, and interests, all of which can limit opportunities for designers to develop a deeper working knowledge of ecological science than they may have been exposed to during their schooling or receive through continuing education. This is especially evident in the continuing debate about the interplay between **form** and **function** in landscape architecture (section 6.4.2).

3.3. Initial Challenges for Integrating Ecology and Design

It has been said that ecologists have been happy if they understand a system while designers are happy if they invent something new (Charlie Canham, 2007 *personal communication* to Alexander Felson). When they meet to collaborate, ecologists and designers often use identical terms to refer to very different concepts and practices (**Figure 3**; **Box 1**; and Ellison and Buckley Borden, 2021). A key point of disassociation is that ecologists consider "ecology" as a single entity and scientific discipline whereas designers often speak of "ecologies" (e.g., Reed and Lister, 2014). Ecologists tend to lack an appreciation for design thinking and rarely consider concerns other than ecological criteria (i.e., issues from ecology as a science), including material constraints, accessibility, and visual cues that influence site design. On the other hand, design professionals, environmental consultants, and other practitioners tend to rely mostly on technical reports, assumptions, and dated ecological knowledge derived from non-urban systems to inform their work (Forman, 2002; Johnson et al., 2002). Although designers have limited time to interpret and synthesize ecological information, **working methods of design** are flexible enough to incorporate ecological input at different stages (Felson et al., 2013b). These cultural differences, distinct ideologies, vocabularies, methods and tools, scales of activity, and approaches to work need to be addressed in any collaboration between ecologists and designers (Ellison and Buckley Borden, 2021). Working closely with ecologists to define these methods is a way forward to ensure that ecological theory informs design outcomes (Felson et al., 2013b). Landscape ecologists have had the most success in building collaborative bridges with designers, providing connections between ecology, sustainability, **aesthetics**, and design (Musacchio, 2011). Additional contemporary hurdles that ecologists and designers navigate are highlighted through our case study (section 4) and current ecology-oriented designs (section 5), and elaborated in the final section (section 6).

4. CASE STUDY: JAMAICA BAY, NEW YORK

We use a single case study to illustrate the complex and often compromised landscapes in which urban food webs exist and where ecological concerns and design goals intersect and may come into conflict. Government management agencies with limited resources and conflicting demands face daunting challenges when attempting to reconstruct habitats while

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[4] The nexus of this continuing work is The Ian L. McHarg Center for Urbanism and Ecology in the Stuart Weitzman School of Design at the University of Pennsylvania: https://mcharg.upenn.edu/.

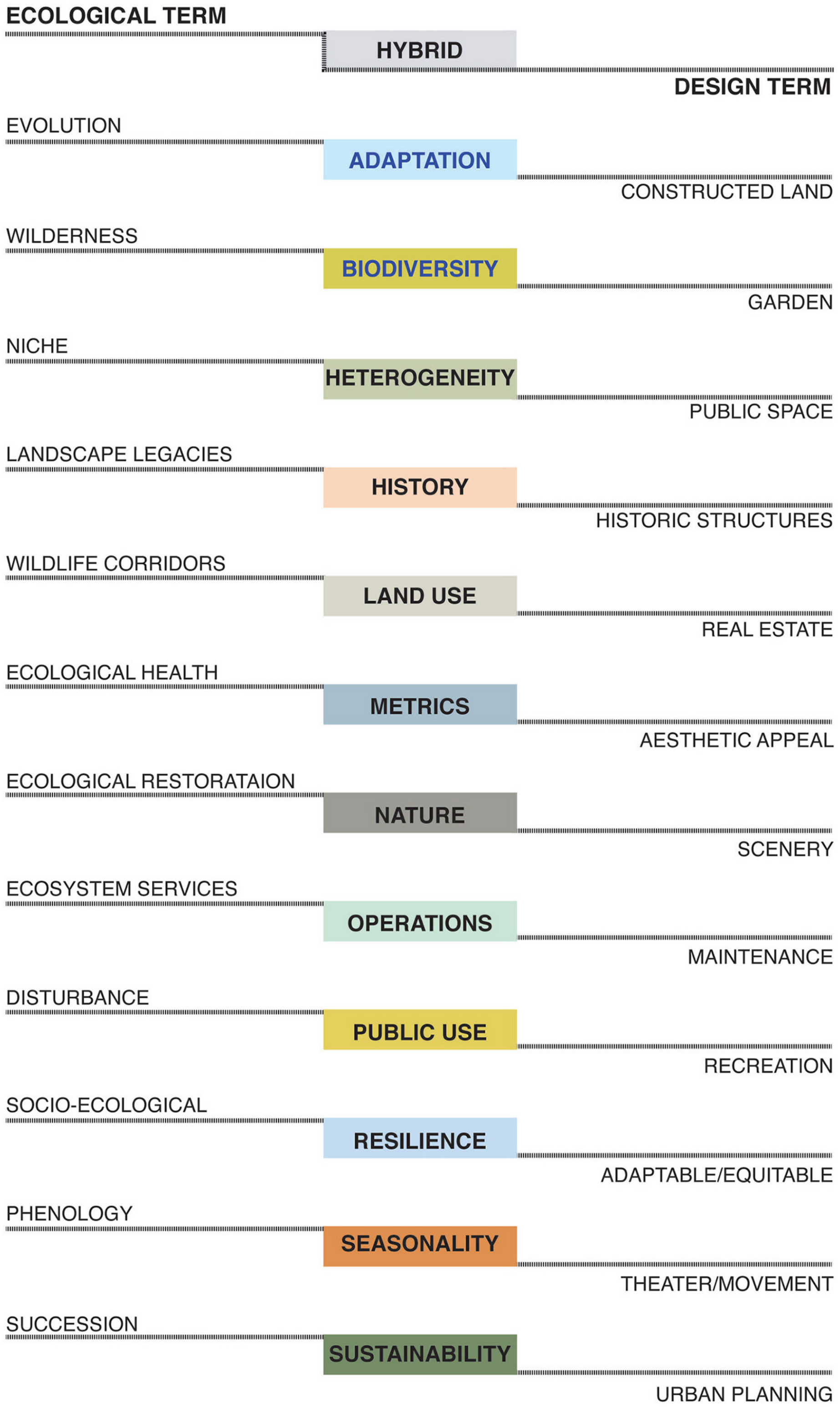

FIGURE 3 | Ecologists (E) and Designers (D) have different interpretations of and use different words for similarly-named conceptual domains. The interpretations or words on the left and right are intentionally offset to suggest differences in understanding between the ecologists and designers (paralleled by the form vs. function "see-saws" in **Figure 1** and "sliders" in **Figure 10**). These differences make it difficult for ecologists and designers to communicate and collaborate effectively. The interpretations and words are connected through their concepts (center) to suggest opportunities for alignment. Identifying and building on areas of shared interest and teaching across disciplines can strengthen collaborations between ecologists and designers. Image by Alexander Felson/UED Lab and used with permission.

simultaneously managing individual species, food webs, and human uses. Past land-use practices in and around Jamaica Bay and what is now Gateway National Park have created a landscape under stress with limited management options and opportunities for change. This same location was the site for influential design competitions in 2007 and 2013 to envision and propose reworking of the landscape and environment of Gateway National Park to integrate people with non-human species, biodiversity, and ecosystems (Brash et al., 2011; Orff, 2016). Together, the case study and design competitions illustrate a shared interest in geographic scales and site challenges, suggesting an expanded collaborative role for designers to work with ecologists in urban food web design and management.

4.1. Managing an Urban Food Web

Jamaica Bay is a 50,586-ha (125,000-acre) saline-to-brackish, nutrient-rich estuary located in the borough of Queens in New York City, New York, USA. Jamaica Bay includes the 10,767-ha (20,607-acre) Gateway National Recreation Area of the US National Park Service (NPS) and the 3,683-ha (9,100-acre) Jamaica Bay National Wildlife Refuge (NWR) managed by the US Fish and Wildlife Service (FWS). Positioned at the intersection of the Atlantic Ocean and the Hudson Raritan estuary, the area includes forests, wetlands, grasslands, dunes, marsh islands, mud flats, tidal creeks, and open water. These ecosystems provide critical habitats for migratory birds, insects, and fish. Jamaica Bay is home to more than 330 seasonal and permanent species of regulatory concern or listed status (including 120 bird species and 48 fish species), and is an important stop-over site for migratory waterbirds (Rafferty and Babson, 2015). The extensive habitats of Jamaica Bay historically supported complex food webs, providing refugia from predators, feeding, spawning grounds, and nurseries for finfish and shellfish, and habitat for resident and migrating birds, crustacean species, and transient and resident fish (Gateway National Recreation Area, 2007). The horseshoe crab (*Limulus polyphemus*) is a central species in the Jamaica Bay food web that connects it to an intercontinental one (**Figure 4**). Horseshoe crabs are fished recreationally and commercially along the eastern US coast and have been overfished to unsustainable levels in the northern portion of its range (including New York; Walls et al., 2002; Smith et al., 2017). Horseshoe crabs also are an important food source for migratory birds, particularly the IUCN-designated "near-threatened" Red Knot (*Calidris canutus*) and Semipalmated Sandpiper (*C. pusilla*); Red Knot populations along their migratory routes are declining in parallel with declines in horseshoe crab populations (Niles et al., 2009).

Because Jamaica Bay is both a wildlife sanctuary and a place for recreation, it is managed by several different state and federal agencies. It has been mapped and designated by FWS and the US Federal Emergency Management Agency (FEMA) as a protected beach unit pursuant to the 1982 Coastal Barrier Resources Act (US Public Law 97-348; reauthorized in 2000 as Public Law 106-514). The New York State Department of State has designated Jamaica Bay and adjacent Breezy Point as Significant Coastal Fish and Wildlife Habitats, and the New York Department of Environmental Conservation (DEC) has designated portions of the bay as Critical Environmental Areas.

Despite its multiple layers of legal protection and management, the amount of habitat and its quality within Jamaica Bay has declined with increasing urban development and associated land- (and water-) use changes, including channel dredging, shoreline hardening, jetty and causeway construction, sewage treatment operations, and sewage and stormwater discharge from Queens. Concurrently, sea levels have risen and islands that once hosted emergent salt marshes have become intertidal or subtidal mudflats (National Parks Conservation Association, 2007; Rafferty and Babson, 2015). Horseshoe crab populations are impacted as much by this habitat degradation and sea-level rise as they are by overfishing (Smith et al., 2017), and since 1998 they have been managed by the Atlantic States Marine Fisheries Commission under the regulatory direction of the Horseshoe Crab Fishery Management Plan (Kirchoff et al., 2009).

Working in partnership with several environmental non-profit organizations, NPS has been tagging and monitoring horseshoe crab populations since 2013 to determine the extent of their relationships with shorebird populations and migration patterns (Gateway National Recreation Area, 2007). The Science and Resilience Institute at Jamaica Bay built on the extensive research from Delaware Bay to understand dynamics of the horseshoe crab and Red Knot (e.g., Kirchoff et al., 2009; Niles et al., 2009) and develop management actions, including translocation and fishing restrictions for these species and associated food webs in Jamaica Bay.

A number of restoration projects have been developed and implemented in response to the habitat degradation and loss of salt marshes within Jamaica Bay NWR. A collaborative interagency restoration project including NPS and DEC restored and monitored Big Egg Marsh from 2003 to 2008. This restoration project and others completed with the US Army Corps of Engineers under the Hudson–Raritan Estuary Comprehensive Restoration Plan (U.S. Army Corps of Engineers, 2012) have improved salt-marsh restoration approaches that are being used to enhance planning and implementation in other salt marshes (National Parks Conservation Association, 2007; Rafferty et al., 2011). These restoration projects also established collaborative partnerships focused on biodiversity and food-web development. For example, with each successive restoration project, partners were increasingly willing to adapt the NPS's preference for restoring marshes at higher elevations to support distinct species assemblages and add resiliency of the system to sea-level rise. Research also identified effects of hydrologic modifications, eutrophication, and sea-level rise on salt-marsh loss (U.S. Army Corps of Engineers, 2012).

Ultimately Jamaica Bay faces many complex challenges that require coordinated, collaborative efforts not only by ecologists and designers but also by many other stakeholders and individuals who use and care about its long-term sustainability. Two design competitions began the process of envisioning and developing these efforts.

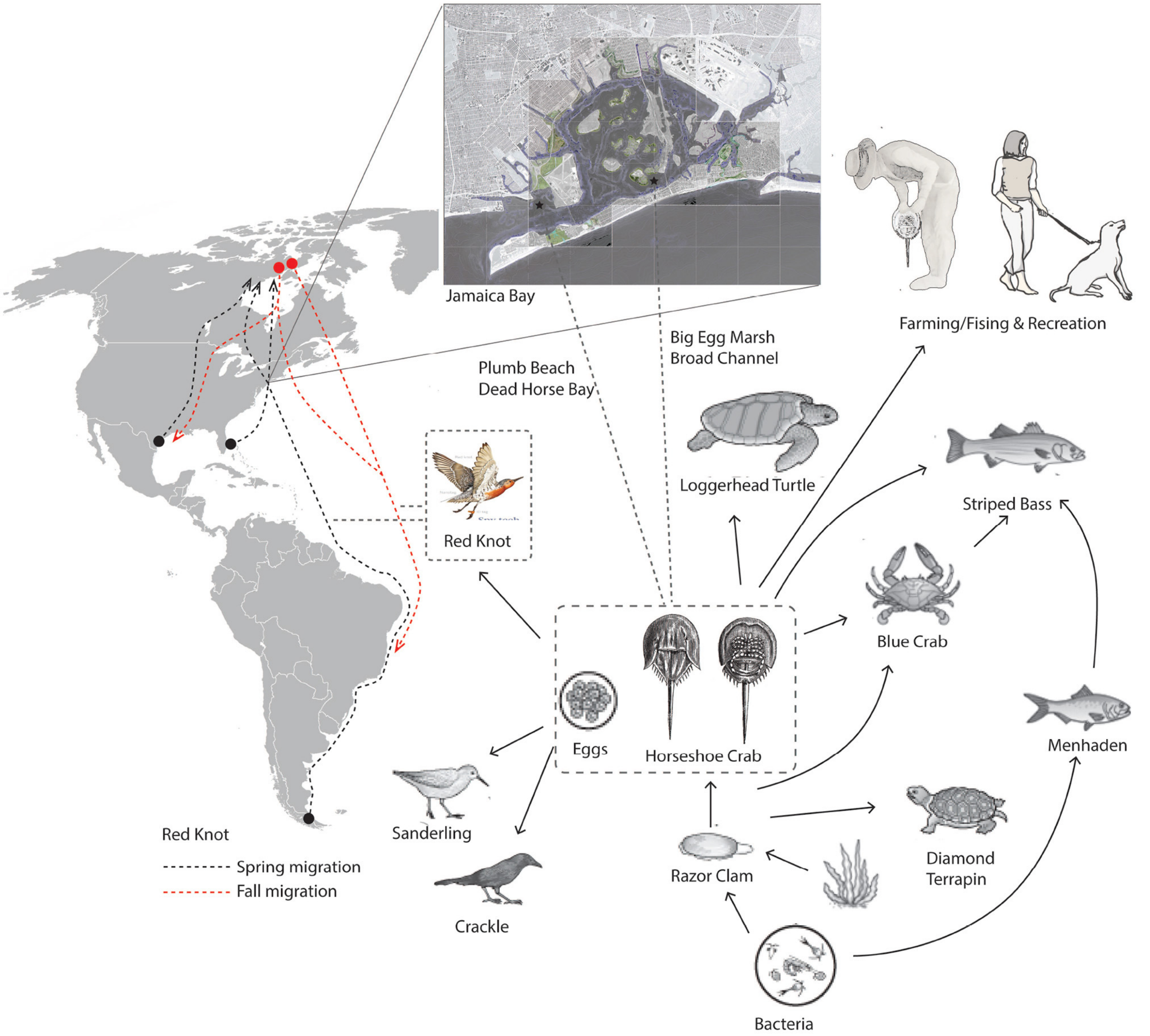

FIGURE 4 | The food web of Jamaica Bay has local, continental, and transcontinental linkages, and includes people and marine and terrestrial species, some of which are of conservation concern. Image by Alexander Felson/UED Lab and used with permission.

4.2. The "Envisioning Gateway" Design Competition

In January 2007, the "Envisioning Gateway" international competition invited design teams to propose new visions for the future of the Gateway National Park. Set up as a collaboration between the Van Alen Institute, National Parks Conservation Association, and Columbia University Graduate School of Architecture Planning and Preservation, "Envisioning Gateway" sought to tackle the overlapping challenges of this regional national park. This competition encouraged applicants to focus on history, recreation, ecology, waste management, access and transportation, and economic strategies in creating a vision for landscape adaptation that incorporated the converging fields of design, science, technology, and art[5]. The winning entries, from a pool of 230 entrants representing 23 countries, articulated a vision for human investment in ecosystems for the sake of the species alongside the value for humans. They also illustrated how landscape designers are well ahead of applied ecologists and

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[5] The description framing the competition conveyed this critical intersection of human and natural concerns: "*Gateway presents a significant regional resource with incredible infrastructural, ecological and cultural value in the New York metropolitan region, hosting endangered birds, fish and shellfish breeding grounds, marinas, playfields, and cultural relics. It is also the site of combined sewer outfalls, treated wastewater effluent, abandoned buildings, degraded habitat, drowned marshes, former landfills and vast asphalt runways.*" See https://www.vanalen.org/projects/envisioning-gateway/#resources.

restoration ecologists in their willingness to create truly **novel ecosystems** with embedded assemblages of interacting species, including people.

The overall winning design, "ECOTONES" by Ashley Kelly and Rikako Wakabayashi (Brooklyn, NY, USA), focused on "zones of ecological tension...an urban park that creates a microcosm of shifting habitats, program and landforms." Their design proposed that "people should be educated that ecosystems are in necessary flux, a cycle increasingly complex with today's global climate shifts" (**Figure 5**). The second place winner, "Reassembling Ecologies" by North Design Office (Toronto, Canada), focused on "optimal conditions for strengthened aquatic and terrestrial ecologies in and around the Gateway National Recreation Park...[and] the need for a stronger definition of programs and activities for park users and an enhanced structure that allows for sensitive ecologies and recreation to coexist." Reassembling Ecologies also proposed intertwining salt-marsh habitats with beach-goers and boaters in way that would have requiring a heightened sensitivity among human users to avoid the likely seasonal conflicts between people and nature.

One honorable mention was awarded to [UN]NATURAL SELECTION by Archipelago Architecture and Landscape Architecture (New York, NY, USA). This proposal highlighted differences between classical (rural) national parks focused on preserving scenic beauty and urban national parks, such as Gateway, which are "all about human contact with the environment and the resulting changes that take place over the centuries. ...[I]n the 21st century we are beginning to understand that human health and ecology exist within and not separate from the surrounding environment. Gateway, situated between the nation's greatest concentration of humanity and the ocean, presents a special opportunity to explore the connections and tensions inherent in this amalgam of ocean, land, air and settlement." A second honorable mention was awarded to Urban Barometer by LOOP|8: Christopher Marcinkoski, Andrew Moddrell (Larchmont, NY, USA), which suggested that Gateway "should be re-designated as a National Eco-urban Research Zone—a territory that both promotes stewardship of existing natural and native resources, but also engages in the active exploration of the relationship between dynamic ecosystems and ongoing anthropologic urbanization."

"Envisioning Gateway" was a good first step toward addressing the complexity of Jamaica Bay but it didn't take on many ecological challenges. The competition brief provided only a habitat-based framing[6]. In its formulation, it recast ecological goals in the context of a working landscape in the public realm and de-emphasized species interactions, food webs, and other concerns of academic ecologists.

4.3. The "Rebuild by Design" Competition

Six years after the Envisioning Gateway design competition, "Rebuild by Design" revisited the role of collaborative design thinking for Jamaca Bay. This global competition was launched in 2013 by the US government and the Rockefeller Institute in response to Hurricane Sandy[7]. Designers were invited to address coastal resilience and flooding from large storms and ongoing sea-level rise. The winning proposal, "THE SHALLOWS: Bay Landscapes as Ecological Infrastructure," was submitted by the landscape-architecture firm SCAPE and their interdisciplinary team of ecologists, engineers, and designers (including Parsons Brinckerhoff, Philip Orton, SeArc Ecological Consulting, LOT-EK, MTWTF, the Harbor School, and Paul Greenberg) and clearly illustrated an integrated design-thinking approach to addressing challenges of habitat restoration, biodiversity, and large-scale urban food webs.

THE SHALLOWS reimagined how coastal flood infrastructure could simultaneously reduce flood risk and be a social and ecological catalyst. It presented alternative hydrologic regimes and the introduction of a series of architectural habitat generators, including a "habitat breakwater" and "constructed reefs" (SCAPE/Landscape Architecture [Kate Orff], 2015). The breakwater systems were designed to slow erosion and reduce wave action while also supporting the reestablishment of marine ecosystems through the creation of distinct niches. The architectural materials were intended to mimic the distinct "micro-complexity" of the historic reefs of Raritan Bay, creating shelter and foraging areas for finfish, lobsters, and shellfish (SCAPE/Landscape Architecture [Kate Orff], 2015). Like ECOTONES, THE SHALLOWS has not yet been built[8].

"Rebuild by Design" represents a much greater emphasis on integrating ecology, design, and broader communities in envisioning and constructing a sustainable future. The results of the Hurricane Sandy competition and others subsequent to it point toward a future of stronger collaborations between ecologists and designers in many urban landscapes.

5. DESIGNING URBAN LANDSCAPES WITH FOOD WEBS IN MIND

In section 4, we illustrated the growing interest in visions focused on urban design and construction for biodiversity, food webs, and ecosystem services that blend nature into human environments. Work in this area is being led by designers and landscape architects; ecologists are infrequently included in the design, engineering, or construction teams. We recognize that designing food webs—urban or otherwise—remains aspirational for ecologists (including ourselves) and that standard ecological approaches, such as species (re)introductions or removals, habitat restoration, and rewilding to promote specific assemblages of species are still being developed, studied, and tested. Although ecological theory tends to be included sporadically in the design professionals' toolbox, it is clear that the goal of establishing stable urban food webs is part of the growing interest in biocentric design and construction of biophilic cities. If ecologists can collaborate with designers

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[6] https://www.vanalen.org/projects/envisioning-gateway/#sitebrief.

[7] http://www.rebuildbydesign.org/our-work/sandy-projects.

[8] The low likelihood of winning designs in design competitions actually being built may dissuade ecologists from working with designers on them. But in the design community, winning a design competition is a lot like an ecologist getting a paper accepted and published in *Nature* or *Science*!

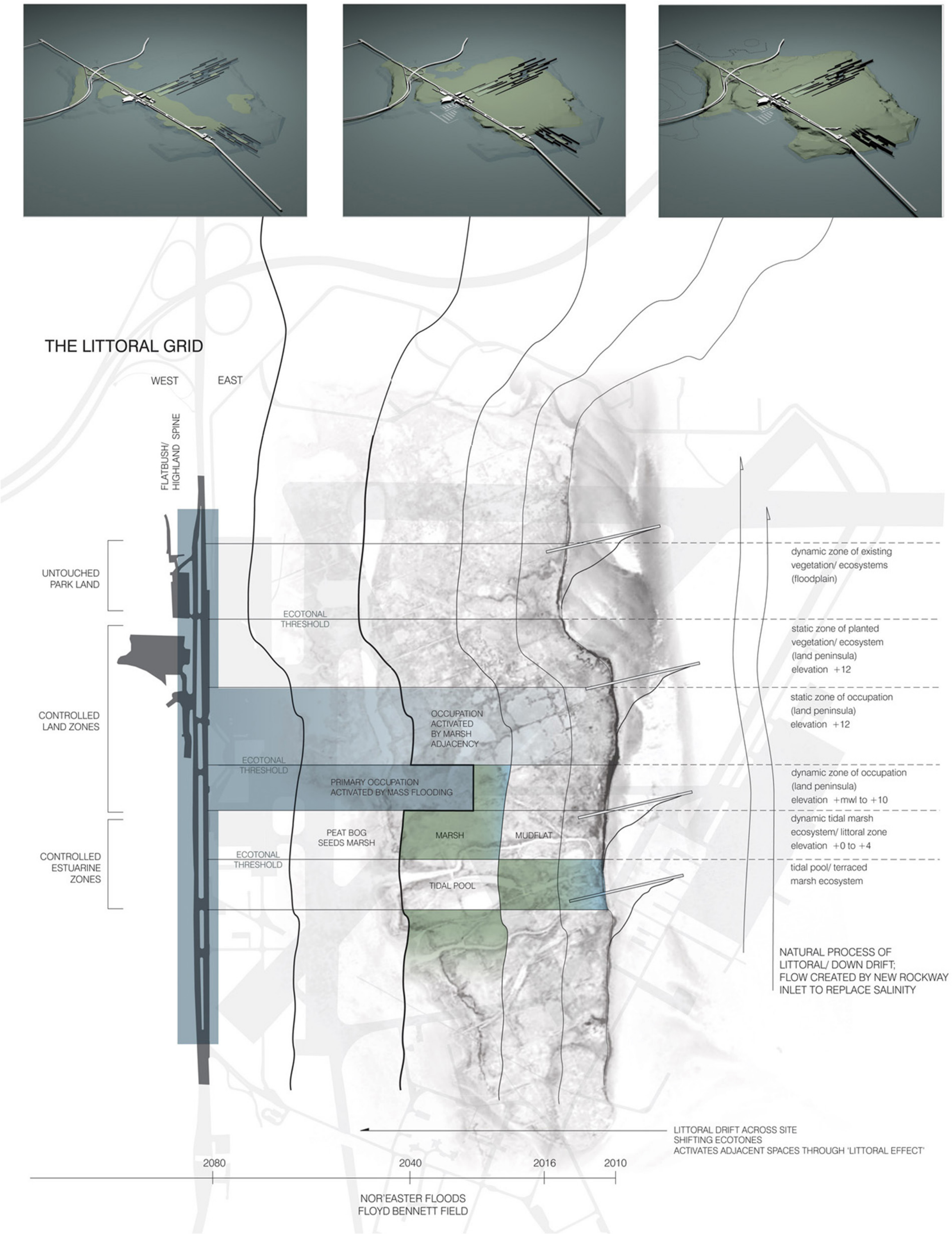

FIGURE 5 | ECOTONES, the winning design of the Van Alen Institute's Envisioning Gateway design competition for Jamaica Bay by Ashley Kelly and Rikako Wakabayashi. Illustration by Ashley Kelly and Rikako Wakabayashi and used with permission.

in creating urban landscapes and in managing them, there is a higher likelihood of creating stable food webs in them. But pursuing such collaborations will require ecologists to commit to working in tandem with informed design professionals.

Collaborations between ecologists and designers to date have led mostly to unintentional (un-designed) food webs (**Figure 2** and section 5.1), but in a few, recent instances, these collaborations have led to more intentional, designed ones (section 5.2). In either case, these collaborations have tended to target vegetation, soil, or water (e.g., Calkins, 2012); emphases have been on biodiversity and ecosystem services. Notable examples include the UrBioNet program[9] and the annual International Urban Wildlife Conference[10].

5.1. Unintentional Urban Food Webs

We distinguish three types of, or approaches to, the rehabilitation, restoration, or planting of vegetation and developing habitats that might eventually support foodwebs. Each of these approaches will be familiar to ecologists. Although the eventual occurrence of food webs in these habitats may be desired, we emphasize that the actual species composition that may arise usually is unintentional.

First, urban food webs with desirable or undesirable species may arise spontaneously (Del Tredici, 2010) as "unintended side effects of land management." For example, unintentional foodwebs can assemble in urban stormwater retention ponds or catch-basins that create wetlands and attract wildlife or act as refugia for species of conservation concern (e.g., amphibians; Hassall, 2014; Palta et al., 2017). Depending on their location and upland land uses, these "refugia" may actually function as ecological "sinks" (*sensu* Pulliam, 1988). Similarly, bird feeders set out by urban and suburban gardeners are used to attract birds, but may also attract squirrels, black bears, white-tailed deer, white-footed mice, deer ticks, and spirochetes, all of which can carry a variety of diseases (Goddard and Dougill, 2010; Sterba, 2012). In contrast, forgotten clover-leaves at highway interchanges or abandoned vacant lots, for example, can support multi-trophic food webs that have assembled themselves without any human intervention save for the inadvertent creation of "unused" space (Grewal et al., 2011).

Second, designers may "let nature take its course" by leaving a system alone either intentionally or because of lack of time, knowledge, or resources (see also the "Wildlands" approach of Foster et al., 2010 and the urban re-greening studied by Gallagher et al., 2018). Letting nature take its course builds on the well-established conservation strategy where parcels of land, typically of relatively high ecological value (e.g., waterways, riparian buffers, steep slopes, wetland complexes, or remnant forest patches), are identified and conserved over time. Landscape architects also have a history of embracing aspects of letting nature take its course in their designs. For example, in remediating degraded sites, Julie Bargmann (University of Virginia) embraces their beauty and the need to preserve, rather than erase, their histories. Her D.I.R.T. [Dump It Right There] Studio incorporates an economy of means alongside urban succession to help reshape and regenerate post-industrial sites. The goal of Bargmann's design approach with "the good, bad and the ugly of post-industrial sites is to challenge the persistent pastoral ideal. That beauty emerges through empathy with how that landscape has come to be, what is latent within it and what potential has yet to be revealed. Sort of like the story of the ugly duckling that transforms into a swan[11]".

Finally, ecologists and design professionals may intentionally create habitat using plants (and perhaps new soil); it is then expected that any desired fauna will colonize the constructed habitats. Palmer et al. (1997, 2010) has called this expectation the "field of dreams" hypothesis; we call this approach "build-it-and-they-will-come." The build-it-and-they-will-come approach is used extensively in rehabilitation, restoration, and regeneration projects (Fraser et al., 2015) and there are many examples of successful restoration projects using this approach in natural areas or areas with relatively light human footprints (e.g., Lister, 2007; but see Palmer, 2009). The build-it-and-they-will-come approach is being extended by design professionals who are creating biodversity and hybrid projects incorporating urban design and restoration ecology; THE SHALLOWS (section 4.3) includes many build-it-and-they-will-come aspects. Many other designed landscapes are exemplars of build-it-and-they-will-come habitats: botanical gardens (e.g., the Lady Bird Johnson Wildflower Center in Texas, USA[12]) and urban nature parks (e.g., Forest Park in Portland, Oregon, USA[13]) often host mutualists fauna in bipartite networks (e.g., plants and their pollinators) or simplified food webs.

5.2. Intentional, Designed Urban Food Webs

We also distinguish three broad approaches to the intentional design of urban food webs. The first occurs through what we call "ecological theory applied through design," and includes the use of foundation species (*sensu* Ellison, 2019) or ecological engineers (*sensu* Jones et al., 1994) to catalyze food-web development. The second, "research partnerships informing design" involves the co-development and use by designers and ecologists of deliberate experiments to enhance designs and create opportunities for embedded research (Felson et al., 2013a; Ellison et al., 2020). The third approach, "creative and aspirational ecological design" includes biocentric design, urban species introductions, and rewilding (Corlett, 2016; Svenning et al., 2016). As these intentional-design approaches likely will be less familiar to ecologists, for each of them we provide an example and elaborate on their values and constraints. These examples reveal how intentionality can inspire design thinkers to envision innovative strategies in urban settings that respond to these constraints

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[9] https://sites.rutgers.edu/urbionet/.

[10] https://www.urban-wildlife.org/.

[11] https://landscapeaustralia.com/articles/interview-dirt-studios-julie-bargmann-1/.

[12] https://www.wildflower.org/.

[13] https://forestparkconservancy.org/forest-park/.

and create novel circumstances geared toward more successful outcomes. In section 6, we identify key challenges that can arise when ecologists, landscape architects, and designers collaborate on intentional-design projects, highlighting that the different discipline-based theoretical concepts of form and function can lead to the parties talking past one another.

5.2.1. Ecological Theory Applied Through Design

Designers have some history working directly with research ecologists (section 3), but one obstacle has been identifying ecologists willing to work with designers on projects that may have limited value as research sites or for conservation (Felson, 2013). Designers frequently subcontract environmental consultants to delineate critical habitats (e.g., wetlands) and secure environmental permits, but they also have a history of working with professional or academic geographers and landscape ecologists (Deming and Swaffield, 2011; Musacchio, 2011). Ecologists are increasingly developing theory in areas that seek to bridge with design and management. Ecological landscaping, for example, takes a systems approach for investigating biotic and abiotic conditions, biodiversity, and ecosystem patterns and processes in urban landscapes (Byrne and Grewal, 2008). Plant scientists are increasingly engaging in critical questions about novel urban plant communities, such as ruderal landscapes (Del Tredici, 2010) and hybrid native-exotic vegetation, and testing ways in which such communities can support a rich native invertebrate fauna that also fosters predatory vertebrates (Hitchmough, 2008)[14]. Urban reconciliation with wildlife has focused on living roofs and green walls (Francis and Lorimer, 2011). Ornithologists working with designers have contributed expertise on addressing bird-friendly urban design and building systems to address particular target species of concern (City of Calgary, 2011; Holloway, 2018). More recently, entomologists, herpetologists, icthyologists, and marine scientists are collaborating with designers on designed rehabilitation and restoration projects (e.g., Ellison et al., 2020). These collaborations are certainly benefiting the design professions while creating new avenues for ecological inquiry.

Oyster-tecture (**Figure 6**) is an example of a design that incorporated ecological ideas and concepts to create an **ecological infrastructure**. The design for *Oyster-tecture* was created by the landscape architecture firm SCAPE for the 2010 "Rising Currents" exhibition at the Museum of Modern Art (Orff, 2010). Configured as an infrastructural armature, *Oyster-tecture* proposed using an ecological engineer to anchor the design. The oyster reefs would simultaneously take up nutrients, remove toxins from New York's Gowanus Canal, physically attenuate waves, and increase habitat around Governor's Island.

The design in this project occurs at the interface and integration of the species lifecycle with the urban systems and fabric. The design reinterprets and restructures the lifecycle of the organism into different zones of the city to perform an ecological function. At the same time, the organism contributes to the aesthetic and form-making of the proposed interventions. The organism's lifecycle is overlaid and nested into the urban infrastructure and remnant ecosystems of the city (Steiner et al., 2013; Orff, 2016). The outcome is a spatially and temporally designed lifecycle attuned to the hydrology and industrial pollution of the city. The oyster would reprogram the harbor geographically, respond to anthropogenic impacts to the harbor, and take advantage of urban planning and technology to bring the oyster as an ecosystem engineer up to the scale necessary to initiate remediation of the harbor ecosystem (Orff, 2010). Although this design has not been built, it contributes to the Gowanus By Design initiative[15].

Like the habitat-based build-it-and-they-will-come approach (section 5.1), *Oyster-tecture* focused on a single species. But unlike a plant-based creation, *Oyster-tecture* had as an explicit goal that an ecological engineer would catalyze the assembly of a stable food web. This design proposal went on to influence multiple subsequent built projects, including SCAPE's Jamaica Bay project and the Rebuild by Design Living Breakwaters project in Staten Island, New York, both of which incorporated food-web considerations (see section 4.3 and Orff, 2016). Although ecological designs such as *Oyster-tecture* seek to leverage ecological processes, they are distinct from conservation or restoration efforts. They accept anthropogenic impacts, stress innovative solutions, and direct attention and resources to cities where the majority of people live and where investments increasingly are being targeted for improving wildlife habitats while providing education about ecology to broader audiences.

5.2.2. Research Partnerships Informing Design

Research partnerships informed the design of the Elliott Bay Seawall in Seattle (Washington, USA), built in 2017. The landscape architecture and urban design firm, James Corner Field Operations, led a large multi-disciplinary team of structural, coastal, storm water, and geotechnical engineers, habitat engineers, and an artist to combine infrastructure with habitat creation as part of a 1,128-m (3,700-ft) public waterfront and seawall replacement (**Figure 7**)[16]. The three primary project goals were to enhance nearshore marine habitat that would support the algae and detritus fed on by the preferred prey of juvenile Pacific salmon (*Oncorhynchus* spp.), re-establish an interrupted migration route from natal streams out to the ocean, and create a high-quality pedestrian promenade (Sawyer et al., 2020). The team built an elevated seafloor and "fish-smart" seawalls (Cordell et al., 2017; Morris et al., 2018) faced with textured substrates. The elevated seafloor included an engineered pocket beach, a saltmarsh eco-bench, an offshore rock-kelp-reef system, and a riparian zone all functioning together as an integrated habitat. The seawalls simulated the physical configuration of natural shallow-water habitat, increasing invertebrate colonization (e.g., barnacles) and abundance of epibenthic invertebrate prey preferred by juvenile salmon (Toft et al., 2013; Coombes et al., 2015; Cordell et al., 2017; Strain et al., 2018). Above this belowground corridor is a pedestrian promenade that includes glass surfaces and perforated

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[14] https://dirt.asla.org/2012/08/22/another-winner-the-london-olympics-landscape/.

[15] https://www.gowanusbydesign.org.

[16] https://www.asla.org/2017awards/320768.html.

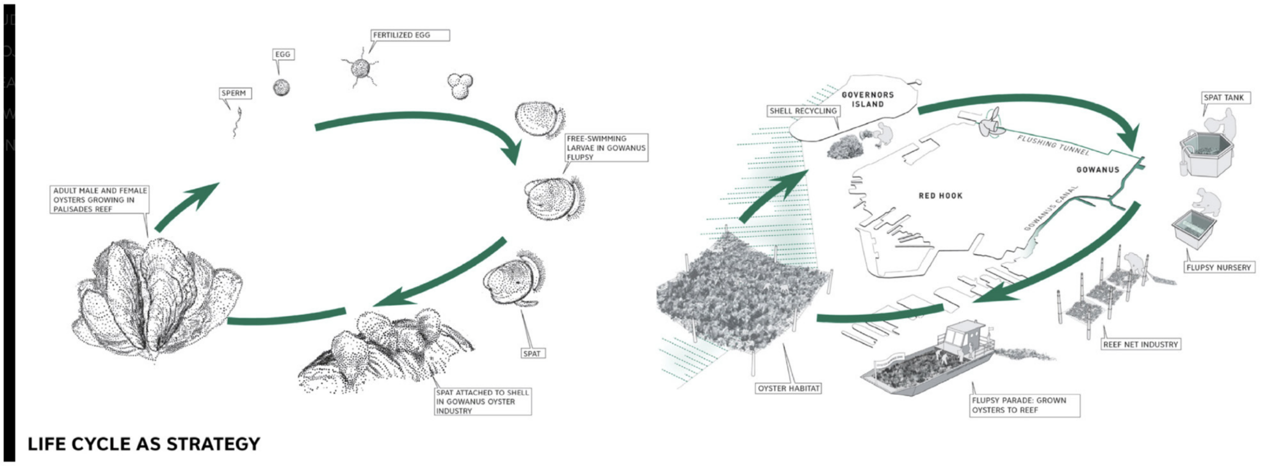

FIGURE 6 | The proposed design for *Oyster-Tecture* is an example of Ecological Theory Applied through Design. Illustration courtesy SCAPE/Kate Orff, developed for the Musem of Modern Art's 'Rising Currents' exhibit (2009).

panels to allow light to penetrate and support growth of algae and other plants. The promenade also serves as an integrated public realm that engages visitors in understanding marine food webs. This is also an example of an innovative "broader impact" that ecologists doing research on designed food webs could realize through collaborating with designers.

5.2.3. Creative and Aspirational Ecological Design

Alongside the two previous nascent approaches to intentional design of food webs, more creative and aspirational designs are emerging that build on investigations of roles that humans and urbanization play in eco-evolutionary dynamics (Alberti, 2015; Lambert and Donihue, 2020). The recognition that strong selection occurs in urban areas in response to rapid environmental changes, and that ecosystem services, stability, and species interactions are affected on short time scales raise questions about evolutionary consequences and dynamics in designed projects. Comprehensive designs of novel ecosystems are one avenue to addressing these (and other) constraints and focus on the likelihood of success of alterations to habitats or landscapes (Hobbs et al., 2014; Kareiva et al., 2018).

An example of a collaborative creative and aspirational project is the East River Marsh Planter—developed by Ken Smith Landscape Architect with the first author—that was proposed for installation along the bulkhead at 34th Street in New York City (**Figure 8**; Amidon, 2006). This project was built, but in a different configuration[17].

The original design was intended to be a food-web catalyst and illustrated a design approach to influencing multiple trophic structures. Its key design concept is the insertion of an artificial, technologically managed "interlayer": pumps bring in water with organisms from low trophic levels that are inserted into a series of perforated "planters"—elevated boxes planted with native saltmarsh cordgrass (*Spartina alterniflora*). Populations of these organisms are grown and then reintroduced in an *in situ* "production line" to foster food-web assembly in the East River. In an engaging, educational display of ecological dynamics, the pumps would flood the planters every morning and evening, co-incident with people's daily commute.

The design of the East River Marsh Planter aesthetically combines ecological principles and technology. The planter was intended to use engineered soils including hydrogel—a polymer that holds water—to act as an artificial mud layer and allay concerns about desiccation in an elevated artificial marsh. The macroinvertebrate larvae and minerals pumped from the East River supplied nutrient-rich water for the planters while seeding them with species occupying a low trophic level, yet one higher than the plants ("resources") where build-it-and-they-will-come projects usually stop. The nutrient-rich water and the associated invertebrate larvae supported a set of intermediate-level consumers, including deposit-feeding fiddler crabs (*Uca* spp.) and salt-marsh snails (*Melampus bidentatus*) located in cells within the planters. A freshwater emitter would periodically flush out detritus and plankton from the planters through perforations in their bases, providing nutrient-rich food for higher-level consumers, including blue crab (*Callinectes sapidus*), flounder (*Pseudopleuronectes americanus*), and brown bullhead (*Ameiurus nebulosus*). As mesocosms, the planters would enable ecological studies of food-web development and evaluation of the ecological value of habitat design and restoration strategies. Finally, the ecological mesocosms would be situated along the routes of >30,000 daily commuters, presenting ecosystem services to broader audiences in an aesthetically pleasing way.

The marsh planter is an example of a **designer ecosystem** that attempts to balance site constraints with human social and economic needs. The designers recognized that site conditions and pressures on organisms have changed, and they sought to maximize ecosystem services in a human-modified environment.

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[17] https://www.architecturalrecord.com/articles/14187-pier-35-by-shop-architects-and-ken-smith-workshop.

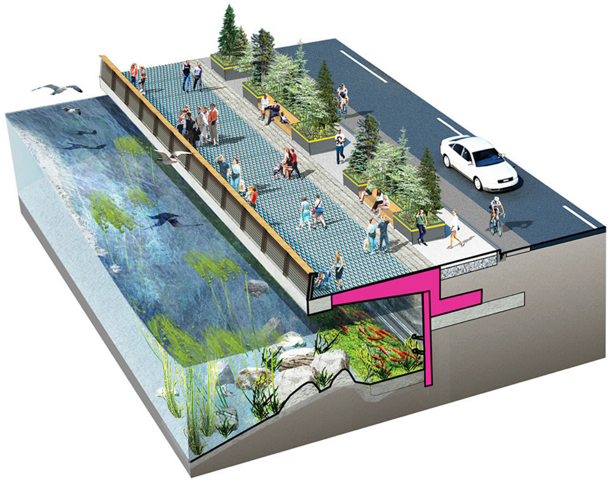

FIGURE 7 | The Elliott Bay Seawall is an example of a Research Partnership Informing Design project. It includes an elevated seafloor and bench re-established a long-interrupted salmon migration route. The light penetration and water depth conditions mimic a natural shoreline and provide a refuge from predation. The precast concrete seawall face panels and habitat shelves and cantilevered light-penetrating system panels support the salmon migratory path. Overall, the restoration activities have increased substrate complexity and epibenthic invertebrate taxa richness. The project team for the seawall includes WSP (formerly Parsons) (Prime Firm, Program Manager, Design Manager, Permitting, Environmental Documentation, Procurement, Utilities Engineering, Geotechnical Engineering, Landscape Architecture, and Structural Engineering), Magnusson Klemencic Associates (Lead Engineer for Public Realm and Habitat Design Integration), Perteet and SvR (Civil and Drainage Engineering), Moffatt and Nichol (Coastal Engineering), COWI North America and Exeltech Consulting, Inc. (Structural Engineering), Shannon and Wilson, Inc. (Geotechnical Engineering), Hart Crowser (Habitat Engineering), Harrison Design Landscape Architecture (Local Landscape Architect), Haddad/Drugan (Artist), and Jacobs (Construction Management). Image used with permission from James Corner Field Operations.

The planters themselves are an integral part of a new ferry terminal; they mark the entrance to the terminal (economic service) and the design provides educational opportunities to daily commuters (social service). Although designer ecosystems and classically managed systems, such as Jamaica Bay (section 4) each blend human social, political, and economic demands with technology, creativity, and innovation to support multiple species and provide ecosystem services, designer ecosystems are much more flexible in the species that are used (e.g., not necessarily native) and are more likely to include regular human interventions to enhance ecosystem services and facilitate system longevity.

Some rewilding projects also are creative and aspirational. Rewilding takes re-introductions to a new level by replacing locally extinct species with either translocated populations of the same species or ecological "proxies" that can replace the ecosystem services of extinct species ("trophic rewilding," sensu Svenning et al., 2016). Interest in rewilding is growing rapidly, but is still much debated (Donlan et al., 2005) and criticized by ecologists (Caro, 2007; Oliveira-Santos and Fernandez, 2010; Lorimer et al., 2015). Yet, rewilding projects are moving forward.

Trophic rewilding privileges restoring trophic interactions, not necessarily involving original fauna (Svenning et al., 2016). Most examples of trophic rewilding involve reintroduction of missing herbivores (reviewed by Svenning et al., 2016). These can be succeeded by sequential reintroductions of extirpated animal species, including carnivores, that foster restoration of ecological interactions with the remaining flora and fauna (Fernandez et al., 2017). The reintroduction of wolves into Yellowstone National Park is the best example of re-establishment of a trophic cascade following reintroduction ("rewilding") of a top predator (Dobson, 2014). An example of urban rewilding is the reintroduction of two seed dispersers, the red-humped agouti (*Dasyprocta leporina*) and the howler monkey (*Alouatta guariba*) into the Tijuca National Park[18], a 3,953-ha forest fragment located within the city limits of Rio de Janeiro, Brazil (Fernandez et al., 2017). Because the Tijuca forest is surrounded by the city, re-introduced fauna cannot disperse to other forest fragments (Fernandez et al., 2017). Twenty-five dung-beetle species have been observed using the howler-monkey dung (i.e., a trophic effect) and there is the hope that the agoutis and monkeys will foster forest regeneration (Fernandez et al., 2017).

6. CHALLENGES AND OPPORTUNITIES IN DESIGNING (FOR) URBAN FOOD WEBS

The examples in sections 4 and 5 are intended to suggest ways forward for bringing ecologists and design professionals together in collaborative efforts to deliberately design and study habitats or landscapes for urban food webs. There are many theoretical, practical, and cultural obstacles to designing, building, and maintaining urban food webs and building collaborations between ecologists and design professionals. In this section we identify the key challenges and opportunities to address them.

6.1. A New Frontier for Research and Collaboration

Recognizing existing remnant food webs, cultivating them through architectural design, and exploring ways to construct novel urban food webs will require new research and practical development. Ecologists will need to identify how human activities have already altered food webs (e.g., Christie et al., 2010). Design professionals will need to work with ecologists to translate findings from ecological theory into projects that can be built and maintained efficiently and cost-effectively. Designers are particularly keen to apply ecological theory at spatial and temporal scales relevant to people living in cities (Jabareen, 2013; Mosbach, 2013; Steiner et al., 2013). Designers also bring aesthetic considerations and methods of translating form into (ecological) functions and services (and *vice-versa*). Ecologists and designers alike need to understand each other's vocabulary and aesthetics and work outside their individual comfort zones to collaborate effectively on the design, construction, and monitoring of, and research into, urban food webs (see section 6.4).

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[18] https://www.icmbio.gov.br/parnatijuca/.

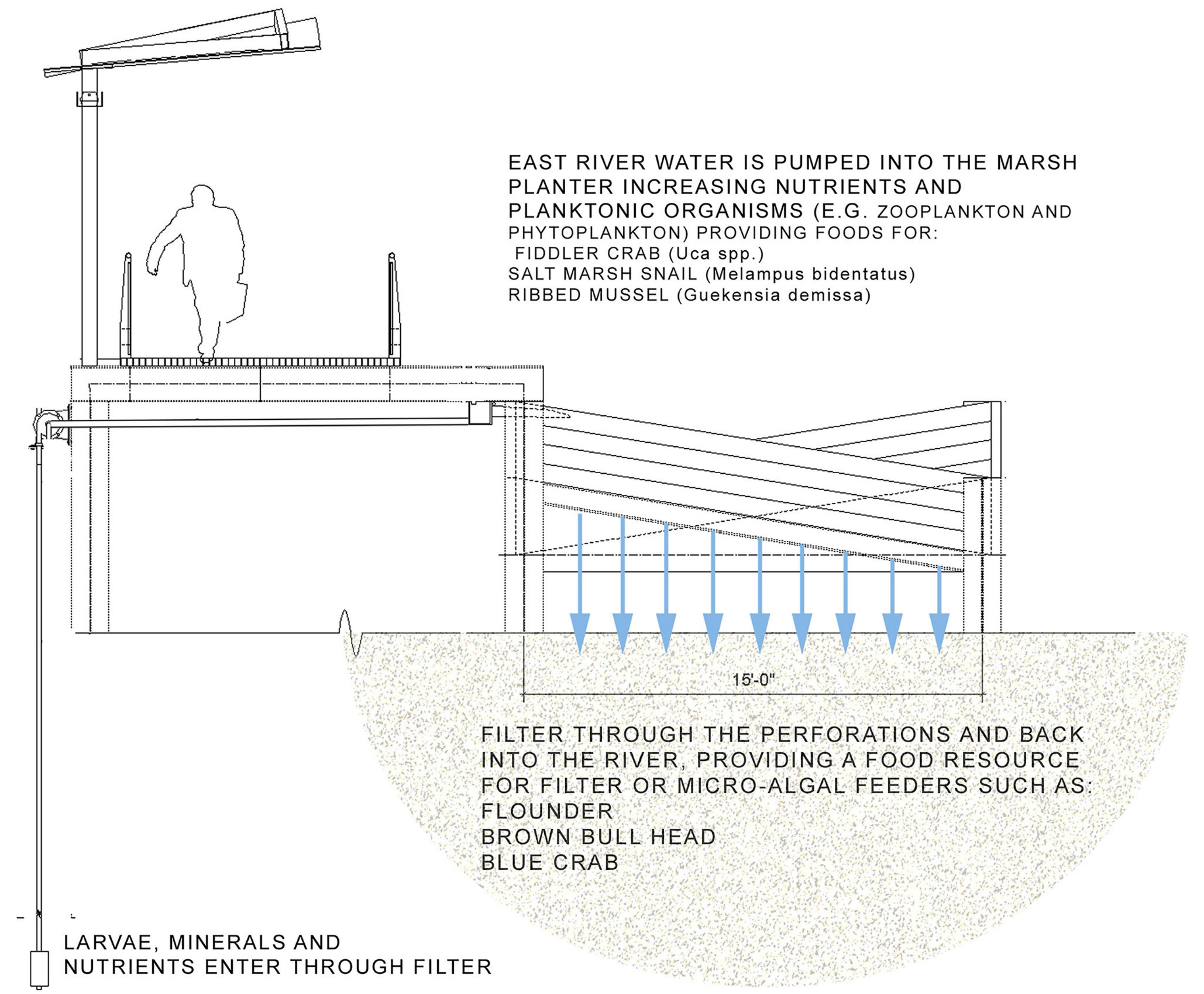

FIGURE 8 | The proposed design for the East River Marsh Planter is an example of Creative and Aspirational Ecological Design. Design by Ken Smith Landscape Architect; drawing by Alexander Felson and used with permission.

6.2. Ecological Considerations

6.2.1. Biodiversity and Habitats Are Parts of, But Not a Proxy for, Urban Food Webs

Individual species are parts of food webs (**Figures 1A–C**) and recent efforts to identify and enhance biodiversity in cities (e.g., Kowarik, 2011; Uchida et al., 2021) are a first step in its appreciation not only by design professionals but also by the broader public. Yet, knowing there are, for example, many species of ants on Broadway (Pećarević et al., 2010) or that there are unique scaling relationships for urban biodiversity (Uchida et al., 2021) does not describe functional roles of species in a food web. Urban food webs also may have lower species diversity and occur in weakly or unconnected habitat patches (e.g., Start et al., 2020). Such patches in designed and maintained landscapes may function as habitat sinks or ecological traps for species in cities, but such sinks rely on ex-urban populations for replenishment. Ecologists need to identify combinations of species in multiple trophic levels that can survive and function in the fragmented landscapes of cities (e.g., Start et al., 2020) as a prelude to designing habitats that could support urban food webs (Roudavski, 2020).

(Re)building or restoring urban food webs also will require habitat rehabilitation co-incident with, or followed by, management of introduced or undesired species. Such efforts are already underway in many parts of the world. For example, the blue-banded bee (*Amegilla cingulata*) in Melbourne, Australia, is being actively supported using planted habitats, but the designers and managers are already considering how to construct and maintain metanetworks of bee subpopulations (Mata et al., 2019). Although their approach does not yet include additional trophic dynamics (e.g., introducing neon cuckoo bees or bird species that prey on blue-banded bees), this build-it-and-they-will-come approach recognizes that food webs are often anchored by a few key species. Establishing and managing urban food webs will require additional research and analysis on how specific species interact with the rest of the altered regional and urban systems.

6.2.2. Urban Food Webs Integrate Human and Ecological Concerns

All of the documentation and analysis of non-urban food webs is unlikely to be captured in cities where human behavior and urbanization reshape the networks of species interactions and food webs can unravel and disassemble (McCann, 2012; Start et al., 2020). Urban environments will be influenced and constrained by geometry and fragmentation of the urban habitat matrix. Such influences, constraints, and limitations range from effects of urban forms (buildings, street layouts, and landscaping) on local microclimates to hydrological alteration and interruption of migration corridors at larger spatiotemporal scales. The sizes, forms, and networks of urban parks, waterways, gardens, lawns, and street tree populations have interacting impacts on food-web structure, dynamics, and the services they provide. Remnant urban food webs are species-poor, sparse patches surrounded by inhospitable environments (Start et al., 2020).

Urban food webs occur in human-dominated landscapes and people often are apex predators (as in Jamaica Bay). Yet, there are few examples outside of fin- and shellfisheries (e.g., Pauly et al., 2000; Moreno, 2001; de Boer and Prins, 2002; Reitz, 2004; Essington et al., 2006; Braje et al., 2007; Graham et al., 2017) of ecological studies of "natural" food webs that include people as predators, even though people routinely kill (although rarely eat) herbivores and predators in cities. Ecologists need to study and understand how human behaviors and desires (e.g., controlling or eliminating insects, rodents, or other "pests") affect urban food-web dynamics and ecosystem services. Results of such studies, perhaps included in experiments within designed landscapes, could help identify critical factors that define or constrain the management of urban food webs. Ecologists also need to educate and communicate to homeowners and park managers that different landscaping practices will have different effects on food webs. For example, embracing snags and downed wood as part of our parklands and introducing artificial nesting environments may sustain food webs whereas raking, removing, and composting leaves or chipping and burning coarse woody debris removes critical habitat and trophic connections (e.g., Hoyle et al., 2017).

6.2.3. Temporal Dynamics May Be Unappreciated or Arrested in Urban Food Webs

Like plant ecologists studying succession, animal ecologists have long appreciated that food webs assemble and change through time as individual species progress through their life-histories, some species go (locally) extinct, or other species colonize habitats that are already occupied (**Figure 1D**; MacArthur and Wilson, 1967; Drake, 1990a,b, 1991; Ellison and Gotelli, 2021). Ironically, most ecological research on food webs has emphasized their static description (see reviews in McCann, 2012; Ellison and Gotelli, 2021), but that does not negate the fact that food webs are dynamic, constantly changing entities (e.g., de Ruiter et al., 2005; Pilosof et al., 2017). Ironically, many restoration projects (regardless of defined baseline) seek to manage conditions to maintain a specific, temporally invariant species composition.

Similarly, a specific and temporally invariant species composition may be desired by city planners, designers, or clients for designed habitats and urban food webs, Maintaining these habitats or restored sites to preserve a particular design or assemblage of species is analogous to arresting ecological succession (Del Tredici, 2007). Design and restoration goals often contrast with ecological reality; the former's emphasis on planning, cultivation, neatness, rigidity, and status contrasts with the latter's emphasis on spontaneity, wildness, messiness, adaptive, and flux and change (Del Tredici, 2007). Design practitioners similarly focus on **maintenance**, whereas ecologists focus on management. For designers, operations and maintenance is an ongoing, underfunded challenge, whereas for ecologists, management is an expected, fundamental component of habitat or trophic restoration. A central challenge for ecologists working with designers is to identify and co-design resilient urban ecosystems while promoting effective and adaptable experimental management strategies appropriate for urban sites and human subjects.

6.2.4. Urban Food Webs as Experimental Systems

Restoration ecologists have attempted with mixed results to reintroduce (or rewild) species (see section 5.2.3; and Caro, 2007; Svenning et al., 2016; Perino et al., 2019). Some reintroductions seek to restore **ecosystem functions** or services (e.g., the reintroduction of the golden lion tamarin, *Leontopithecus rosalia*, in northern Rio de Janeiro; Kierulff et al., 2012; IUCN/SSC, 2013). Other species may be relocated or introduced into new locations where they can survive. For example, in the construction of the Railyard Park in Santa Fe, a population of prairie dogs that lived on the railyard was deemed a problem for the planned park; the prairie dogs were relocated to a new location where they would not come into conflict with people (Chacón, 2013). The deliberate introduction of species into urban natural areas has a long history (e.g., starlings and sparrows [Wing, 1943]; arthropods for biological control [Vickery and Kevan, 1983; Aslan et al., 2014; Evans, 2016]) but these are not considered as reintroductions or rewilding. Ecologists have a new research opportunity to consider introductions of predators into urban environments as a way to study (re)introductions of species into food webs in the "unnatural" systems represented by cities where ecological "disasters" or trophic cascades may be of less concern than they might be in more "natural" (i.e., non-urban) systems (e.g., Aslan et al., 2014; Egerer and Kowarik, 2020).

6.3. Design Considerations

6.3.1. The Need for Maintaining a Messy Aesthetic

Aesthetic and functional goals are important considerations for designers. The relationship between aesthetics and ecology as it relates to landscape design, planning, and management can be well-aligned or in opposition to one another. Aesthetic goals of, for example, scenic beauty or appreciation of nature may map closely or clash with indicators of ecological quality (Gobster et al., 2007). Embracing "messy order" as an aesthetic that allows some geometry and refinement of spaces while allowing for organic growth to occur will help create habitat to support urban food webs (Nassauer, 1995). Yet designed landscapes, even messy ones, need to be maintained in their desired states (see also section 6.2.3). Parks departments are always challenged to keep up with maintenance and many designers posit maintenance issues as the major factor leading to failure of landscape designs (Aronson et al., 2017; Hoyle et al., 2017; Riley et al., 2018). Controlling and managing multiple species is difficult enough in restoration projects (Fraser et al., 2015); keeping food webs in a desired state will be difficult for already overextended maintenance crews. Designers need to create robust systems that have management strategies embedded in the design. This may be achievable by increasing redundancy within urban food webs but ecologists still do not know whether or how often food webs contain functionally replaceable taxa, and which taxa they might be (e.g., Chalcraft and Resetarits, 2003; Petchey and Gaston, 2006; Hagen et al., 2012; Gray et al., 2016). Maintenance staff will need additional training to increase their habitat management expertise.

6.3.2. Using Design to Change Human Perceptions of Food Webs

People rarely think about predation, and when they do, perceptions are generally negative (e.g., Archer-Lean et al., 2015; Bencin et al., 2016; Miranda et al., 2016; Eshete et al., 2018). Including food webs in cities will require overcoming these negative perceptions. Zoos are well-known examples of designed environments that prioritize persistent form over ecological dynamics and function. Visitors witness single or interacting species in artificial, static conditions that are modeled to look "natural" in immersive exhibits but where actual predator-prey interactions cannot occur (**Figure 9**). What are reasonable designs that could induce favorable perceptions of predation and inculcate an aesthetic desire for stable food webs (Roudavski, 2020)? Will these designs align aesthetics with ecological function (Gobster et al., 2007)? Designers must also consider the challenges of designing and managing urban food webs and the "cans of worms" that could be unleashed with proposals that use animals or trophic structures as material components (Roudavski, 2020).

6.4. Urban Food Webs Require Finding Common Ground That Recognizes and Resolves Differences in Language and Vocabulary

The aforementioned technical challenges of designing (for) urban food webs are complicated by the differences between ecologists and designers in aesthetic priorities, desired ecosystem services, and perspectives about how the world "works" (Gobster et al., 2007). In short, ecologists and designers speak very different languages (Johnson et al., 2002; Ellison and Buckley Borden, 2021). In our experience, the crux of these differences lies in the different meanings, values, and cause-and-effect relationships between form and function that are used and understood by ecologists and designers.

Our intent here is not to review the 150 years of ecological research on food webs (reviewed by McCann, 2012; Ellison and Gotelli, 2021) or the similarly long debate among architects and designers on the relationship between form and function. Rather, we first identify how ecologists define and describe the form of food webs and measure their function. We then turn to the aesthetic and intellectual challenges designers face when using ecological knowledge to design landscapes intended to support specific ecological functions (e.g., **Figures 1D,E**; Mozingo, 1998; Lister, 2007; Musacchio, 2011; Nassauer, 2012; Forman, 2014, 2016; Orff, 2016; Parris et al., 2018). We emphasize that designers' more fluid perspectives on the relationship between form and function differs strongly from ecologists' generally inflexible expectation that form derives from function (for a notable exception, see Gould and Lewontin, 1979).

6.4.1. Form and Function in Ecological Food Webs

Ecologists describe the "form" of a food web (which ecologists also call its "structure") with illustrations of its network topology (i.e., food-web diagrams: **Figures 1A–C**). Structural properties of food webs (and of networks in general), such as the number of species ("nodes"), the number of trophic levels, the sizes or traits of individual species, and the degree to which species are linked to one another (via "edges"), are system-independent metrics that ecologists use to compare forms of different food webs and create general theories of food-web structure (Lau et al., 2017). The edges of food-web diagrams illustrate predator-prey interactions or the flow of energy and nutrients between species, and within and across trophic levels. These edges define the ecological "functions" or "services" of a food web: processes, such as production (e.g., Baiser et al., 2013), consumption or predation (Paine, 1966; Valls et al., 2015), and energy flow (e.g., Borrett, 2014).

Ecologists generally assume that the form (structure or topology) of a food web is a direct consequence of its ecological functions (predator-prey relationships and energy flow between individual species: **Figures 1A–C** and the set of images from lower left to upper right in **Figure 2**). This idea parallels Sullivan's 19th-century maxim for designers that "form ever follows function" (Sullivan, 1896). However, an ecologist's "form" is not related to the physical form of a designer's habitat or landscape, but only defines the topology of the food web itself. In support of this hypothesized "one-way street" on which function defines form, ecological studies of food webs that emphasize function (bottom-up or top-down control or fluxes of energy and nutrients) generally pay little attention to the underlying topology (form) of food webs beyond identifying particular functional groups of organisms (e.g., primary producers, herbivores, carnivores, or decomposers).

6.4.2. Form and Function in Landscape Architecture and Design

Landscape architects have gone through multiple cycles of signifying "natural" ecological relationships in designed

FIGURE 9 | Zoo enclosures are artificial, theatrical spaces where animals are put on display. Only the form of the food webs to which the animal belongs is illustrated; its sole function is to represent nature. The Hagenbeck Tierpark Zoo near Hamburg, Germany (constructed 1907) was the first zoo to replace cages with open enclosures and moats and berms to manage animal. The African exhibits illustrate potentially interacting species whose functions in their ecological community are entirely lost. People standing at the primary viewpoint (indicated in the bottom panel) see an assemblage of species that each occupy their own enclosure. This is an example of form-driven aesthetics created by perspectival viewpoints. The theatrical arrangement follows functional considerations related to vantage points and perspective (top and middle panels). Image sources: **(Top)** Carl Hagenbeck's Tierpark, Hamburg. 1935. Die Entwicklung des Hauses Carl Hegenbeck (https://www.davidrumsey.com/luna/servlet/detail/RUMSEY~8~1~299571~90070511:Text--Die-Entwicklung-des-Hauses-Ca; CC BY-NC-SA 3.0); Diagram by Alexander Felson; **(Middle)** Plan, Carl Hagenbecks Tierpark, Altona/Stellingen/Hamburg, (https://www.davidrumsey.com/luna/servlet/detail/RUMSEY~8~1~299572~90070510:Plan,-Carl-Hagenbecks-Tierpark; CC BY-NC-SA 3.0); **(Bottom)** Diagram by Alexander Felson and used with permission.

landscapes as "forms" (**Figure 10**; Treib, 1989; Girot, 2016). For example, in the 17th-century French style of André Le Nôtre and others, form (as formal composition), not function, was the fundamental driver of landscape design (form; **Figure 10B**). This evolved in the 18th-century English Picturesque style into complicated overlays of formal composition through the design of paths and vantages points, with scenes imitating nature (**Figure 10C**). In a more "form-follows-function" approach, 20th-century Modernist architects stripped buildings of ornament and other formal devices while embracing and foregrounding the functional aspects of buildings (**Figures 10D,E**). Landscape architecture reinterpreted the modernist framework with complex geometries and imposed controls that allowed designers to use plants and soil as materials, and permitted organic growth and messiness (section 6.3.1; Nassauer, 2012).

Since the simultaneous publication of *Design with Nature* (McHarg, 1969) and the emergence of a broad environmental consciousness, landscape architects increasingly embraced environmental ("ecological") systems and strategies. In the last 50 years, designers have alternately moved toward and away from this embrace as they have struggled to define their role and determine or prioritize methods for integrating ecology with design (Spirn, 2000; Johnson et al., 2002). This back-and-forth is illustrated by comparing San Jose's Guadalupe Park (1991–2005)[19] with the rooftop garden atop New York's Museum of Modern Art (2002–2005)[20]. Guadalupe Park redesigned an urban river floodplain into a park (form) that provided flood control (ecosystem service). Although many areas of the park are designed to flood, the area of the former floodplain that rarely floods today includes land forms constructed to look like a miniature field of drumlins or a ski slalom. These landforms have little to no ecological function but they aesthetically express a glacial history and the flow of water. In contrast, MOMA's rooftop garden is entirely artificial. This form has no ecological function, but instead reflects engineering constraints (the load the room could bear) and a strong and particular aesthetic: camouflage from surrounding buildings.

In 21st-century practice, the intertwined interrelationships between form and function are deeply embedded in design but form is rarely, if ever, derived directly from function. Although designers seeking certain functions may explore what types of form produce them ("function follows form"), if a particular form is desired or specified by a client, a designer may seek the functions to produce it ("form follows function"). In practice, designers deal with both form and function. For example, **landscape urbanism** (Waldheim, 2016) and **parametric design** (Cantrell and Mekies, 2018) claim to explore ecological processes or applications and their relationships to form, representation, and materiality. Both focus on surfaces, materiality, and aesthetics, and provide interacting analysis of these through process diagrams and graphical layouts that ecologists would find challenging to interpret or in which to connect form and function (**Figure 1D**).

6.4.3. Toward a Common Language

The intentional design of urban food webs links ecological theory with the performative and functional properties of built environments. In urban planning and landscape architecture, form, aesthetics, and desired services (functions) may drive the design, but only a subset of organisms can be cultivated in, or otherwise inhabit, a designed landscape. These "simplified" food webs can provide new research opportunities for ecologists, such as testing hypotheses about assembly and temporal dynamics of food webs and the ecosystem services they provide (sections 6.2.3, 6.2.4). The design process itself requires intentionality in the formulation of a desired reality (i.e., creation of a designed habitat or landscape; **Figure 10**). Designers working alone may make uninformed assumptions about individual species or ecological processes. Designers and ecologists working together with a common language will improve one another's ability to incorporate complicated challenges and resolve inconsistencies or conflicts among design elements in urban ecosystems (Felson et al., 2013a). Opportunities to intentionally shape urban food webs arise from the recognition that populations of different organisms may respond to different cues and configurations within newly structured conditions in a designed, urbanized environment. If food-web form can be coupled directly to its functionality (e.g., Ulanowicz et al., 2014), food webs will be more appreciated because of their utility to society. If such services can persist costs-effectively over the lifetime of a project, they could prove attractive to the clients who might have to pay an initial premium to incorporate a designed food web into a project. For example, identification of foundation or keystone species within food webs (or sub-webs) that could yield desired services (Baiser et al., 2013) could be a logical and reasonably-priced starting point for designing a food web into a particular project.

6.5. The Importance of Human Intention in Urban Food Webs

People add aesthetic and functional criteria that influence the design of landscapes and will apply similar criteria to designing (for) food webs. Standard design practice introduces vegetation. Although food webs often follow, any that do are essentially unintentional (via the "build it and they will come" approach; section 5.1). In contrast, deliberately designing food webs using species introductions or other bio-manipulations of food webs (e.g., fostering trophic cascades or lengthening food chains), translocations, or other direct interventions with fauna remains mostly untested, especially in urban landscapes. Permitting and regulatory constraints and public concerns about managing or manipulating wildlife remain formidable hurdles. Urban food webs are unlikely to be self-sufficient and will require regular maintenance, including resource inputs and re-introductions, to preserve and sustain their intended forms and functions. To ensure that designed urban food webs meet the demands of clients and public users, they must be monitored and, if contracts and designs permit, managed, augmented, or adapted. By working within design constraints, including ongoing designed experiments, and taking on central roles in managing these systems, ecologists may be able to direct, accelerate, or slow the process of food-web assembly and development.

Human intention also will distinguish designed urban food webs from those embedded in classical ecological restoration projects. The former should provide services (primarily utilitarian, but also aesthetic) for people and urban wildlife, whereas the latter usually are intended to recreate a Prelapsarian "nature" that pre-dates European colonization or does not

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[19] http://www.hargreaves.com/work/guadalupe-river-park/.

[20] http://www.kensmithworkshop.com/moma-roof.html.

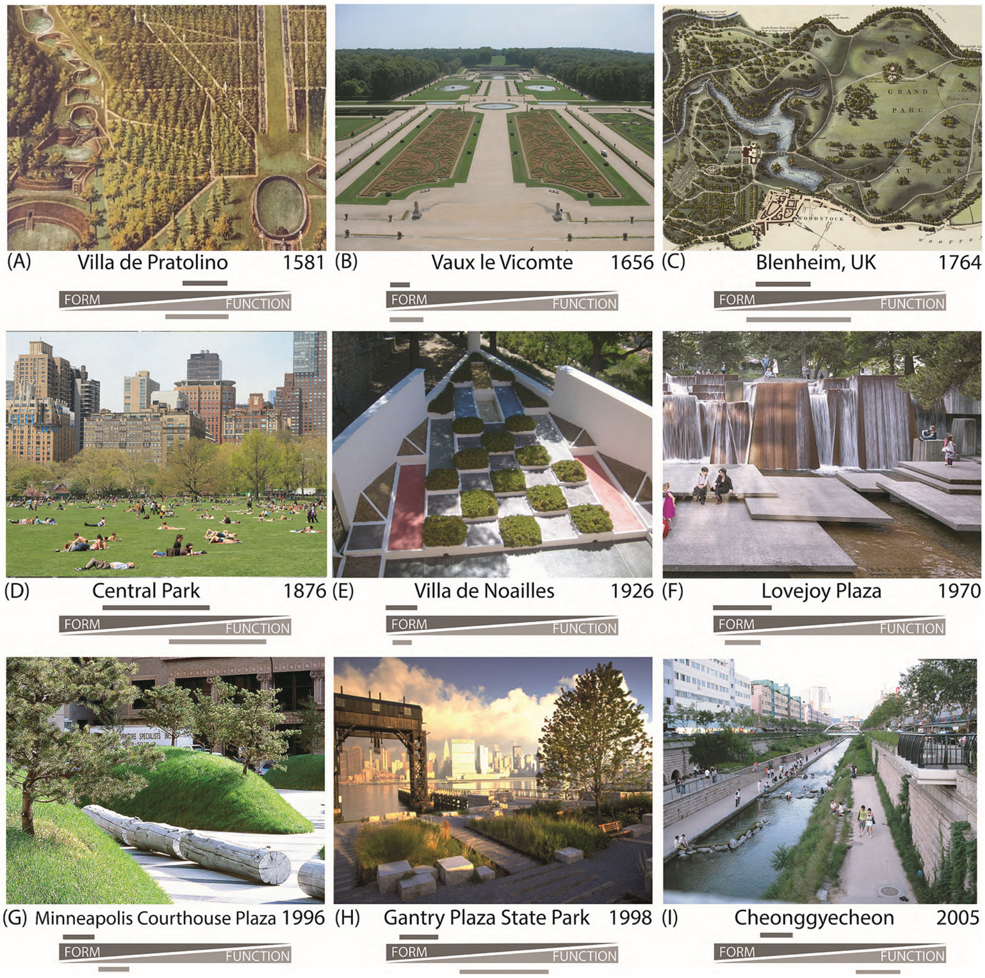

FIGURE 10 | Representative shifts in the attitude and aesthetics of form and function in landscape architecture (images) and the relative importance of form and function from ecological and design standpoints ("sliders" below the images). The color photographs illustrate particular designed landscapes arranged in chronological order. The "sliders" underneath them reflect the relative emphasis of form (dark gray) and function (light gray); the bars above and below the sliders reflect the qualitative magnitude (Form: left to right = large to small; Function: right to left = small to large) and apparent range (length of the bar). **(A)** Abstraction and form influenced a managed forest and waterway converted into a series of pools in a partially constructed natural fountain in the Italian garden at the Villa di Pratolino. Functions, such as direct path connections and flow of water influenced the aesthetically-driven form of the large forested area and waterway. **(B)** André le Nôtre's French garden, Vaux-le-Vicomte, abstracted nature and carved geometric forms onto the landscape using vegetation, water, and paths. The symmetric landscapes represented power and created a sense of theater; ecological function was irrelevant. **(C)** The extensive landscaped park, woodlands, and formal gardens of the English Picturesque Blenheim Palace (Oxfordshire, England) included clumps of forest arranged around large sinuous water bodies across hills to create a series of naturalist views. The idea of nature and organic form was crafted through extensive grading and manipulation to imitate nature visually rather than functionally. **(D)** Frederick Law Olmstead's Central Park in New York imported the concepts from the English garden into an urban area and initiated large park designs across US cities. The park introduced the picturesque and naturalistic style. **(E)** In the Garden at Villa de Noailles, Gabriel Guevrekian created a highly geometric and artificially-conceived landscape influenced by Cubist painting. The project was, in part, a critical response to the Modernist assertion that form following function should drive design. **(F)** The Lovejoy Plaza (formerly Ira Keller Fountain) by Lawrence Halprin Associates with Angela Danadjieva is part of the Portland Open Space Sequence. The project introduced a metaphorical mountain cascade that falls through a series of fissured concrete geometric forms to create crags and eddies as part of a waterfall and adjacent riparian zone. The forms have been aestheticized and turned into stepping and seating platforms. The sound of the water and the cascading pattern and microclimates mimic the natural mountain cascade. **(G)** Martha Schwartz's Minneapolis Courthouse Plaza created a highly orchestrated and form-driven metaphorical landscape of drumlins and cut logs to represent, respectively, the memory of geological forms and the state's history of timbering. **(H)** Gantry Plaza State Park by Thomas Balsey Associates reclaimed and remediated an industrial waterfront. This project used an organic aesthetic organized around industrial ruins to create both ecosystem services and programmatic values. **(I)** Cheonggyecheon is a public recreational retrofit of a former highway infrastructure. The 10.9-kilometer-long (6.8 mi) urban renewal project is built where a stream once flowed. Image sources: **(A)** Villa Pratolino (https://commons.wikimedia.org/wiki/File:Pratolino_utens.jpg; public domain); **(B)** Esther Westerveld (https://commons.wikimedia.org/wiki/File:Kasteel_van_Vaux-le-Vicomte_-_Maincy_06.jpg; CC-BY-2.0); **(C)** British Library (https://en.wikipedia.org/wiki/File:A_new_and_accurate_plan_of_Blenheim_Palace_-_L'Art_de_Cr%C3%A9er_les_Jardins_(1835),_pl._1_-_BL.jpg; public domain); **(D)** Ingfbruno (https://commons.wikimedia.org/wiki/File:3015-Central_Park-Sheep_Meadow.JPG; CC-BY-SA-3.0); **(E)** SiefkinDR (https://commons.wikimedia.org/wiki/File:Cubist_Garden_Villa_de_Noailles_Hyeres.JPG; CC-BY-SA-3.0); **(F)** Hagar66 (https://commons.wikimedia.org/wiki/File:Portland-Ira_Keller_Fountain.jpg; CC-BY-SA-3.0); **(G)** City of Minneapolis (https://msp.world/minneapolis-courthouse-plaza-minneapolis-mn-usa/; public domain); **(H)** https://www.newyork-architects.com/en/thomas-balsley-associates-new-york/project/gantry-plaza-state-park#image-10; used with permission; **(I)** madmarv00 (https://commons.wikimedia.org/wiki/File:Korea-Seoul-Cheonggyecheon-01.jpg; CC-BY-2.0).

include humans at all (Jordan and Lubick, 2011). As urban ecological restoration projects become more common (e.g., Ingram, 2008; Clarkson and Kirby, 2016), they increasingly are using different baselines, creating novel ecosystems, and anticipating and grappling with both rapid evolutionary dynamics and human behavior in their design and execution (Alberti, 2015; Lambert and Donihue, 2020). Whereas successful restoration projects ideally become self-maintaining, many require extensive inputs or ongoing maintenance. For designed urban food webs, management and maintenance will be required. Building some capacity for self-reassembly or succession into urban food webs may reduce maintenance, a key design-criterion for successful and sustainable urban landscapes.

Finally, we note that intentionality in the process of designing food webs differs from that of species re-introductions, including re-population of historic ranges and rewilding of sites (Donlan et al., 2005; Caro, 2007; Lorimer et al., 2015; Corlett, 2016; Svenning et al., 2016; Felson and Dugapolski, 2017; Perino et al., 2019). For example, re-introduction of an endangered species requires an RFP, a scope of work, permitting, and site evaluation long before species are re-introduced and monitored for many years thereafter. With urban food webs, design and construction considerations are as crucial as the evaluation and permitting, but the latter are considered rarely, if at all. Given the critical role that people play in supporting, residing in, and learning from our urban habitats and ecological areas, coupling the process of food-web creation with the design process (Felson et al., 2013b) and exploring form and function of urban food webs should be seen as opportunities to study and achieve greater ecological reconciliation and resilience (Felson, 2013).

7. CONCLUSION

As landscape architects are expanding their role as designers and builders at site-to-city-to-regional scales, they are looking for research and analysis to guide the form, function, sustainability, and resiliency of their projects. Even though "biodiversity planning" may be part of the design process, such planning rarely extends beyond different kinds of plants or their pollinators; complete, or at least stable food webs rarely are considered explicitly. Designers increasingly are seeking to develop ecological habitats that embrace ecological theory and foster biological richness and abundance. The opportunity exists for ecologists to work together with designers to move the design process to the next level: designing environments for supporting food webs and designing food webs directly.

To move forward, ecologists and designers will need to reconsider form and function and how these might play out in an urban context in relation to constructed habitats that foster food webs. The predominant focus on function for ecologists contrasts with the varied philosophical interpretations of principles of aesthetics and a more fluid relationship between form and function that continues to emerge in design through technological advances, adaptations of existing concepts, and development of new theoretical frameworks. Even though these disparate stances may make it difficult for ecologists and designers to find common ground, their dual approaches must come together in the intentional design of urban food webs.

Rather than simply providing ecological theory for designers to interpret and translate into practice, ecologists should participate directly in the design process (Felson et al., 2013b) and co-develop strategies based on emerging ecological theory and ongoing research to create novel food webs and relevant design strategies that work for cities (Felson et al., 2013a). These collaborations will help designers expand their own range of aesthetic and ecologically functional spaces. Establishing habitat designs and associated food webs that are responsive to stakeholders will help to inform the intentional design of urban food webs that make ecological processes visible, lead to the development of new types of green spaces, contribute to new aesthetics for parks and recreation, and permit urban land uses that may be more valuable than a particular restored ecosystem or one recreated for one or a few specific ecosystem services provided by food webs (Felson and Pickett, 2005; Gobster et al., 2007; Aronson et al., 2017). Finally, collaborations that lead to successful design, construction, and installation of urban food webs will of necessity expand beyond ecologists and landscape architects to include urban planners, civil engineers, transportation planners, and municipal authorities, among many others. The case study of Jamaica Bay (section 4) and new directions in designed landscapes (section 5) lay out the possibility of a future in which ecologists are integral participants in the design and construction of urban landscapes where ecological research is a central part of the long-term and ongoing management of urban food webs and the landscapes they help create and support.

ETHICS STATEMENT

Written informed consent was obtained from the minor's legal guardian for the publication of the potentially identifiable image in **Figure 2** of this article.

AUTHOR CONTRIBUTIONS

AF and AE contributed equally to the conceptualization, writing, and editing of the manuscript. Both authors contributed to the article and approved the submitted version.

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