

# Supporting the “virtuous cycle” in urban ecosystems: How research can inform plans, policies, and projects that impact urban resilience

**Edited by**

Michele Romolini, Sophie S. Parker, Gregory Blair Pauly  
and Eric M. Wood

**Published in**

Frontiers in Sustainable Cities  
Frontiers in Ecology and Evolution



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ISSN 1664-8714  
ISBN 978-2-8325-3058-0  
DOI 10.3389/978-2-8325-3058-0

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# Supporting the “virtuous cycle” in urban ecosystems: How research can inform plans, policies, and projects that impact urban resilience

## Topic editors

Michele Romolini — Loyola Marymount University, United States

Sophie S. Parker — The Nature Conservancy, United States

Gregory Blair Pauly — Natural History Museum of Los Angeles County, United States

Eric M. Wood — California State University, Los Angeles, United States

## Citation

Romolini, M., Parker, S. S., Pauly, G. B., Wood, E. M., eds. (2023). *Supporting the “virtuous cycle” in urban ecosystems: How research can inform plans, policies, and projects that impact urban resilience*. Lausanne: Frontiers Media SA.  
doi: 10.3389/978-2-8325-3058-0

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EDITED AND REVIEWED BY  
Victor L. Barradas,  
National Autonomous University of  
Mexico, Mexico

\*CORRESPONDENCE  
Michele Romolini  
✉ michele.romolini@lmu.edu

RECEIVED 11 July 2023  
ACCEPTED 28 July 2023  
PUBLISHED 08 August 2023

CITATION  
Romolini M, Parker SS, Pauly GB and Wood EM  
(2023) Editorial: Supporting the “virtuous cycle”  
in urban ecosystems: how research can inform  
plans, policies, and projects that impact urban  
resilience. *Front. Sustain. Cities* 5:1257069.  
doi: 10.3389/frsc.2023.1257069

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# Editorial: Supporting the “virtuous cycle” in urban ecosystems: how research can inform plans, policies, and projects that impact urban resilience

Michele Romolini<sup>1\*</sup>, Sophie S. Parker<sup>2</sup>, Gregory B. Pauly<sup>3</sup> and  
Eric M. Wood<sup>4</sup>

<sup>1</sup>Center for Urban Resilience, Loyola Marymount University, Los Angeles, CA, United States, <sup>2</sup>California Chapter, The Nature Conservancy, Los Angeles, CA, United States, <sup>3</sup>Urban Nature Research Center and Department of Herpetology, Natural History Museum of Los Angeles County, Los Angeles, CA, United States, <sup>4</sup>Department of Biological Sciences, California State University Los Angeles, Los Angeles, CA, United States

## KEYWORDS

social-ecological, green infrastructure, urban biodiversity, collaborative ecosystem management, public-private partnerships, citizen science, community science, virtuous cycle framework

## Editorial on the Research Topic

Supporting the “virtuous cycle” in urban ecosystems: how research can inform plans, policies, and projects that impact urban resilience

Advancing urban resilience goals requires collaboration across sectors, jurisdictions, organizations, and disciplines. It also requires the ability to cultivate resilience across social and ecological scales. An integrated, collaborative approach presents a great opportunity to address complex problems, but also can present great challenges. This Research Topic aims to showcase projects connecting urban social-ecological research and practice, and provide examples of the process and potential benefits and barriers. We were inspired by the Virtuous Cycle Framework: “The aim is to create a virtuous cycle that will be the engine for continued accrual of the benefits to both people and nature, by mainstreaming conservation so it becomes a part of and product of business as usual” (Morrison, 2015, p. 14). Specifically, the Virtuous Cycle Framework envisions a system in which an intervention aimed to improve the diversity and resiliency of a given place catalyzes a positive feedback loop by providing benefits from nature (e.g., ecosystem services) to people, who are then mobilized to impact policies and/or practices to improve the place, which then produces increased benefits to both nature and people. This centering of the positive impacts of human actions contrasts with much of the past urban social-ecological systems literature that highlights the negative impacts of humans and anthropogenic change (Tidball and Stedman, 2013). This Research Topic includes papers illustrating the Virtuous Cycle Framework and the power of creating regenerative cycles in urban ecosystems.

Our Research Topic of 11 papers represents a range of urban social-ecological research areas incorporating the relationships between people, places, and nature. These relationships underpin the Virtuous Cycle Framework, which examines how conservation might be relevant to people, recognizing that conservation “depends on social, economic, political,

and cultural systems to sustain it” (Morrison, 2016, p. 9). For example, Bixler et al. introduce a framework for reflexive co-production of knowledge, applying that framework to assess three initiatives for urban greening and climate impact risk reduction in Austin, TX, USA. They emphasize three iterative phases for co-production: Recognize, Reflect, and Respond, and describe how that process, when effectively implemented, can serve as a virtuous cycle toward building urban resilience.

Three articles explore urban forestry and the harvesting and use of urban wood. Grove et al. apply the concept of regenerative cultures and ecologies to highlight the urban wood systems in Baltimore, Maryland, USA as a case study and model for virtuous cycles, arguing that virtuous cycles are most impactful, adaptive, and resilient when they include both positive and negative feedbacks and synergies. Through three interacting examples, they describe how a team-of-teams approach is critical for tackling complex, social-ecological problems, and boundary objects are useful tools to collaborate and eventually build consensus. de Guzman et al. evaluate the Tree Ambassador, or *Promotor Forestal*, program in Los Angeles, USA, which aims to address urban forest equity and wellbeing by training, supporting, and compensating residents to organize their communities. They use the results of the study to produce a “Socio-ecological model of community-based tree stewardship,” which can be applied across levels of social organization, and spatial and temporal dimensions. Treglia et al. introduce the concept of “practical” urban tree canopy analysis in New York, USA, which considers where additional canopy can fit within the existing constraints and opportunities of the landscape. They describe how practical canopy analysis can be the driver for conversations, stakeholder engagement, and actions toward urban forestry goal setting and implementation.

Many of the studies are from Los Angeles (L.A.), California, USA, illustrating how the virtuous cycle can operate in various realms, even within a single urban area. Wohldmann et al. explore how urban resilience can be furthered through efforts aimed at building soil health. They describe the Healthy Soils for Healthy Communities Initiative, which collected survey and focus group data to study attitudes, beliefs, and behaviors around land and soil. They explore strategies for deepening community engagement, addressing knowledge gaps, and shaping policies, and they describe how their data are being used to inform community-based interventions. Zellmer and Goto describe how wildlife corridors may be used to connect fragmented wildlife populations, despite challenges posed by the multitude of barriers, habitat patches, and property owners present in an urban context. Their case study demonstrates the value and importance of a collaborative approach that includes scientists, non-profits, government agencies, and communities. Cooper et al. describe conservation efforts in wildlands in and near L.A., cataloging information on more than 3,000 parcels of public open space to understand the history of how and when lands were conserved. They argue that the act of open space protection furthers advocacy efforts that promote conservation-benefiting land use policies and additional habitat conservation efforts, therefore constituting a virtuous cycle of conservation.

Four papers focus on urban form and highlight how the built and manicured environment can feed into the Virtuous

Cycle Framework by providing benefits for both people and nature. Katagi et al. detail ongoing restoration efforts along the L.A. River, a managed waterway that plays a crucial role in connectivity for wildlife and human communities as it bisects the city, crossing numerous municipalities. The paper discusses conservation of “iconic” species, such as the endangered steelhead trout (*Oncorhynchus mykiss*), that can build support for broader initiatives to promote urban biodiversity and recreational opportunities for city residents. Vasquez and Wood focus on how urban parks in “park poor” sections of L.A. likely provide important habitat for birds as there are few other green options in the surrounding cityscape. They detail the importance of parks to birds and also discuss park development in underserved communities as a straightforward “win” when considering the benefits to both wildlife and people. English et al. focus on unmanaged grasslands along an urban-to-rural gradient and how grasslands within the cityscape have lower diversity of plant species, which peaked in intermediate zones along this gradient. Conserving such “remnant” and unmanaged patches of habitat within cities may be key to providing important habitat for plant species. Lastly, Beninde et al. harness the power of iNaturalist observations to create species distribution models for 1,200 species based on climate and landscape variables across the entirety of Greater L.A.—from the natural areas to the urban core. The paper provides one of the largest species distribution modeling efforts in an urban area, providing wall-to-wall predictions for many plants and animals. The paper is an example of the benefits of community science initiatives that generate excitement, build community, and connect people to nature while providing critical data for urban conservation.

The studies in this Research Topic illustrate the Virtuous Cycle Framework by describing interventions that can produce benefits for both people and nature in a given location, and/or discussing datasets that can help to identify potential interventions and appropriate locations for them (e.g., using iNaturalist observations or camera trap data). While the Virtuous Cycle Framework offers a general model or heuristic for affecting positive outcomes, a clear challenge—and an opportunity—is in quantifying those outcomes to evaluate whether there are measurable benefits to biodiversity, people, and place. Ideally, when designing virtuous social-ecological cycles, resources can be directed toward assessing outcomes, which may include conducting surveys, interviews, and/or otherwise quantifying benefits to people. Strategic evaluation can then inform ongoing management such that the cycle can be optimized to achieve the desired benefits. Such evaluation efforts are likely to be helpful in ensuring that interventions are not top-down but are developed collaboratively with the relevant communities, further supporting the positive feedback loop envisioned by the Virtuous Cycle Framework.

## Author contributions

MR: Writing—original draft, Writing—review and editing. SP: Writing—original draft, Writing—review and editing. GP: Writing—original draft, Writing—review and editing. EW: Writing—original draft, Writing—review and editing.

## Acknowledgments

We thank Kat Superfisky from Los Angeles City Planning, Michelle Barton from Los Angeles Sanitation and Environment, and Edith de Guzman from the University of California Los Angeles for helping to develop this Research Topic, which resulted from a session at the 2021 Ecological Society of America conference.

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## OPEN ACCESS

## EDITED BY

Michele Romolini,  
Loyola Marymount University,  
United States

## REVIEWED BY

Kirsten Schwarz,  
University of California, Los Angeles,  
United States  
Keith Gordon Tidball,  
Cornell University, United States

## \*CORRESPONDENCE

Morgan Grove  
jmgrove@gmail.com

## SPECIALTY SECTION

This article was submitted to  
Urban Resource Management,  
a section of the journal  
Frontiers in Sustainable Cities

RECEIVED 13 April 2022

ACCEPTED 05 July 2022

PUBLISHED 02 August 2022

## CITATION

Grove M, Carroll J, Galvin M, Hines S,  
Marshall LL and Wilson G (2022)  
Virtuous cycles and research for a  
regenerative urban ecology: The case  
of urban wood systems in Baltimore.  
*Front. Sustain. Cities* 4:919783.  
doi: 10.3389/frsc.2022.919783

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# Virtuous cycles and research for a regenerative urban ecology: The case of urban wood systems in Baltimore

Morgan Grove<sup>1\*</sup>, Jeff Carroll<sup>2</sup>, Michael Galvin<sup>3</sup>, Sarah Hines<sup>1</sup>,  
Lauren L. Marshall<sup>4</sup> and Gene Wilson<sup>5</sup>

<sup>1</sup>United States Department of Agriculture Forest Service, Baltimore, MD, United States, <sup>2</sup>Urban Wood Economy, Baltimore, MD, United States, <sup>3</sup>SavATree, Annapolis, MD, United States, <sup>4</sup>Arbor Day Foundation, Lincoln, NE, United States, <sup>5</sup>Room & Board, St. Paul, MN, United States

The field of urban ecology has progressed since the mid-1990s through four major phases: an ecology *in*, *of*, *for*, and *with* cities. This progression reflects an interest to address the complexity of urban systems with social-ecological approaches. Further, this progression signifies an interest to address societal issues by co-designing and co-producing research in collaboration with diverse stakeholders from government, non-governmental organizations (NGOs), businesses, and community associations. What remains unaddressed in this progression is a research mission orientation. While there may be a range of goals for an ecology *with* cities, a focus on regenerative urban ecologies is crucial. Regenerative ecologies may be seen as an endpoint along a continuum from degenerative ecologies to sustainability to regenerative ecologies. Regenerative ecologies rely upon feedback loops, similar to coral reefs and climax forests. In urban systems, these feedbacks in social-ecological systems may be considered virtuous cycles that create reinforcing, positive benefits for people and nature over time. Virtuous cycles or feedbacks are often conceived as a singular, positive feedback loop. However, virtuous cycles may be most impactful, adaptive, and resilient when they contain multiple positive and negative feedbacks and synergies. Research has several important roles in advancing virtuous cycles and regenerative urban ecologies. In this paper, we use our urban wood systems project in Baltimore as both a case study and model to illustrate an approach and lessons learned for regenerative ecologies, virtuous cycles, and the role of research. We conclude with lessons learned and consider opportunities and constraints for virtuous cycles, research, and regenerative urban ecologies in Baltimore and to other urban systems.

## KEYWORDS

regenerative, urban, ecology, Baltimore, wood, virtuous

## Introduction

Urbanization continues to grow globally in terms of area, population, and the teleconnections among urban and rural areas. The science of urban ecology has also grown significantly since the 1990s. As the field of urban ecology has developed, it has become more inclusive of disciplines and practices and open to collaboration among actors to address societal issues (Pickett et al., 2022). We propose that a new focus on regenerative urban ecologies is needed to address these combined trends of societal transitions in urbanization and an urban ecology to promote social and ecological health, wellbeing, and equity.

In this paper, we note some of the major themes of urban ecology that lead to regenerative urban ecologies. We then outline the fundamental characteristics of this focus and how it is distinct along a gradient from degenerative to sustainable urban ecologies. Cultures and practices are needed to operationalize this approach, and we discuss strategies and tools for implementing regenerative urban ecologies. We use three cases studies based upon our urban ecological work in Baltimore to illustrate this approach. First is our urban wood system approach that creates “wealth from waste” from deconstructed buildings and the city’s removal of dead or dying trees. A key component of this approach is jobs for individuals who have barriers to employment from historically segregated neighborhoods. The second examines the role of one of our key partners in this urban wood system, Room & Board, and the prospects for the private sector to play a regenerative role through B Corporation approaches. Finally, our third case study completes the circle from building deconstruction to neighborhood rejuvenation by exploring the opportunity for community-based, neighborhood revitalization through the design, construction, and maintenance of parks and novel financing from social and environmental impact bonds. Within and across these three cases studies, dynamic feedbacks that create virtuous cycles are fundamental to a regenerative urban ecology approach. In the final section of this paper, we conclude by examining the generalizability of our urban wood systems approach to other urban areas in the United States, and the ability of our lessons learned for a regenerative urban ecology approach to be applied to other urban social-ecological concerns.

## Sections

### Trends in urban ecology

Since the 1970s, urban ecology has deployed four increasingly inclusive paradigms, from an ecology *in* cities to an ecology *of*, *for*, and *with* cities (Cadenasso and Pickett, 2013; Childers et al., 2015; Pickett et al., 2022). This progression in urban ecological paradigms is manifest along several

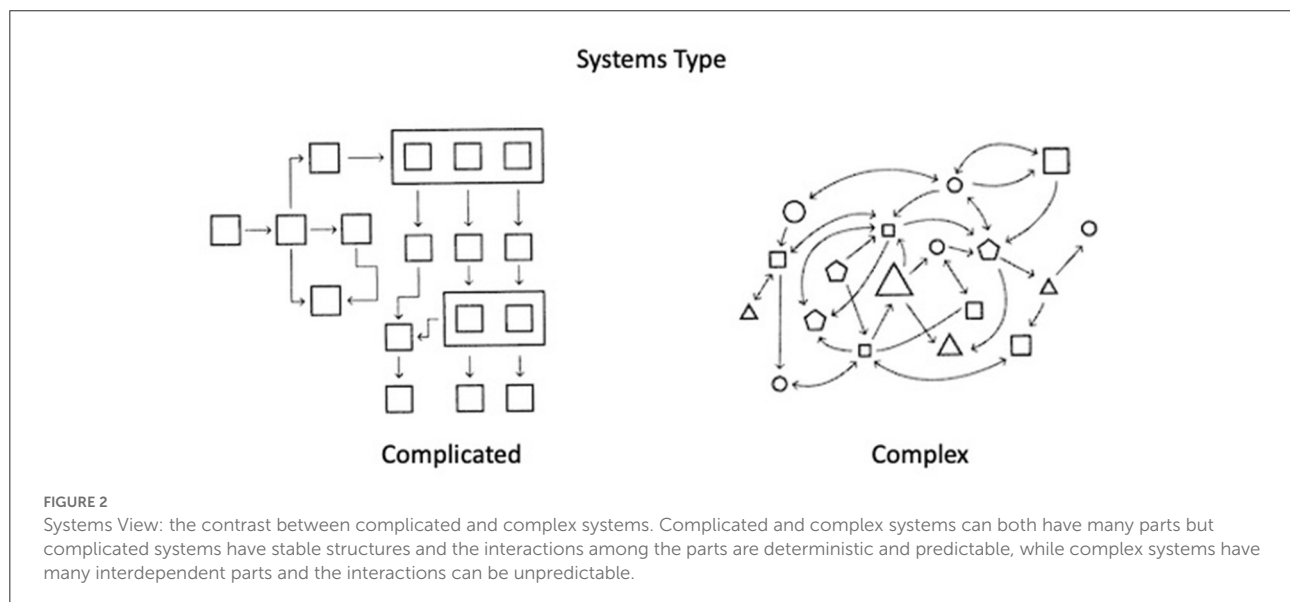
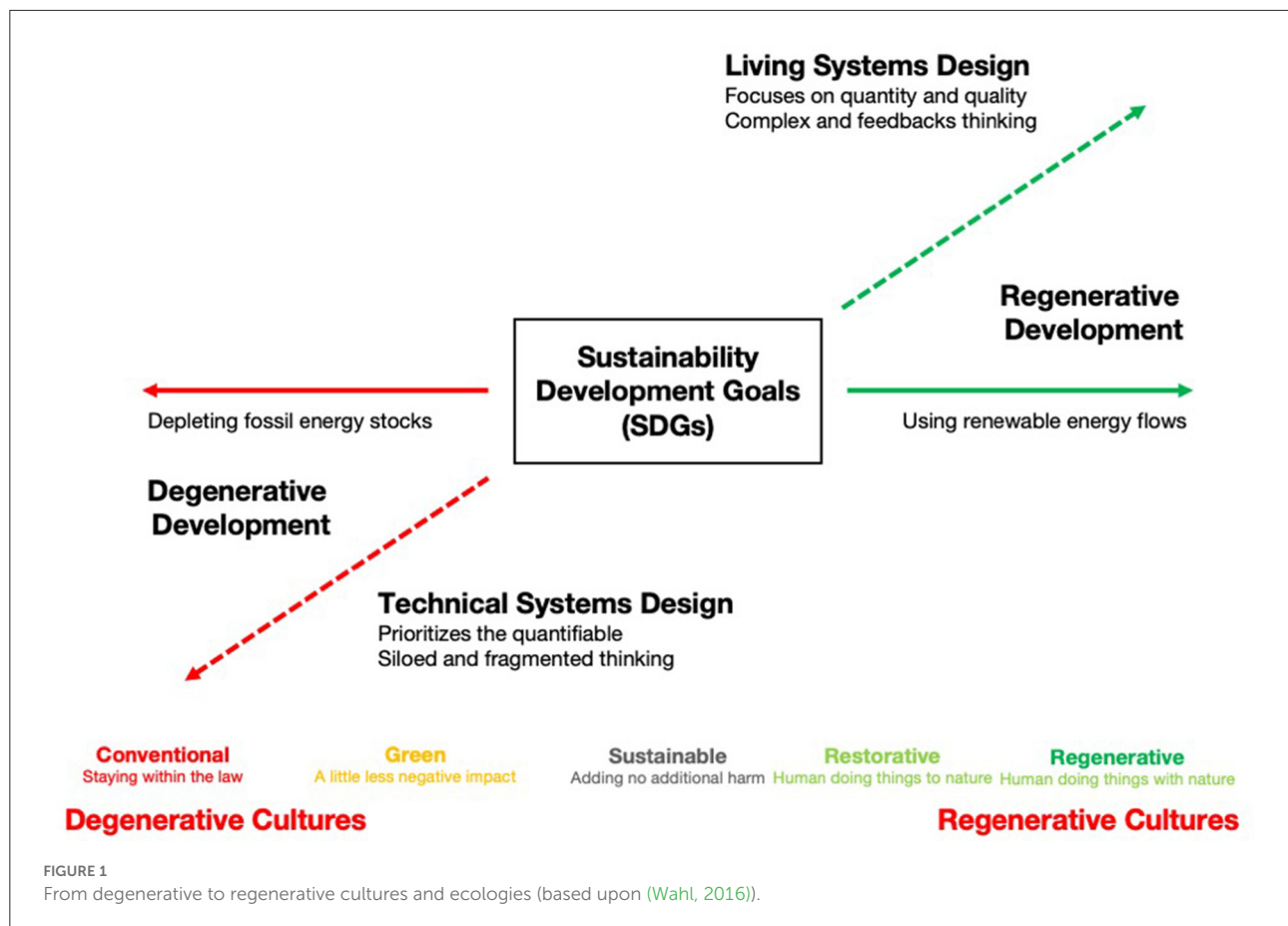
dimensions in terms of place, time, scale, knowledges, and how knowledge is produced (Pickett et al., 2022). The shift in place is represented by moving from a focus on forests in parks to the consideration of the entire urban mosaic. The shift in time is represented by moving from considering only contemporary events to including temporal lags and legacies over centuries. The shift in scale is evident from considering only human individuals to nested hierarchies that include households, neighborhoods, municipalities, and global systems. The shift in knowledges is demonstrated by relying upon only biophysical explanations to recruiting diverse sciences and humanities for understanding. Finally, the shift in the production of knowledge is evident in transdisciplinary approaches that engage diverse communities in the co-design and co-production of knowledge (Childers et al., 2015; Zhou et al., 2017; Pickett et al., 2022). Finally, this progression in urban ecological paradigms is manifest in the fundamental conception of cities as complicated systems to conceiving of cities as complex systems that are co-produced by interacting ecological and social phenomena (Pickett et al., 2022). What is missing, however, in this progression in paradigms is a sense of mission, culture, and practice. We propose that a regenerative urban ecology is a direction to pursue.

### Regenerative cultures and ecologies

Regenerative cultures and ecologies (Wahl, 2016) may be best understood in contrast to and along a continuum from degenerative to regenerative cultures and ecologies (Figure 1). In its most simple form, regenerative cultures and ecologies emphasize “leaving it better than you found it” by advancing a range of United Nations’ Sustainability Development Goals (UN SDGs). Two key strategies are to “think like nature” and to employ natural components and processes. These same strategies are often referred to as biomimicry within design disciplines (Kennedy et al., 2015). Some examples of SDGs include actions to address poverty; hunger; health and wellbeing; education; equality; water and sanitation; energy; work and economic growth; industry; innovation and infrastructure; and consumption and production.

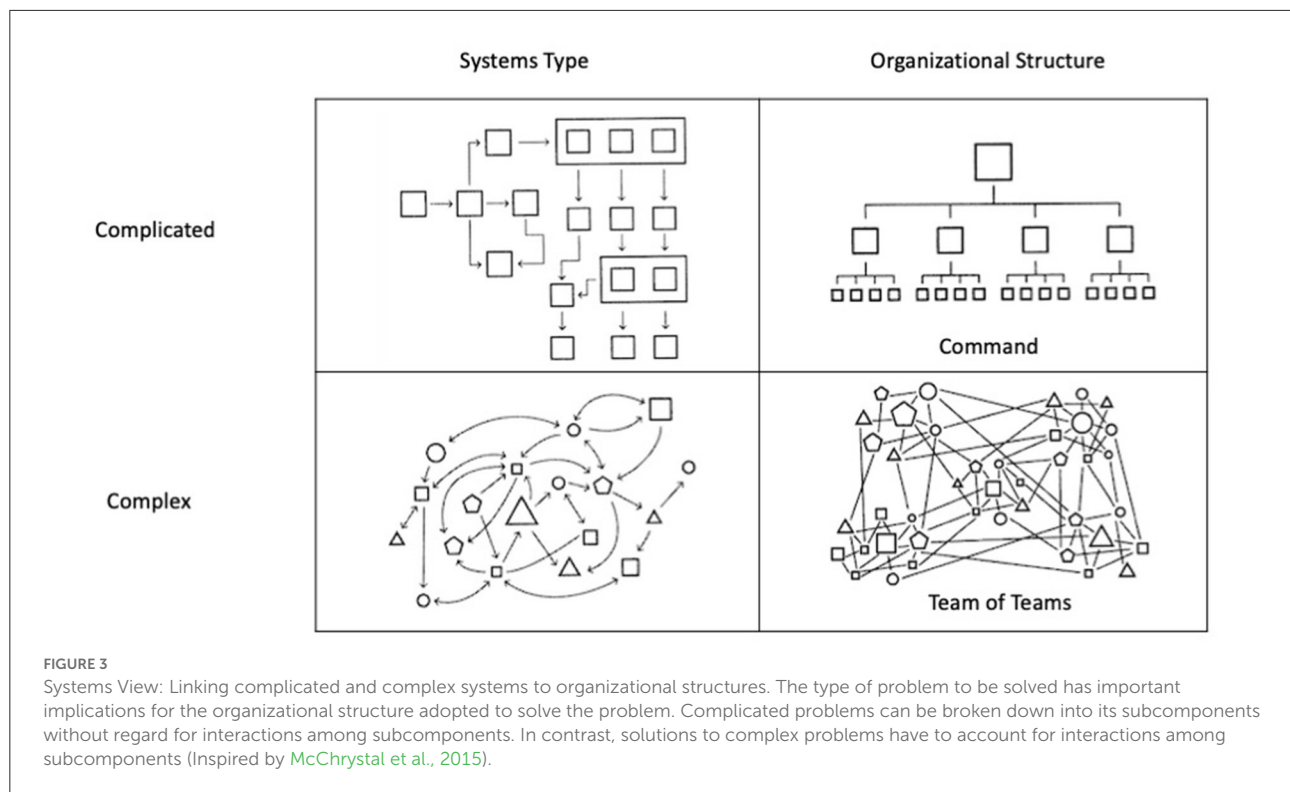
To think like nature may be rephrased as ecological thinking. A key feature of ecology and ecological thinking is its focus on the dynamic interactions among the parts of the system. An essential characteristic of these dynamic interactions is highlighted by the distinction between complicated, mechanical systems and complex, organic systems. Both complicated and complex systems can have many parts. However, complicated systems have stable structures and the interactions among the parts are deterministic and predictable (Allen et al., 2018). In contrast, complex problems have many interdependent parts and the interactions among the parts are unpredictable (Figure 2).





A second key feature of ecological thinking are models of highly retentive ecosystems of energy and nutrients such as coral reefs and climax forests (Johnson, 2002; Simard, 2021). In these

cases, outputs from one species are inputs to another species, and energy and nutrients are recycled and conserved in the system. To rephrase in more human-centric terms, “there is very



little waste”. Feedback loops are essential characteristics to highly retentive ecosystems. Feedbacks can be positive and reinforcing or accelerating such as the greenhouse effect. Feedbacks can also be negative and balancing or self-regulating such as a rheostat and the temperature of a room. In essence, positive feedbacks amplify changes in the system and negative feedbacks dampen changes in the system ([Tidball and Aktipis, 2018](#); [Tidball et al., 2018](#)). The terms “positive” and “negative” are not normative, value statements; rather, they describe how the dynamic interactions of the system promote, regulate, or diminish growth. In social-ecological systems, both positive and negative feedbacks may be considered virtuous cycles that create reinforcing, positive benefits for people and nature over time. It is important to note that virtuous cycles are often conceived of as a singular, positive feedback loop ([Morrison, 2015](#)). However, virtuous cycles may be most impactful, self-regulating, adaptive, and resilient when they contain a combination of social, economic, and environmental outcomes ([Morrison, 2015](#)) and have multiple positive and negative feedbacks and synergies.

## Strategies and tools for putting regenerative cultures and ecologies into practice

### Team of Teams

Regenerative cultures and ecologies emphasize the use of nature in achieving social goals. This recalls that urban

ecological systems are co-produced by ecological and social phenomena. In this case, it is important to consider what organizational social structures are needed to design and manage complex, social-ecological systems to produce virtuous cycles and positive social outcomes. As we have noted before, complicated systems have stable structures and predictable interactions among the parts. Reductionist approaches and siloed organizations can be highly effective for solving complicated problems. In contrast, complex problems have many interdependent parts and the interactions are unpredictable. Networked systems are often needed to solve complex problems ([Figure 3](#)). Further, no single organization has sufficient diversity in perspectives, motivations, and capacities to comprehensively address complex, social-ecological problems. Partnerships are needed for coordination and collaboration, often across sectors, specialties, and disciplines from government, civic organizations, business, and academia. Over time, collaborative teams that endure can develop to have their own intrinsic value. To build and sustain collaborations, we subscribe to a “Team of Teams” approach.

We rely on General Stanley A. McChrystal’s conception of a “Team of Teams” as an essential practice for regenerative urban ecologies to tackle complex problems ([McChrystal et al., 2015](#)). General McChrystal developed his ideas around a team of teams as the Director of the Joint Staff of the Joint Special Operations Command of the U.S. military’s fight against Al-Qaeda from 2003 to 2008 in Iraq. During this time, McChrystal and his staff recognized that the U.S. military

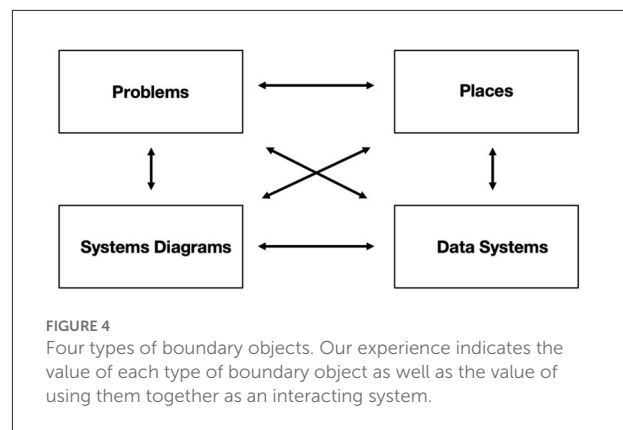
was organized hierarchically, which was appropriate to solve complicated problems. However, Al-Qaeda was a networked and decentralized, complex adversary, and the task of defeating Al-Qaida presented a complex problem (Figure 3). McChrystal and his staff realized that they needed to develop the necessary social organization and teamwork for a complex rather than a complicated problem.

There are several critical features for teams, or a team of teams, to solve complex problems (McChrystal et al., 2015). Such features include specific organizational structures, cultures, and interactions among teams. Organizationally, there is a need to shift to small teams and from an emphasis on efficiency to adaptability. Culturally, what makes small teams adaptable are trust, common purpose, shared awareness, and the empowerment of individuals to act. These adaptive features are critical because they invigorate teams with an ability to solve problems that could never be foreseen by a single leader. Ideas and innovations often emerge through the bottom-up result of interactions, rather than from top-down directions. Trust is crucial within a team and among teams. Strong lateral ties are essential for developing trust and the construction of shared awareness.

### Boundary objects for collaboration and building consensus for a team of teams approach

Boundary objects can be essential tools for working across sectors and disciplines to support a team of teams approach. The initial conception of boundary objects was intended to describe and understand the cooperative nature of scientific work in the absence of consensus (Star, 2010, p. 604). Star and Griesemer (1989) observed that scientific problems and their solutions often appear to be ill-structured, inconsistent, ambiguous, illogical, and complex. At the same time, science often requires cooperation among actors to create common understandings, to ensure reliability across scientific domains, and to collect information. These requirements can create fundamental conflicts between reconciling divergent viewpoints and the desire to produce generalizable findings. Thus, a key question in science, particularly when addressing ill-structured and complex problems, is how to manage diversity and cooperation among actors. These challenges appear identical to the organizational challenges for developing regenerative ecologies with a virtuous cycles framework.

Many models of cooperation assume that consensus must occur before cooperation can begin. However, teams may often develop strategies for cooperation without first requiring consensus (Star, 2010). Boundary objects can be an effective tool for building a team of teams approach. The idea of boundary objects is that they sit in the middle among different perspectives (Star, 2010, p. 608). In this usage, boundary objects are flexible and shared intellectual or physical structures that enable groups



to work cooperatively and manage diversity to address ill-structured or complex problems.

Boundary objects are useful in several ways. They allow team members to cooperate and work collectively (1) without having good understandings of each other's work; (2) with different perspectives; and (3) have different goals and motivations (Cash et al., 2003). An important test of boundary objects is their ability to encompass, change, and adapt to multiple perspectives while increasing communication among perspectives (Star and Griesemer, 1989). This is an essential cultural practice for different actors in teams to conceive of and negotiate regenerative problems and to conceptualize how they fit in and identify the appropriate roles for their participation (Cash et al., 2003; Barry et al., 2008).

While Star focused on projects that primarily involved scientists, we have found the idea of boundary objects to be a valuable, practical set of tools to tackle complex problems with a team of teams approach (Figure 4). We have found four types of boundary objects to be particularly useful. We use examples here in anticipation of our case study below. *Problem(s)* definition can often start loosely and iteratively as team members offer their different perspectives on the problem. For instance, "how to reduce the amount of wood waste entering landfills?" requires a variety of perspectives on how wood is generated, alternative ways that it can be processed, and a range of ideas for how wood could be used. *Places* are often relatively familiar locales that have the same boundaries but whose contents will appear differently to different team members. Places may be multi-scaled, nested places such as an urban region and its municipalities and neighborhoods or different types of places such as the organization and linkages of wood sort yards, wood processing yards (drying and rough milling), and manufacturing shops for making products such as furniture or flooring. An example of how various people may see the same locale differently is how those charged with deconstructing buildings or removing dead trees may see the city differently from those who are interested in making furniture

or those who are interested in creating job opportunities for returning citizens. *System diagrams* and other forms of symbolic abstraction are not intended to precisely describe the details of a place or a thing. They are abstractions from relevant knowledge domains. These diagrams “serve as a means of communicating and cooperating symbolically—a ‘good enough’ road map for all parties” (Star and Griesemer, 1989, p. 410). Finally, *data systems* that support diverse perspectives of the place, problem, and system are a crucial tool for collaboration and understanding.

Star did not suggest that boundary objects be used interactively as a system. However, we have found that the value of each type of boundary object increases when used in combination with the others. Using the four boundary objects iteratively can promote novel insights as well as test, evaluate, and validate how well the team has described and understood the problem. For instance, the team might first start by trying to describe the problem. They may convert their discussion to a system diagram. The team might discuss how well the system diagram describes and maps to their place of interest. And then the team might assess how well existing data systems enable them to understand the problem, system, and place. This example is not to suggest that teams try to employ all four boundary objects simultaneously at the beginning of a project. Rather, our point is to identify these four types as useful tools and that any one of these types is a good place to start.

The development and maintenance of a team of teams and the use of boundary objects do not happen spontaneously. Both Cash et al. (2003) and McChrystal et al. (2015) emphasize the importance of organizations that act as intermediaries among sectors, specialties, and disciplines from government, non-profit organizations, business, and academia. These “boundary organizations” play several roles. They are good at mapping organizations to the combination of problem-place-systems boundary objects; helping organizations identify their place in the system; and understanding different organizations’ perspectives for how they see the system, their roles, motivations, and capacities. Finally, boundary organizations often have developed skills, tools, and procedures to manage functions of communication, translation, and mediation at the boundaries among sectors, specialties, and disciplines.

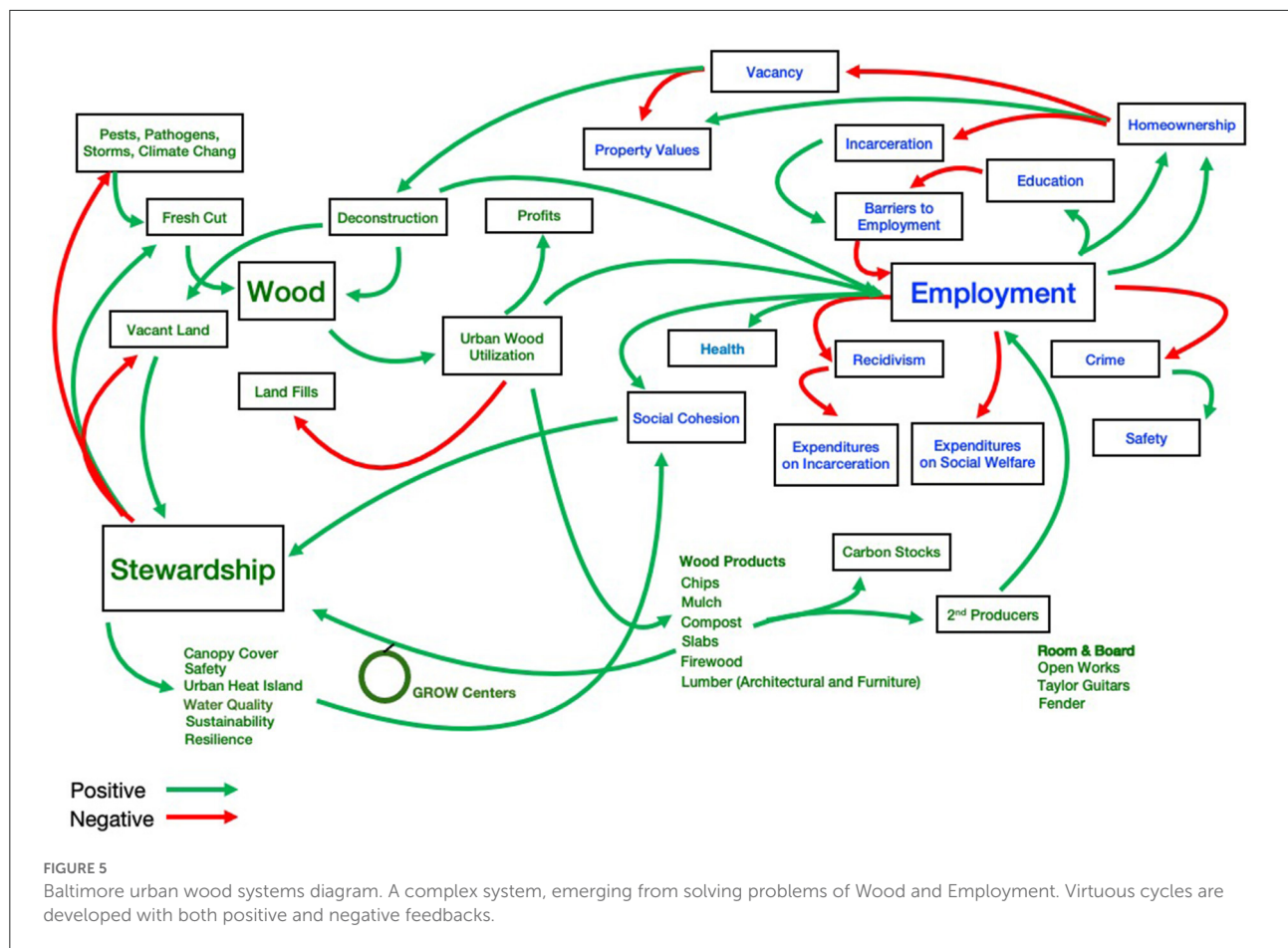
### Key actors for a “team of teams” approach and the use of boundary objects

To advance regenerative urban ecologies and virtuous cycles, there are several key actors for a team of teams approach and the use of boundary objects. Government agencies and civic organizations are often the principal actors in regenerative urban ecologies. We suggest that research and the private sector are underutilized and essential actors to be included. Research has several important roles to play. First, research can organize existing knowledge and data into systems understandings of key

components and interactions. Here, we emphasize that these components and interactions may be both ecological and social. Given that these are complex systems, it is critical to identify linkages, feedbacks, and leverage points. Additionally, because of a team of teams orientation, organizational analysis of existing and potential partners in the system and their perspectives, motivations, and capacities is an important feature. Second is to identify uncertainties and unknowns where new research may be needed. Third is to reduce uncertainties and increase confidence for creating solutions in the form of policies, plans, and projects from multiple sectors: public, private, and civic. Fourth is to quantify positive and negative outcomes to verify the effectiveness of actions; to market the value of activities for both traditional and novel sources of investment such as pay-for-success models and social and environmental impact financing; and to support adaptive management and learning.

Finance and the private sector also have several important roles to play in regenerative cultures and ecologies. These roles are associated with recent shifts in perspectives, motivations, and capacities in finance and the private sector. First is an expanded view of how to create value. Creating value is frequently seen in terms of profits and revenues. However, value can also be created by avoiding costs, such as the cost of healthcare, crime, incarceration, trash, or water pollution. By creating financial instruments that value and pay for avoiding costs, incentives are created for developing feedbacks and tightly-coupled systems that recall the ecological thinking and complex, forested communities mentioned earlier. Second is an expanded view of financial instruments. Historically, many public, private, and civic activities and services have been paid for through taxes, profit, or philanthropy. New practices, financial instruments, and markets have recently emerged for social and environmental financing that support “pay-for-performance” activities, often focused on maximizing the avoidance of costs (e.g., Quantified Ventures, 2018, 2019). These new markets and tools are an essential means to support feedbacks in regenerative urban ecological systems.

Finally, the emergence of B Corporations and B Corporation thinking over the past 10 years signals an expanded view of corporate organization and their behaviors (Marquis, 2020). When businesses incorporate as B Corporations, their governance regime shifts from maximizing value for shareholders to maximizing value for stakeholders. In this context, stakeholders include employees, customers, society, and the environment. Crucial to environmental and social concerns, B Corporations are chartered to internalize what had been treated as environmental and social externalities—air and water pollution downstream, employees requiring public assistance because of low wages—and to perform in ways that maximize sustainability. Additionally, B Corporation culture values business to business (B2B) cooperation among B Corporations and sharing lessons learned for innovation and improving sustainability



practices (Marquis, 2020) that support knowledge feedbacks in the system.

## Discussion: Case study

We use our urban wood systems project in Baltimore as both a case study and model (Hines et al., 2019) to illustrate an approach and lessons learned for regenerative ecologies and virtuous cycles. Our project started with two boundary objects: Problem and Place. Our urban wood systems project originated to solve two problems in Baltimore: wood waste and unemployment. Traditionally, wood from building demolitions and tree removals in cities are treated as waste, often being disposed of in landfills. Nationally, 70.7 million tons of urban wood waste was generated in the United States in 2010, including 36.4 million tons from “Construction and Demolition Waste” such as construction, remodeling, or demolition of residential and commercial structures, and 34.3 million tons from “Municipal Solid Waste” (MSW) such as wood chips, pallets, and yard waste; tree trimming and storm damage; and construction or demolition wood. Of these 70.7 million

tons, the USDA Forest Service estimated that nearly 29 million tons of wood waste (41%) was suitable for recovery and reuse rather than being disposed in landfills. Our second major problem is unemployment in high poverty areas of Baltimore, particularly for individuals who have been previously incarcerated. In Baltimore, the poverty rate has increased from 18% in 1970 to 22% in 2016, over twice the average rate of about 10% across the State of Maryland. In high poverty areas, the unemployment rate ranges from 23 to 30%.

A third boundary object—Systems Diagram—is an effective way to describe and summarize existing and potential components, linkages, and team members in the project (Figure 5). There are several steps to creating a systems diagram. First is to identify the parts of the system and then the connections among the parts. The parts and the connections can be environmental, social, or economic. These connections are characterized as positive or negative. As we noted before, our use of the terms positive and negative are not meant to signal a normative value of good or bad. Rather, the terms positive and negative are used to indicate increase or decrease. For instance, a positive relationship between “deconstruction” and “wood” means that as deconstruction increases, wood increases



(green arrow). And, as urban wood operations increase, wood to landfills decreases (red arrow). Subsequently, we identify existing or potential feedback loops feedback loops, which can be positive or negative as we noted earlier.

On the left side of the diagram, we start by identifying the major sources of wood (Problem) in Baltimore. We have fresh cut and deconstruction. In Baltimore, the National Renewable Energy Lab (NREL) estimates that ~78,000 tons of urban wood waste is generated each year from MSW. Vacancy and abandonment are major drivers of deconstruction. The city has identified 16,577 vacant buildings. However, the total number is estimated to be as high as 46,000 vacant buildings. For fresh cut, the City's wood yard takes in ~8,000 tons of municipal logs and chips per year.

Increasing urban wood utilization is key to diverting wood from landfills. Urban wood utilization creates wood products such as chips, mulch, compost, slabs, firewood and dimensional lumber. Wood products such as chips, mulch and compost can be used to support the Department of Public Works' GROW Centers (Green Resources & Outreach for Watersheds), which are multi-partner, mobile "pop-up" centers to promote stewardship in historically racially-excluded neighborhoods. Stewardship can address a number of issues including increasing canopy cover, promoting safety, improving stormwater water quality, and contributing to sustainability and resilience. Stewardship can also reduce vacant lands and the effects of climate change. This cycle from wood utilization has both positive and negative feedback loops by reducing wood waste to landfills and the extent of vacant lands, and increasing stewardships to produce positive benefits and reduce drivers of climate change.

On the right side of the diagram, we start by identifying factors affecting employment in high poverty areas of Baltimore, particularly for individuals who have been previously incarcerated. The city's incarceration rate of 1,255 out of every 100,000 people is three times higher than both the state and national averages. Baltimore City residents comprise 10% of Maryland's general population but 33% of its prison population. Two major barriers to employment are education and incarceration. Increasing education decreases barriers, while increasing incarceration increases barriers. Increasing employment decreases crime and increases safety. Employment decreases recidivism and expenditures on incarceration and social welfare. The state of Maryland spends \$300 million each year to incarcerate people from Baltimore City at a cost of ~\$38,000 per inmate. Former inmates often return. Estimates suggest that 73% of citizens returning from prison to Baltimore are incarcerated again within 3 years.

Increasing employment can also increase education through employer reimbursements as well as lead to homeownership and increasing property values. Employment and wood connect through people being employed in deconstruction, wood utilization, and "2nd Producers" who make things such

as home furnishings, furniture, and flooring. Employment is also connected to wood by increasing homeownership, which decreases vacancy, and then decreases the need for deconstruction. A key lesson from this illustration and analysis is that there can be numerous, virtuous cycles that consist of multiple, positive and negative feedback loops.

Our systems diagram has been essential to advance a team of teams approach, working among sectors, specialties, and disciplines from government, non-profit organizations, community groups, business, and academia. Key players include the non-profit organization, Humanim, its subsidiaries Details and Brick + Board, community partners from the Station North and Johnson Square neighborhoods, government agencies at local (Baltimore City Division of Forestry and Departments of Housing and Community Development and Public Works), state (MD Departments of Housing and Community Development and Natural Resources' Forest Service) and Federal levels (USDA Forest Service), private sectors including the home furnishings company Room & Board and the consulting firm Quantified Ventures,. These sectors brought key perspectives and knowledges from wood, wood processing, and wood making; job training and employment, and social-environmental valuation and analysis.

Systems diagramming enabled team members to contribute their expertise and question the team's collective understanding. We could then identify additional information needs and uncertainty as well as improve our operations and maximize the feedbacks that produce virtuous cycles. It also enabled us to communicate the diversity of benefits and outcomes and attract new members to the team, particularly those who were interested in creating value by avoiding costs associated with landfills, incarceration, crime, and stormwater pollution. Finally, the systems diagram as a data system (fourth boundary object type) has two important roles. First, it enabled us to quantify and communicate the diversity and magnitude of direct social, economic, and environmental outcomes in terms of monetary value. This can be crucial for reshaping policies and regulations as well as attracting social and environmental impact investment. Second, it supported adaptive management through monitoring and evaluation.

## Phase II: Completing and expanding cycles of "what could be"

We shift from what has already been accomplished in the urban wood systems in Baltimore to opportunities to complete and expand new cycles. From our existing systems diagram (Figure 5), we focus on two components: the role of Room & Board and its potential role in circular economies, and land reclamation of vacant lots as a specific form of Stewardship. These two examples address several interconnected problem



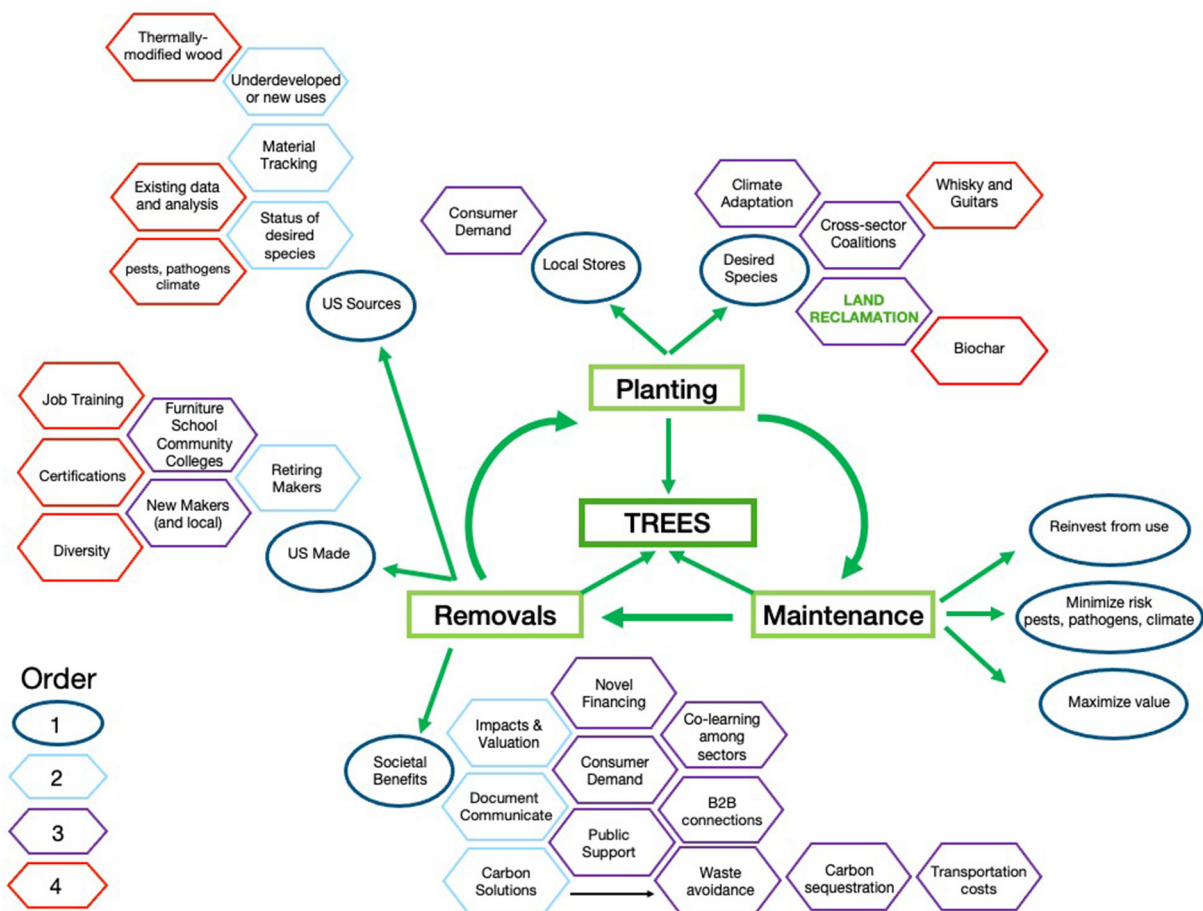


FIGURE 6

Room and Board and its potential role in circular economies. Understanding the perspectives and motivations of specific actors in the urban wood systems (re. Figure 4) can be used to analyze how they can engage with the virtuous cycle of tree planting, maintenance, and removals.

areas: how to invest and support circular economies; how to convert wood materials from low value to high value; how to restore vacant, degraded lands; and how to adapt to climate change.

### Room and board and its potential role in circular economies

Our first example focuses on the perspectives and motivations of one of our team members, the private sector company Room & Board. We conceive of this example in terms of a virtuous, circular economy of tree planting, maintenance, and removals and the role that Room & Board could play to advance a regenerative urban ecology (Figure 6). In this example, we first return to the tenets of B Corporations that we discussed earlier. Although Room & Board is not a B Corporation, it can still adopt B Corporation thinking, including a shift from shareholders to stakeholders, business to business collaborations among B Corporations,

and shared learning within and among sectors. Room & Board is already well-positioned to adopt a B Corporation approach for several reasons. First, Room & Board already has a stakeholder approach. It is positioned primarily in the U.S. for almost all of its materials and production. The company cultivates and values its long-term relationships with its makers, who are often small firms. It also cultivates and values its relationships with its employees and customers. Finally, the company has a long-term, sustainability ethic in terms of both its products and the environment. The question remains, however, how could a company like Room & Board go from a sustainable to a regenerative approach (re: Figure 5).

At the center of this circular economy is the organizing idea of a regenerative, healthy, and climate adapted canopy of trees in urban areas. This involves three primary activities: tree planting, maintenance, and removals (Figure 5). The diagram is organized into first through fourth order levels of interaction. We start with tree removals because this is the initial point of contact that

Room & Board has with urban trees. For removals, the first order items are U.S. Sources, U.S. Made, and Societal Benefits.

Room & Board sources most of the materials for its wood products from the United States. The primary wood species they use are white oak, ash, maple, and walnut. Given its long-term dependence on these species, it is crucial to understand the current status of these species in terms of quantities and quality, supply chains, and potential future threats due to pests, pathogens, and climate change. Room & Board could partner with government agencies and universities to analyze current conditions and forecast long term trends in supply and possible alternatives if species supplies are at risk. A second area of focus is the development of novel, material tracking systems. These tracking systems can be used in two ways. First is to track wood from the initial source to the final product. Here, the goal is to retain the value of the sustainability story from the address where the tree had lived, to who made the product, to its eventual arrival at a home. This could add significant value because urban wood makers have found that there can be as much as a 40% increase in the value of their products if they can include the origin and journey of the wood. The second reason for material tracking systems is that individual, urban wood systems are often not sources of large volumes of high-quality material. Thus, there is the need to aggregate within urban wood systems and even regionalize among urban wood systems. For instance, there may be the need for regional aggregation of white oak from cities extending from Richmond, VA to Philadelphia, PA to meet the quality and quantity needs of a regional or national maker. Thus, material tracking systems enable local urban wood systems to develop through participation in regional aggregation markets. Finally, there is the need to explore underdeveloped or new uses for wood. For instance, some tree species are widely available but have often not been desirable for furniture making. In the mid-Atlantic of the United States, tulip poplar (aka yellow poplar) is an example of an undesirable species for furniture making. However, tulip poplar can be cut to dimensions and dried through thermal modification to be rot resistant for outdoor uses. This converts a low-value species to a high-value product and reduces the dependence on tropical wood imports for outdoor furniture.

Making products in the U.S. is becoming challenging as the owners of the small firms that Room & Board traditionally works with are retiring. While this creates challenges, it also creates opportunities. Room & Board could develop new partnerships with existing makers, particularly makers who may be local to urban wood sources and Room & Board stores. They could also partner with training programs for new makers, such as furniture schools and community colleges. These programs can be essential for job training and certifications, expanding the workforce of makers. Room & Board could also support scholarships to support workforce diversity.

Societal benefits involve the ability to measure and value the impacts of Room & Board's regenerative activities. This is key for transparency and the ability to attract novel financing for Room & Board and its partners. The ability to document and communicate the social benefits could increase consumer demand for its products and build support for regenerative approaches, particularly those that involve the private sector. The ability to measure and communicate also connects to co-learning among sectors and establishing business-to-business partnerships among B Corporation aligned companies. Finally, carbon solutions are a societal benefit, which occurs through avoiding waste (re. [Figure 5](#)), sequestering carbon in durable wood goods, and minimizing transportation costs if the distance among wood sources, manufacturing, and customers is minimized.

Tree planting is a major component of this virtuous, urban tree cycle. Room & Board could invest profits from their use of tree removals to support local tree planting organizations in locations where they source their wood or have their stores, such as Baltimore, MD, Sacramento, CA, Austin, TX or Boston MA. These local investments could also create opportunities to engage and activate local customers to participate in local tree planting activities and increase local customer interest in Room & Board products. As trees are planted, it is important to consider desired species to promote climate adaptation, their use when they reach the end of their life cycle, and their use for land reclamation. Cross-sector coalitions may provide valuable input and investments in tree planting. For instance, the whisky industry depends upon white oak for its barrels, and guitar makers seek ash and other desirable tonal woods for their acoustic guitars. Land reclamation could involve new planting and soil amendments with biochar for establishing trees and improving stormwater quality (we discuss this further in our next example).

Tree maintenance is the third major component of this cycle. Profits from tree removals could again be used to support this component. Support for maintenance could be used to minimize risk to pests, pathogens, and climate change to extend the life of trees. Maintenance through pruning could also maximize the value of trees for when they must be removed.

### Neighborhood revitalization through land reclamation of vacant lots

Our third case study completes the circle from building deconstruction (re. [Figure 5](#)) to neighborhood rejuvenation by exploring the opportunities for community-based, neighborhood revitalization through the regenerative design, construction, and maintenance of parks and novel financing from social and environmental impact bonds. Our example draws from the work of The Parks & People Foundation and



FIGURE 7

Vacant buildings, deconstruction, and revival for McKean Park. (Photos: J. Morgan Grove, USDA Forest Service). Vacant buildings (A), deconstruction (B), and revival (C) for McKean Park.

its community partnership projects in historically racially-segregated neighborhoods such as McKean and Johnston Square neighborhoods (Place). These neighborhoods face a number of problems, including vacant buildings and vacant lands; unemployment; crime; absence of parks and green spaces; financial resources to build and maintain green spaces; and vulnerability to climate change such as flooding from microbursts of precipitation, heatwaves, droughts, and severe storms that can cause loss of electricity. Figure 7 shows the progression from vacant buildings to deconstruction and land clearance, and a new, neighborhood park.

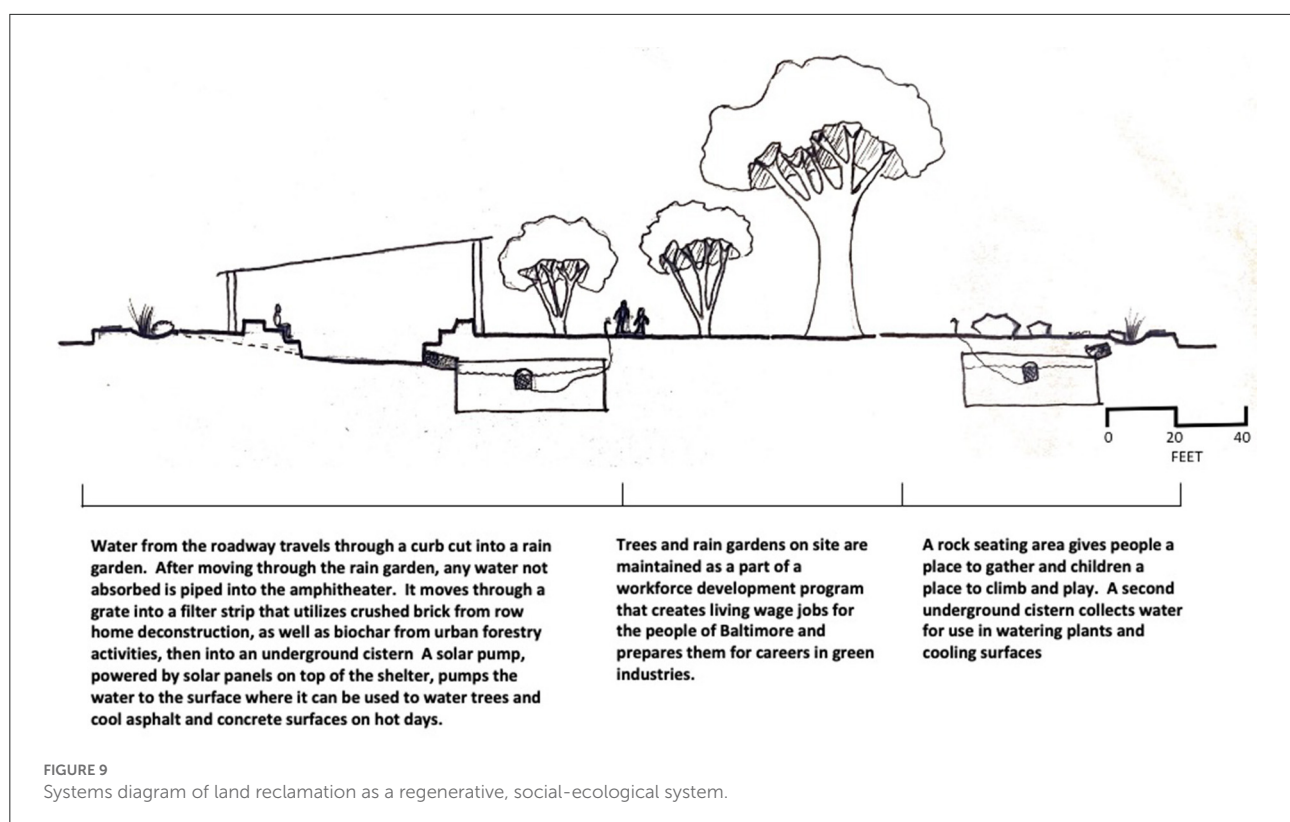
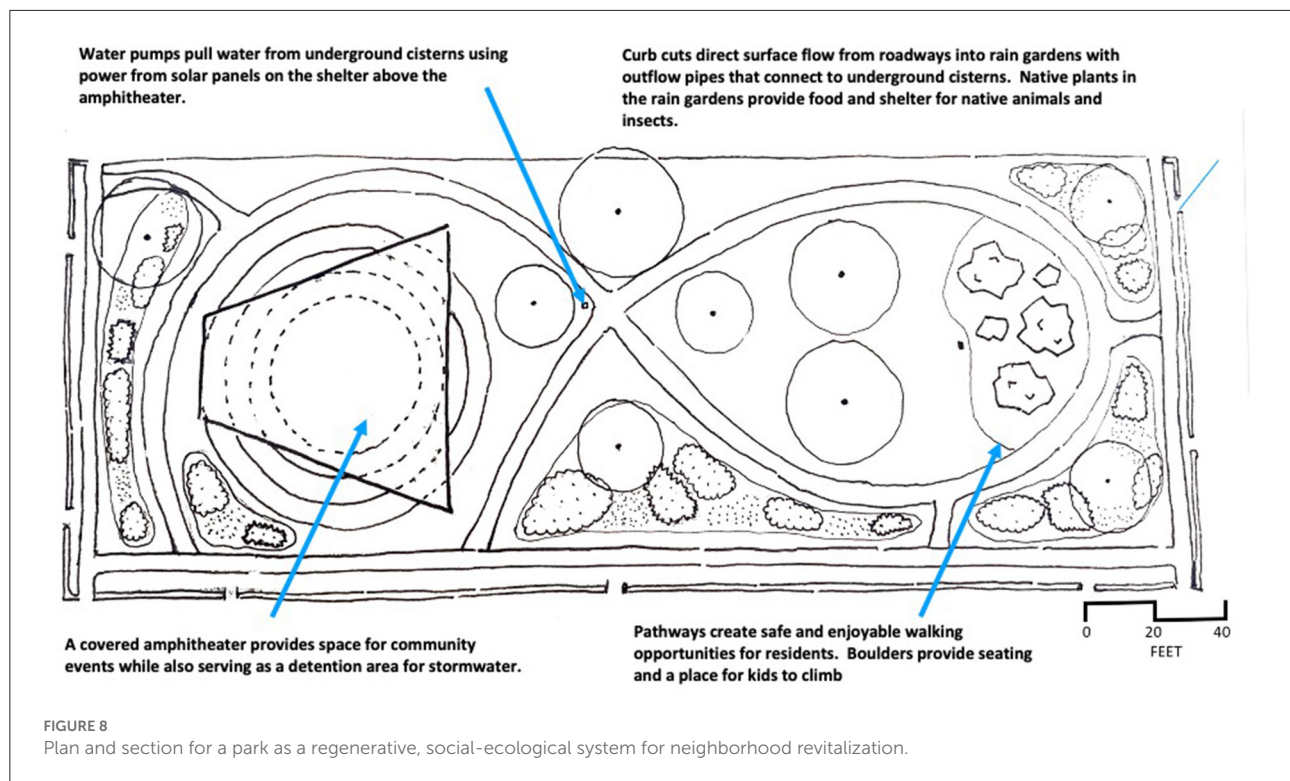
Figure 8 illustrates how these types of neighborhood parks could be designed and maintained to address these multiple problems with a regenerative, integrated design. For instance, the design offers an attractive community space integrated with rainwater capture, treatment, and storage; solar panels for micro-grid electricity; and the ability to avoid societal costs that could be captured with social and environmental impact bond financing to support the design, construction, and maintenance of the park (Figure 9).

Our regenerative, systems design for neighborhood parks retains components and feedbacks from our urban wood systems, particularly the Employment section (re. Figure 5). New to this systems diagram is the general benefits that

neighborhood parks can play to reduce vacancy and increase human health and social cohesion. Particular to this regenerative, systems design are the added features to address the many negative effects of climate change that are and will be experienced in vulnerable urban communities. These climate effects include increased precipitation, particularly microbursts, which increase flooding and reduce the water quality of stormwater runoff; increased drought which reduces plant health and the capacity of vegetation to cool the neighborhood through evapotranspiration; increases in the number and severity of heatwaves, which lead to human mortality or illness; and severe storms, which can lead to loss of electrical supply.

Water capture through curb cuts and the amphitheater roof, and water storage through below-ground cisterns reduce flooding. Water treatment through a filter system of crushed brick and biochar to improve stormwater quality. Energy from the solar panels on the amphitheater roof power pumps from the cistern to irrigate park trees, plants, and grass and adjacent street trees, which reduce the effects of drought. The trees, plants, and grass help to cool the neighborhood, which is particularly beneficial during heatwaves. Additionally, water from the cisterns can be applied to and cool impervious asphalt and concrete surfaces such as streets, which can reduce the local effects of heatwaves from 10° to 15°F. The energy from these solar panels can also be used after storm events and the loss





of electricity to recharge phones and other devices. Further, construction costs would be reduced by reusing crushed brick in the water filters, paths, and amphitheater from building deconstruction and biochar in the water filters and as a soil amendment from wood utilization operations (re. [Figure 5](#)).

The park design features we outline produce substantial “avoided costs” in healthcare to treat people for heatwave related illness and stormwater mitigation associated with the city’s MS4 compliance requirements. These avoided costs can be quantified and monetized through social and environmental impact bonds to finance employment for park design, construction, and maintenance. Additional “avoided cost” benefits could be researched and calculated for reducing stress and crime and increasing government revenues from increased property values.

## Conclusion

We conclude by examining the generalizability of our urban wood systems approach to other urban areas in the United States, and the ability of our lessons learned for a regenerative urban ecology approach to be applied to other urban social-ecological problems. The generalizability of the Baltimore urban woods systems can be characterized in several ways: opportunity, quantification, and dissemination. In terms of opportunity, there are substantial quantities of urban wood in terms of supply, production, and demand on a national basis. In the near term, we anticipate increasing supplies of urban fresh cut from increasing mortality due to pests, pathogens, storms, and climate change. For instance, a recent study forecasts that 1.4 million street trees in urban areas and communities will be killed by introduced insect pests by 2050. This represents 2.1–2.5% of all urban street trees ([Hudgins et al., 2022](#)). In the long term, trees being planted now as part of “1 Million Tree” initiatives in cities will result in significant tree removals in 50–70 years. We are also likely to see changes in production as companies seek to minimize overseas supply chain vulnerabilities and to maximize and market the sustainability of their products to their customers. Demand from customers is also likely to grow due to the changing characteristics of consumers, particularly younger consumers, who are demanding that their materials be sustainably sourced and produced.

In terms of quantification, the generalizability of our urban wood utilization approach depends upon the ability to apply our systems approach to other locations by identifying potential team members; targeting interventions; quantifying the diversity and magnitude of direct social, economic, and environmental outcomes, and monetizing those outcomes for government, civic, and private sector investments ([Morrison, 2015](#)). Intriguingly, [Morrison \(2015\)](#) suggests that the support for and resilience of a virtuous cycles solution increases as the diversity and quantification of social, economic, and

environmental outcomes increases. Although we are only one example, our experience supports this idea. Finally, in terms of dissemination, strategies and mechanisms for diffusion and adoption are crucial for a regenerative approach to urban wood utilization can become widespread. An urban wood network has already emerged that connects passionate champions and boutique operations into chapters that can aggregate to a national network of networks. The USDA Forest Service has emerged as a boundary organization that supports this network of networks through websites, multi-media products, and “Urban Wood Academies” to share knowledge, practices, and lessons learned and to promote network collaborations. In addition to the technical aspects of urban wood utilization, organizational and financial training are key for employing a team of teams approach, the use of boundary objects, and accessing novel financial investments.

Several “lessons learned” have emerged from our efforts. We believe these lessons may be generalizable to a diverse range of complex social-ecological problems. A systems view is crucial to address the complexity of social-ecological problems for several reasons. First, a complex systems view ([Figure 2](#)) encourages the team to engage the numerous positive and negative feedback loops necessary to advance what will likely be multiple and interacting virtuous cycles. This leads to an openness and aspiration to address multiple SDGs and produce one solution to solve many problems. Yet, not everything can be accomplished at the beginning of a project; so it is valuable to acknowledge challenges, aspirations, and “not-quite-ripe” situations.

A team of teams approach is critical for tackling complex, social-ecological problems, and boundary objects are useful tools to collaborate and eventually build consensus. This combination of a team of teams and boundary objects enables the diversity of team members to contribute their particular knowledges and perspectives to a greater and more complete whole. This is valuable because many of the most complex undertakings are never done by a single entity. Complex problems often require an interdependent mix of operational expertise, research, financing, and stakeholder engagement. While not mentioned earlier, in addition to trust, humility is an important characteristic for team members to bring to this collective work. Finally, there is a science and resources for training in transboundary practices and culture that needs to be more widely adopted ([Gordon et al., 2019](#); [Bammer et al., 2020](#)).

## Author contributions

MGr led the writing of the manuscript. SH, JC, MGa, GW, and LM provided ideas, content, and editing. LM provided the ideas and drawings for the land reclamation case study. All authors contributed to the article and approved the submitted version.

## Funding

USDA Forest Service provided support for this research and open access.

## Conflict of interest

Author JC was employed by Urban Wood Economy. Author MGa was employed by SavATree. Author GW was employed by Room & Board.

The remaining authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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## EDITED BY

Michele Romolini,  
Loyola Marymount University,  
United States

## REVIEWED BY

Tenley M. Conway,  
University of Toronto  
Mississauga, Canada  
Pallavi Saxena,  
University of Delhi, India

## \*CORRESPONDENCE

Edith B. de Guzman  
eb3@ucla.edu

## SPECIALTY SECTION

This article was submitted to  
Urban Greening,  
a section of the journal  
Frontiers in Sustainable Cities

RECEIVED 14 May 2022

ACCEPTED 13 July 2022

PUBLISHED 03 August 2022

## CITATION

de Guzman EB, Escobedo FJ and  
O'Leary R (2022) A socio-ecological  
approach to align tree stewardship  
programs with public health benefits in  
marginalized neighborhoods in Los  
Angeles, USA.  
*Front. Sustain. Cities* 4:944182.  
doi: 10.3389/frsc.2022.944182

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# A socio-ecological approach to align tree stewardship programs with public health benefits in marginalized neighborhoods in Los Angeles, USA

Edith B. de Guzman<sup>1\*</sup>, Francisco J. Escobedo<sup>2</sup> and  
Rachel O'Leary<sup>3</sup>

<sup>1</sup>Institute of the Environment & Sustainability, University of California, Los Angeles, Los Angeles, CA, United States, <sup>2</sup>Pacific Southwest Research Station, United States Department of Agriculture (USDA) Forest Service, Riverside, CA, United States, <sup>3</sup>City Plants, Los Angeles, CA, United States

Extreme heat in the United States is a leading cause of weather-related deaths, disproportionately affecting low-income communities of color who tend to live in substandard housing with limited indoor cooling and fewer trees. Trees in cities have been documented to improve public health in many ways and provide climate regulating ecosystem services via shading, absorbing, and transpiring heat, measurably reducing heat-related illnesses and deaths. Advancing "urban forest equity" by planting trees in marginalized neighborhoods is acknowledged as a climate health equity strategy. But information is lacking about the efficacy of tree planting programs in advancing urban forest equity and public wellbeing. There is a need for frameworks to address the mismatch between policy goals, governance, resources, and community desires on how to green marginalized neighborhoods for public health improvement—especially in water-scarce environments. Prior studies have used environmental management-based approaches to evaluate planting programs, but few have focused on equity and health outcomes. We adapted a theory-based, multi-dimensional socio-ecological systems (SES) framework regularly used in the public health field to evaluate the Tree Ambassador, or *Promotor Forestal*, program in Los Angeles, US. The program is modeled after the community health worker model—where frontline health workers are trusted community members. It aims to address urban forest equity and wellbeing by training, supporting, and compensating residents to organize their communities. We use focus groups, surveys, and ethnographic methods to develop our SES model of community-based tree stewardship. The model elucidates how interacting dimensions—from individual to society level—drive urban forest equity and related public health outcomes. We then present an alternative framework, adding temporal and spatial factors to these dimensions. Evaluation results and our SES model highlight drivers aiding or hindering program trainees in organizing communities, including access to properties, perceptions about irrigation responsibilities, and lack of trust in local government. We also find that as trainee experience increases,

measures including self- and collective efficacy and trust in their neighbors increase. Findings can inform urban forestry policy, planning, and management actions at the government and non-profit levels that aim to increase tree cover and reduce heat exposure in marginalized communities.

#### KEYWORDS

urban forest management, urban forest equity, community engagement, tree planting, community-based climate adaptation, collaborative ecosystem management

## Introduction

The Los Angeles (LA), California metropolitan region of the United States (US) faces a range of challenges that are induced or exacerbated by extreme climate change events. Of all of the changes anticipated for the region, extreme heat has the potential to impact the largest number of vulnerable populations (Chakraborty et al., 2019; Li et al., 2020). Continued warming is projected to increase average temperatures 2.2–2.8°C (4–5°F) by mid-century, and by 2.8–4.4°C (5–8°F) by the end of the century, with temperature extremes expressed both in the rising number of extreme heat days, and in the hottest days being up to 5.5°C (10°F) hotter than extreme heat days previously experienced (Hall et al., 2018). In addition, due to climate and topographic variability in the LA region, some cities will have 5–6 times the number of extreme heat days compared to current levels (Hall et al., 2018). As the planet warms, urban areas are heating up at a faster rate than adjacent rural areas, placing in question the habitability of many cities and highlighting the need for solutions to address heat-related public health impacts (Estrada et al., 2017).

During the hottest summer days in LA, there is an 8% increase in all-cause mortality—deaths from all causes combined—as heat puts extra stress on people with a range of underlying co-morbidity conditions (Kalkstein et al., 2014). In particular, consecutive days of intense heat can have a very harmful impact, with all-cause deaths occasionally increasing by 30% above expected levels (Sheridan et al., 2012; Kalkstein et al., 2014). Public health is affected when higher heat exposure is coupled with limited ways of adapting to heat, particularly in the absence of nighttime relief from the heat, which can increase health risk even more than high daytime temperatures (Dousset et al., 2011).

The burden of extreme heat disproportionately affects vulnerable low-income urban populations and people of color in the US (Jesdale et al., 2013). These communities often live in high-density neighborhoods that have older, substandard housing, less urban tree cover (UTC), and limited access to air conditioning or the ability to pay for it, which create a

feedback loop of heating effects. Black Americans are 52% more likely than average to live in areas where a high risk for heat-related health problems exists, while Latino/a communities are 21% more likely to live under such conditions (Jesdale et al., 2013). Residents of neighborhoods that were formerly subject to “redlining”—a Federal practice that determined home lending risk based on racial composition—experience surface temperatures that are on average 2.6°C (4.7°F) and up to 7°C (12.6°F) hotter compared to their non-redlined counterparts in the same city, even more than 50 years after the end of this redlining policy; these higher temperatures are correlated with lower UTC (Hoffman et al., 2020). During extended heat waves in LA, mortality increases about five-fold from the first to the fifth consecutive day; after the fifth day, mortality risk increases 46% in Latino/a communities and 48% in elderly Black communities (Kalkstein et al., 2014).

Despite the growing threat of heat, effective approaches to alleviate urban heat do exist. These include risk mitigation strategies designed to facilitate institutional response during extreme heat events, such as heat alerts, as well as strategies that focus on reducing urban temperatures through measures such as increasing vegetative cover and nature-based solutions, improving building standards, and increasing access to air conditioning (Escobedo et al., 2019; Keith et al., 2020). Air conditioning access is an effective approach for regulating heat and subsequently protecting health, but it is not a sustainable practice in its current form because it generates climate-changing emissions and is often prohibitively costly for low-income households (Barreca et al., 2016). Tree planting is a well-documented heat mitigation strategy that has received increased investment in a growing number of cities around the world (Keith et al., 2020; Esperon-Rodriguez et al., 2022). Investments to increase UTC are understood to provide a range of co-benefits to urban communities such as: reduced urban heat through shading and evapotranspiration; reduced energy demand; carbon sequestration; improved air quality; improved water quality and supply through stormwater runoff management; provision of wildlife habitat; enhanced community cohesion; and improved human health and wellbeing (United States Environmental Protection Agency, 2011; Escobedo et al., 2019).

UTC has also been associated with reduced stress (Hartig and Staats, 2006; Van den Berg et al., 2010; Roe and Aspinall, 2011).

Trees mitigate heat by regulating climate conditions through shading and evapotranspiration, and these mechanisms can have a significant cooling effect—for example decreasing park air temperatures by up to 11°F in comparison to surrounding streets (Vanos et al., 2012). Studies modeling projected benefits of UTC in reducing temperatures demonstrate that mature UTC can facilitate exponential cooling for urban areas (Taha, 2013). Cooling at the micro scale also impacts energy demand because tree shade reduces building heat gain and shaded air conditioners work more efficiently (Akbari, 2002; Kendall and McPherson, 2012). Such heat reduction measures result in decreased cases of heat-related illness and death (Kalkstein et al., 2022).

However, the distribution of UTC and its co-benefits is affected by numerous factors ranging from biophysical conditions such as the necessity of supplemental watering in more arid climates, to socio-economic factors such as the potential for gentrification and displacement that neighborhood improvements like greening can potentially exacerbate (Checker, 2011; Wolch et al., 2014; Roman et al., 2015; Schwarz et al., 2015; Dawes et al., 2018; Riley and Gardiner, 2020; Volin et al., 2020; Donovan et al., 2021; Sharifi et al., 2021). Additionally, lower income and formerly redlined communities have greater amounts of impervious surfaces and are more densely developed, signaling increased barriers to community-driven tree planting initiatives, and requiring significantly greater investments and government coordination for capital improvements (CAPA Strategies for Los Angeles Urban Forest Equity Collective, 2021a,b).

Another complicating factor is that planting, maintenance, management, and preservation of UTC is complex. A broad range of actors—from local users to volunteers to professional managers—play a role in stewarding the urban forest (Krasny and Tidball, 2015; Roman et al., 2015). In LA, the responsibility for planting street trees falls on local government and non-profit organizations, but planting a tree is only the first step. Establishment care during the first 3–5 years must follow (Levinsson et al., 2017). Perennially underfunded UTC management can also exacerbate already entrenched distrust in historically disinvested neighborhoods and increase barriers to achieving urban forest equity, as tree-planting municipalities and organizations working in economically disadvantaged areas operate with limited resources (Pincetl, 2010). This reality exists even in environmentally progressive California, where the importance of greening is widely recognized and where carbon cap-and-trade and other state-administered funding streams produce revenues in support of local greening programs (Bekesi and Ralston, 2019).

In recent years, transdisciplinary frameworks have begun to be used to address the complexities that arise in such socio-ecological systems. For example, applied research in

disciplines concerned with the human dimensions of ecology and environmental management are using socio-ecological systems (SESs) frameworks to better understand the dynamics between social and ecological systems and how these can be used to improve understanding of pressing issues associated with sustainability, environmental policies, and climate change (Partelow, 2018). Such information and knowledge is necessary for effective climate change responses, as urban actors from community members to policy-makers increasingly find themselves adapting to extreme climate impacts to human communities and ecosystems (Ostrom, 2009). In the present context, environmental management and sustainability-based approaches frameworks traditionally used by urban ecologists, foresters, landscape architects, horticulturists, and planners for evaluating tree planting programs (i.e., Ko et al., 2015; Roman et al., 2015) are often insufficient in addressing human wellbeing outcomes because of their focus on biophysical metrics and objectives (i.e., UTC goals, planting a specified number of trees, or minimizing tree mortality). But urban ecosystems and forests are complex and should also include the socioeconomic, human wellbeing, and public health metrics and objectives such as ecosystem service co-benefits and the social and political dynamics involved in urban greening (Dawes et al., 2018). Such metrics, objectives and dynamics can span scales from individual-level human and tree factors such as human self-efficacy and tree survivorship, to societal and UTC level such as policy and governance formulation and watershed quality. They also span temporal factors, such as who should be responsible for maintaining street trees planted in the public right-of-way space in front of a residence over a tree's life span regardless of changes in government or property ownership and whether that responsibility is understood and acted upon by different stakeholder across time. An approach that also focuses on these social, economic, political, and public health factors across space and time is therefore needed (Escobedo et al., 2019).

Socio-ecological frameworks that include those factors are used by disciplines in the medical science and public health fields (Palafox et al., 2018), and thus warrant further consideration because of their focus on desired outcomes (i.e., improvements to human wellbeing, public health outcomes, and climate equity) as opposed to the planting and caring for trees as an intermediate process of activity to indirectly or subsequently advance urban forest equity and climate equity. This differs from SESs frameworks traditionally used in the previously mentioned environmental management and sustainability fields because those frameworks are concerned with understanding the ecology-society nexus (i.e., governance and natural resource conditions) as opposed to tailoring processes to optimize human wellbeing outcomes (e.g., improved public health and other co-benefits) (Golden and Earp, 2012).

More specifically, in public health disciplines, socio-ecological models are used to elucidate complex dynamics by nesting factors into individual, relationship, institutional,

community, and society levels that depict the relational dynamics between them (Golden and Earp, 2012). This approach has been widely used in public health campaigns including in promotion of physical activity, involvement in grandparenting, cancer prevention and control, and violence prevention, and its use is promoted by the Centers for Disease Control and Prevention (Palafox et al., 2018; Centers for Disease Control and Prevention, 2019; Shorey and Ng, 2022). SES models often used in public health disciplines could hypothetically be used to capture key drivers that influence tree stewardship and planting programs. Furthermore, informed by a mixed-method approach, the use of such alternative transdisciplinary frameworks could also be used in other environmental management problems to identify evidence-based determinants and to understand the relational dynamics between them and desired outcomes.

In this study, we present such an approach with the aim to apply a socio-ecological framework from the public health field to evaluate a tree stewardship program in the City of Los Angeles, US. The specific objectives are to:

1. Evaluate the effectiveness of a tree stewardship training and community organizing program in advancing urban forest equity and public health.
2. Identify principal barriers and determinants (e.g., policy, infrastructure, social) encountered by trainees in their communities, which hinder or aid the advancement of urban forest equity.
3. Build a socio-ecological framework to understand the spheres of influence (or levels) within which these factors exist and how the dynamics between them interact.

We then use these objectives to discuss how this novel approach and framework can be used to better inform funding, management, planning, policies, and governance of UTC to maximize equity and public health goals.

## Materials and methods

We evaluate a community and volunteer-based tree stewardship initiative—the Tree Ambassador, or *Promotor Forestal*, program—as a case study. This new English/Spanish bilingual community organizing initiative launched in 2021. The program provides 10 months of paid training to residents to mobilize their community to plant and care for trees and increase resilience around heat-health risk in historically disinvested neighborhoods in Los Angeles. The goal of the Tree Ambassador Program is to create a trained group of community members that can build connections with and amplify the voices of their communities to achieve urban greening goals. Tree Ambassadors, or *promotores*, attend monthly training sessions with expert instructors and work closely within urban forestry

organizations (or “host organizations”) in order to gain the tools, knowledge, and connections needed to increase UTC and community resilience in select marginalized neighborhoods. The program was intentionally modeled after the community health workers, or *promotores de salud*, approach (Scott et al., 2018; Centers for Disease Control and Prevention, 2019), signaling the significance of the application of an SES framework. The community health worker model trains lay people who are trusted members of a community or who have a deep understanding of the community to serve as frontline public health workers (American Public Health Association, 2021). The Tree Ambassador model seeks to mitigate potential for green gentrification (Donovan et al., 2021; Sharifi et al., 2021) by directly compensating and empowering local leaders where they live, work, and play, instead of relying on volunteerism, which often assumes time affluence and excludes residents who work multiple jobs or have family or community responsibilities that preclude regular participation. The first training cohort was composed of 12 Tree Ambassador (TA) trainees who completed the program.

This community-based tree planting partnership is led by City Plants—a non-profit organization that oversees public-private tree planting partnerships in Los Angeles—together with the City of Los Angeles, state, federal, and international urban and community forestry agencies (the LA Department of Water and Power, the California Department of Forestry and Fire Protection, the USDA Forest Service, and Ecosia), and local tree planting organizations (Climate Resolve, Koreatown Youth & Community Center, and TreePeople). Using surveys, focus groups, and ethnographic data collected through April 2022 with this first training cohort, we first evaluate the program and then use the findings to apply and adapt a socio-ecological model of community-based tree stewardship for improved public health outcomes.

## Los Angeles, CA, US and the Tree Ambassador Program case study

Los Angeles, CA is the second-largest city in the US by population, with an ethnically diverse population of 3.9 million people who are 48% Latino/a, 29% white, 12% Asian, and 9% Black; 36% of residents are foreign born (United States Census Bureau, 2021). Median household income was \$65,000 in 2020, and 17% of residents live in poverty, with high socio-economic variability between neighborhoods. The City of LA has an area of 468 square miles and an average population density of 8,100 people per square mile (United States Census Bureau, 2021). Located in a Mediterranean climate, LA is both flanked and bisected by mountain ranges, and the region surrounding the city consequently hosts a variety of smaller climate zones ranging from coastal, to high desert, to montane—with varying seasonal

temperature and precipitation averages ranging from 125 mm (5 in) to over 750 mm (30 in) (Hall et al., 2018; Los Angeles County Department of Public Works, 2021). The City of LA has one mayor and 15 city councilmembers, each who oversees aspects of city services in one of 15 council districts and is responsible for enacting ordinances that are subject to mayoral approval or veto (City of Los Angeles, 2022).

Our study area and evaluation focused on 9 neighborhoods and 12 Tree Ambassadors representing several City of LA neighborhoods (Table 1). Each neighborhood was selected with consideration to factors including income, high concentration of minority residents, and heat vulnerability as determined by heat-related deaths. See Supplementary Table 1 for details on the socioeconomic and demographic composition of the Tree Ambassadors.

## An overview of tree planting programs in LA

In 2007, under the leadership of newly-elected Mayor Antonio Villaraigosa, the City of LA launched Million Trees Los Angeles (MTLA), a private-public partnership designed to rely on non-profit partners to plant trees and help raise the funds necessary to do so (Pincetl et al., 2013). The MTLA initiative had mixed results. It received a fair amount of attention in the media and among LA residents, but clearly fell short of its million-tree goal, succeeding in planting an estimated 400,000 trees (City Plants, personal communication, June 4, 2021). MTLA set out to address tree inequity, but in practice plantings occurred opportunistically where private-public partnerships could be established (Pincetl et al., 2013). Lower-income communities were found to receive relatively fewer trees due to a perception that more UTC provides more spaces for criminals to hide, creating reluctance in some neighborhoods (Pincetl, 2010). An opt-in process for requesting a tree required a signature, which discouraged residents in communities with many immigrants, multi-family homes, or high rentership (Pincetl, 2010). In 2014, Mayor Eric Garcetti rebranded MTLA as City Plants, and the organization has since adopted a tree planting and care strategy of “right tree, right place, right reason.”

More recently, the City of LA's *Green New Deal*, a 2019 update to the City's *Sustainable City pLAN* first published in 2015, calls for increasing tree canopy in disadvantaged communities by 50% in time for the 2028 Olympics in Los Angeles (City of Los Angeles, 2019). Considering the urban forest of the City of LA is composed of ~10.8 million trees (McPherson et al., 2011), increasing tree canopy by 50% is an ambitious goal and will require significant investment and resources. To facilitate achieving these and other urban forestry goals, in 2019 the City of LA hired its first-ever City Forest Officer to oversee citywide coordination in support of these goals (Los Angeles Daily News, 2019).

These developments are critical because in LA, UTC has been documented to have an effect on public health outcomes

and environmental benefits. Higher UTC lowers ambient temperature, with LA city blocks that have more than 30% UTC being about 2.8°C (5°F) cooler than blocks without trees (Pincetl et al., 2013). In the city, the percentage of shaded UTC over the city's streets accounts for more than 60% of land surface temperature variations, compared with only 30% of variation being explained by factors such as topography and distance to the coast (Pincetl et al., 2013). Increasing UTC and albedo of roofs and pavements in LA can reduce heat-related mortality by upwards of 25%, especially in low-income communities and communities of color (Kalkstein et al., 2022). Interventions of higher UTC and albedo also have the potential to delay climate change-induced warming ~40–70 years under business-as-usual and moderate mitigation scenarios, respectively (Kalkstein et al., 2022). Investing in UTC thus has the potential to increase LA's resilience to climatic changes.

## Mixed methods approach

Having described the Tree Ambassador Program and LA's context in the previous section, we now present how we used a mixed methods approach—commonly used in SESs research—to obtain a comprehensive picture of Tree Ambassadors' experiences and accommodate different avenues for them to provide feedback. Such an approach will allow for results to be analyzed thematically and longitudinally. Results from the multiple methods can also be triangulated to derive richer data, address the goals of the research more comprehensively, and confirm results (Wilson, 2014). Results can then be used to adapt available SES models used in the public health fields, addressing the aims of this study.

## Focus group

A focus group ( $N = 9$ ) was held on November 21, 2021 to provide an opportunity for Tree Ambassadors (TAs hereafter) to have their perspectives heard and inform the structure and content of the program. The focus group was held during the sixth of 10 months of training, and was held in an office building in Los Angeles.

All TAs present at the training were invited to voluntarily participate. In total, nine TAs participated. The focus group was held during the last hour of a 3-h training session and participants received a verbal consent that explained that their participation was voluntary, and that any information gathered during the focus group would be treated as anonymous. Attendees were also advised that anyone not wishing to participate could leave or sit back and listen without participating, and that non-participation would not result in any penalty.

The focus group was facilitated in English by the authors using a script (Supplementary Table 2). Simultaneous



TABLE 1 Tree Ambassador neighborhood characteristics.

Tree Ambassador	Neighborhood	% Existing Tree Canopy*	Pollution burden Score**	Heat health action index***
1	Westlake	13%	90	79
2	Pico Union	8%	97	70
3	South LA	10%	89	75
4	South LA	12%	85	77
5	Boyle Heights	13%	87	81
6	Boyle Heights	13%	71	74
7	Canoga Park	26%	68	55
8	Canoga Park	26%	93	64
9	Pacoima, Sylmar	18%	97	61
10	Sunland-Tujunga	26%	67	43
11	Sun Valley	30%	87	54
12	North Hollywood	20%	95	50

\*By ZIP code, or numeric average where a neighborhood is made up of multiple ZIP codes, <https://www.treepeople.org/los-angeles-county-tree-canopy-map-viewer/>.

\*\*Percentile by census tract, with values from 0 to 100 by census tract. Higher values mean higher proportion of disadvantaged individuals per CalEnviroScreen metrics, <https://oehha.ca.gov/calenviroscreen/report/calenviroscreen-40>.

\*\*\*Represents heat vulnerability with values from 0 to 100 by census tract. Higher values mean higher heat vulnerability, <https://cal-heat.org/explore>.

translation in Spanish was also provided by the authors so that TAs with limited English proficiency could participate in the discussion. The focus group was audio recorded. Three note-takers took live notes and notes were subsequently triangulated. A transcript of the focus group was created using the audio recording and notes taken by three note-takers. The transcript was coded using content analysis, and data were then coded and analyzed thematically. The results were used to develop the following survey instrument using the Total Design Method (Lavrakas, 2008).

## Survey instrument

A mid-program survey ( $N = 11$ ) was conducted electronically using SurveyMonkey following the focus group, between the sixth and seventh training sessions. The survey instrument was provided to the respondents in both Spanish and English language and responses were received between December 6 and 13, 2021. Respondents were first asked to provide anonymous identifiers to allow for an individual's responses to the second survey to be analyzed longitudinally. The survey instrument (Data Sheets 1, 2) contained 33 questions (multiple-choice, Likert scale, matrix, and open-ended) to capture the respondent's knowledge, perception, beliefs, and attitudes (Gifford and Sussman, 2012) related to the following themes: content, structure, and pace of the trainings; program materials and support they have received as trainees; and characteristics about the community in which the respondent lives and works.

An end-of-program survey ( $N = 8$ ) was conducted at the conclusion of the program with TAs who had previously responded to the first survey. The survey instrument was once

again provided in both Spanish and English language; responses were received between March 28 and April 7, 2022. TAs were specifically asked to provide feedback about various aspects of the training program, including whether trainings: were easy to understand; covered material relevant to their communities; prepared TAs to plant and care for trees; were too slow or fast; had an appropriate level and amount of content; and allotted too little or too much time to learning by listening vs. learning by doing.

Data were cleaned and formatted in MS Excel and then analyzed with Student's  $t$ -test in R version 4.0.2 (R Core Team, 2020). Specifically, paired sample  $t$ -tests were used to check for significant differences between means for the questions asked in both the mid-program and end-of-program surveys. For knowledge-based qualitative questions, word clouds were created to visually display the key answers and their relative frequencies. The word clouds were made online using <http://www.wordclouds.com>. For qualitative questions that were focused on providing feedback, responses were analyzed in steps. The first step was to look for responses in both mid- and end-of-program surveys that were the same in response content. Then the remaining responses were summarized to facilitate analyses. For the qualitative responses from both surveys, the responses for the end-point survey were sorted by comments that were also provided on the mid-point survey, and those that were new.

## Ethnographic observations

Ethnographic observations were made during different event types during the program: training sessions, TA meetings with their host organizations, informal weekly TA "hangout" meetings held via Zoom that gave TAs an opportunity to



discuss progress and ask question, program team meetings, tree planting events, and tree adoption events organized and supported by TAs between July 2021 and April 2022. The events ( $N = 20$ ) provided a wide variety of settings and conditions for observations through the multiple phases of the program as TAs moved from training to community organizing and holding their own community events. We note that the training program took place during the COVID-19 pandemic, and the initial training sessions were held remotely via Zoom. Some events were thus limited to observations that can be made in digital spaces. Some of the events held remotely included the use of Zoom® chats or web-based Audience Response Systems such as Mentimeter® (Mohin et al., 2020), resulting in additional collection of opinions and feedback which were considered formative evaluation feedback available for incorporation into the remainder of the program. Prompts used during remotely-held events were presented in Spanish and English, and included: “What would you like to learn as a Tree Ambassador?”; “What are your goals before the end of the program?”; “How have you grown or been challenged during the program?”; and “What specific skills or knowledge have you gained as an Ambassador?” Typically, in-person events yielded more engaged interactions among participants and more opportunities to observe the dynamics at play, resulting in richer notes. In addition to observations, several events included opportunities to speak with the TAs and program staff to ask follow-up questions and obtain additional insights.

## Results

### Focus group

The themes that emerged during the focus group are presented in Table 2. The primary themes were: (1) that TAs are motivated by a desire to serve as change agents for their communities and the Tree Ambassador Program provides them an avenue to act on that desire; and (2) that TAs face a variety of challenges—some of which are deep-rooted and intractable—as they try to convince members of their communities to engage in tree stewardship. With several months of training remaining in the program and after the focus group, themes that emerged were incorporated into subsequent training materials. Outreach methods and materials that the TAs were given to engage the community were also tailored accordingly. For example, outreach materials were redesigned to include an image of an unshaded street in the neighborhood against a street that is shaded by a canopy of trees, and paper forms were made readily available to decrease the reliance on internet sign-ups. TAs were also provided with information about how to navigate the process of removing concrete or pavement to create tree planting wells where planting spaces are not available, which is a

common barrier in historically redlined neighborhoods (CAPA Strategies for Los Angeles Urban Forest Equity Collective, 2021a).

### Surveys

Overall, survey findings point to increased TA confidence, knowledge, and care as it pertains to TAs’ relationship with trees and with their community, but a corresponding decrease in the TAs’ perception of how much other community members care for their neighborhood (Figures 3–7). TAs felt moderately or highly prepared to plant and care for trees but indicated that there is room for improving the program in terms of content and format (Figures 1, 2). Another key finding is that despite considerable effort, securing street tree applications, requiring a signed form commitment to water by a tenant or property owner, was very difficult, especially compared with yard tree applications for private property trees (Table 3).

Figures 1, 2 show that all but three of the means decreased from the mid-program of the program to the program’s end; while two items were the same (whether the TAs feel prepared to care for young trees, and how much time was spent listening to presentations vs. learning by doing); and only one increased (feel prepared to care for mature trees). However, none of the differences were statistically significant, most likely due to the small sample size. The results suggest that the training in the second half of the program was not as well-received and should likely be the focus of any changes for the next year.

The TAs responded about skills or knowledge they gained in their time in the Tree Ambassador Program that can be used to benefit their community (Figure 3). As shown in Figure 3, skills related to “community” were the top-mentioned responses. This includes how redlining has impacted communities, advocacy, community organizing, establishing community connections, and community leadership. These skills are transferable to other programs and subject areas. Skills directly relating to trees—how to care for them, when and where to plant them—were the second most mentioned goal. These skills are specific and are of more limited use. Other skills mentioned included communication, relationship building, and connecting small businesses with non-profit programming.

Tree Ambassadors were also asked the following question about their career goals during the end-of-project survey: “Would you like to pursue a career in urban greening or related field? Please share your thoughts. If you are not interested in pursuing a career in this field, do you think this program has prepared you for future careers in other fields? If so, how?”

Six TAs indicated an interest in pursuing a career in urban greening or related fields; one said no but noted “I like having the information on how to help the community”; and one was unclear. The TAs were then asked to provide feedback on the program materials and their confidence in attaining

TABLE 2 Content analysis of Tree Ambassador focus group in Los Angeles CA, US (N = 9).

Themes	Tree Ambassador comments
Seeing oneself as an agent of positive community change	<p>“I wanted to put in my energy and activism and advocacy through community organizing and talking to people. I wanted to gain more formal community organizing skills.”</p> <p>“I was actually really skeptical when I first heard about the program. I thought that no one would be interested in my community. But then after thinking about it, I thought, ‘Has anyone tried to talk to our community?’ Maybe there’s a reason they’re not interested. Maybe they don’t know or they don’t think they have the time.”</p> <p>“I see the benefits of trees in other places and thought that was missing and so wanted to bring that to my own community—this is also an environmental and social justice issue.”</p> <p>“When I found out about this program I thought it was an additional service that I could be part of to help to uplift the community.”</p>
Challenges encountered: Urban greening not a priority for some in the community	<p>“Part of the challenge of getting people to get trees is that there is a long list of priorities that people want to have fixed and trees are not at the top of that list. Even if they are free it’s still a responsibility that they need to take, and people are just frustrated. It’s harder to push for trees when people feel like there are speed bumps or sidewalks or all these other issues that they feel that the city should take care of.”</p> <p>“When we ask people in the community, we receive more noes than yeses. When we ask them if they want trees, they’d say, ‘No, we just want speed bumps, so that people can walk and run.’ They’re not interested in trees.”</p>
Challenges encountered: Cynicism about local government among community members	<p>“I think another one of the barriers is that the city in general, historically has taken a long time to get things done. Even getting potholes fixed takes forever. That’s a big concern with people in the community. Working with the city just takes forever to complete anything or even take initiative, so they just give up just because they don’t think it will ever happen.”</p> <p>“I saw someone describe it as about tree planting guerilla warfare. That would be like them just going out in the street and planting trees in whatever spot people see available. People don’t want to work with the city because there is too much red tape.”</p> <p>“An older disabled person said that the tree had ruined the sidewalk, and he had spent money removing the tree and fixing the sidewalk. They reached out to the city to get it addressed but the city didn’t do anything so they weren’t willing to take a tree.”</p> <p>“One of the residents said that she signed up but never got a tree even though neighbors got trees.”</p>
Challenges encountered: Spatial and physical barriers	<p>“Apartments, especially those that don’t have access to residents directly, where they have a gate... is difficult because we don’t have access to the residents.”</p> <p>“In my neighborhood we don’t have many sidewalks.”</p> <p>“One of the barriers I’ve heard is that people are interested in getting trees but don’t have a car.”</p>
Challenges encountered: Internet access and digital literacy	<p>“Outreach materials are mostly email based, and for some people that’s not accessible... Even for registration links... this interferes with some people not being able to access it.”</p> <p>“As soon as we say something about the internet process, they say, ‘Oh no we don’t want to deal with it.’ They don’t want to subscribe. They don’t want to have to deal with the internet.”</p> <p>“Older Hispanic communities don’t want to deal with the internet.”</p> <p>“Some people don’t know how to navigate the internet, they don’t know how to use a computer.”</p>

program goals. Goals for trainees included securing 30 street tree applications with a commitment from adjacent property owners or tenants to water the tree; securing 30 yard tree applications from community members; hosting at least one tree adoption event; and hosting at least one additional community volunteering event such as a tree planting or tree care event. Figure 4 shows that scores for all but one question (“The program materials I received help me engage my community meet my community’s needs”) increased from the mid-point to the end-point. None were statistically significant, most likely due to the small sample size. Their relative scores at the mid-point corresponded fairly well to whether or not TAs ultimately met that goal. Confidence in securing street tree applications

was lowest, and this goal was ultimately met by only one TA. Conversely, confidence was highest for private property trees and hosting tree adoptions, and these goals were met by the most TAs. Finally, none of the means were 6 or above, and most were under 5, indicating that there is room to improve the program to better meet the trainees’ needs.

The end-point survey asked TAs whether they were able to achieve the program goals (Table 3). Street tree applications—requiring a signed commitment to water form—were the most difficult to secure.

The TAs’ self-reports via the survey are in line with the program metrics compiled by the host organizations and City Plants. Altogether, TAs planted or distributed a total of 1,929

**TABLE 3** Responses to the question “Were you able to achieve the following program goals?”

Goal	Yes	No
Secure 30 or more street tree applications	17%	83%
Secure 30 or more private property yard tree applications	71%	29%
Host at least one tree adoption	86%	14%
Host at least one tree community volunteer event	43%	57%

trees—only 53 of which were street tree applications, making up <3% of the total, despite considerable effort. TAs canvassed an estimated 1,244 residents and held over a dozen events including tabling at places of worship and neighborhood meetings.

The TAs were asked to list both benefits and problems that they believe trees can bring to their neighborhoods. [Figure 5](#) compares the mid-point and end-point responses around benefits that trees bring. At both timepoints, the mental and physical health benefits of trees were noted most often. At the mid-point, “biodiversity” was quite prominent, whereas at the end-point “beautification” was similarly prominent. In both surveys, TAs highlighted how trees improve air quality. They also used the words “reducing” and “lowering” often: reducing heat, lowering energy bills and lowering air conditioner use. Shade and biodiversity were each mentioned a few times; and one TA noted at the mid-point they can help avoid summer power outages.

At the mid-point of the program, Ambassadors most often noted the negative effect trees can have on sidewalks as a problem ([Figure 6](#)). The words “maintenance” and “people” also showed up often, suggesting the problems were not due to the trees themselves but people not wanting the maintenance required of trees. The most prominent theme at the end point was the risk that trees become neglected and not watered. Leaves and branches falling from the trees were mentioned at both points, but not as often as other problems. The word “parkway”—the planting strip between the sidewalk—also appears in comments related to competition with utility poles and limited city resources for providing tree care in this space.

Finally, the TAs were asked several questions about their neighborhood. [Figure 7](#) shows that responses to all but one of the questions went in a positive direction from mid-point to end-point, although none were statistically significant. TAs reported caring about their community more, knowing more neighbors, and being more comfortable asking neighbors (both neighbors they know and those they do not know) for favors. An explanation could be that the canvassing, tabling, and other activities TAs undertook in their neighborhoods enabled them to interact with and get to know more

members of the community. Comments from TAs captured via the ethnographic observations ([Supplementary Table 3](#)) also support these findings. However, the opposite was reported for other people caring about the neighborhood, as there was a decrease in the mean score. An explanation could be that a high number of refusals and difficulties in getting people to commit to water street trees (which benefit more than just the household) made TAs think that other people in the community did not care about the neighborhood. There was also a slight uptick in the response to the question about whose responsibility it is to prepare for disaster (1 = 100% mine, 7 = 100% the government’s), though the mean in both time periods indicates that respondents feel responsibility lies somewhere in between.

Aside from the responses to open-ended questions that were illustrated in the word clouds in [Figures 5, 6](#), additional key insights from TA highlight the conditions and challenges faced in the process of trying to increase UTC in their communities. Here we share a small selection of those insights, which raise issues such as availability of planting spaces, the presence of homeless encampments, awareness of historical injustices, and the challenges of organizing in neighborhoods with high rentership.

Responses to the question “Do you have any comments or recommendations about the materials you have received to help you engage with the community?” included:

“A lot of the material is predicated on availability of space and the assumption that there is a pre-existing community bond within the neighborhood. Although Los Angeles does not have the typical urban spaces that other cities may have, areas with high population of immigrants, low percentage of homeowners/private property, large homeless encampments, and other issues regarding financial, social, and environmental conditions should be taken into consideration in order to create a more intersectional approach.”

“I think asking people of the impacted communities if they are aware of the environmental inequities in LA or their community and what impacts might that cause in their community can help gauge how aware a community is about these topics. I think asking them what impact/problems that inequity could create in their communities can bring more awareness and have them thinking about these topics and motivate them more to engage with their community. I never knew about redlining until just recently. Learning about it, I was shocked and angry. But I finally had an answer for why my community wasn’t as well-resourced as wealthier areas. And why these affected areas continue to remain affected, being stuck in a cycle. I feel like not knowing about redlining, the environmental injustice/inequity in certain communities, etc. made me oblivious or ignorant about the issues they cause. Living in an apartment, I don’t even have space for a tree so I wouldn’t have even passed by

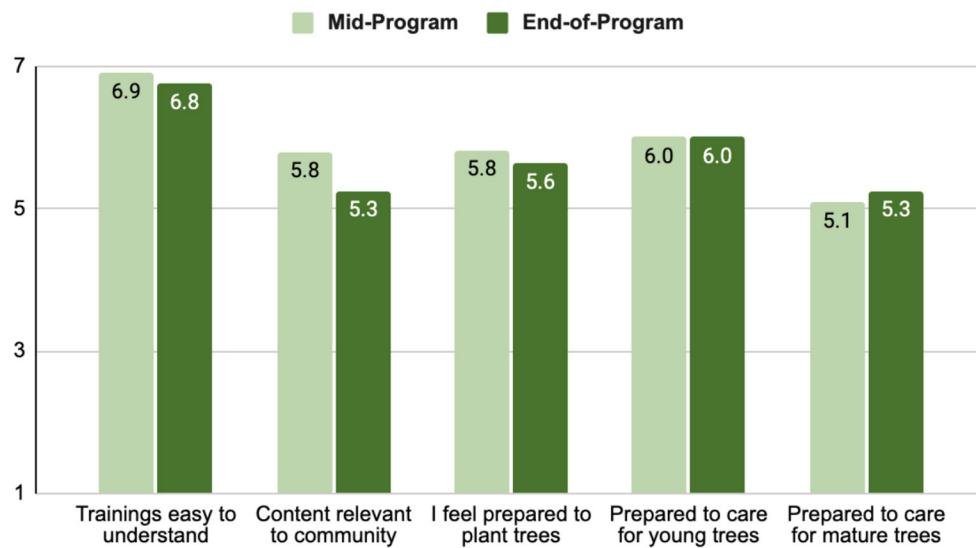


FIGURE 1

Responses regarding the trainings and Tree Ambassador readiness (1 = strongly disagree; 7 = strongly agree).

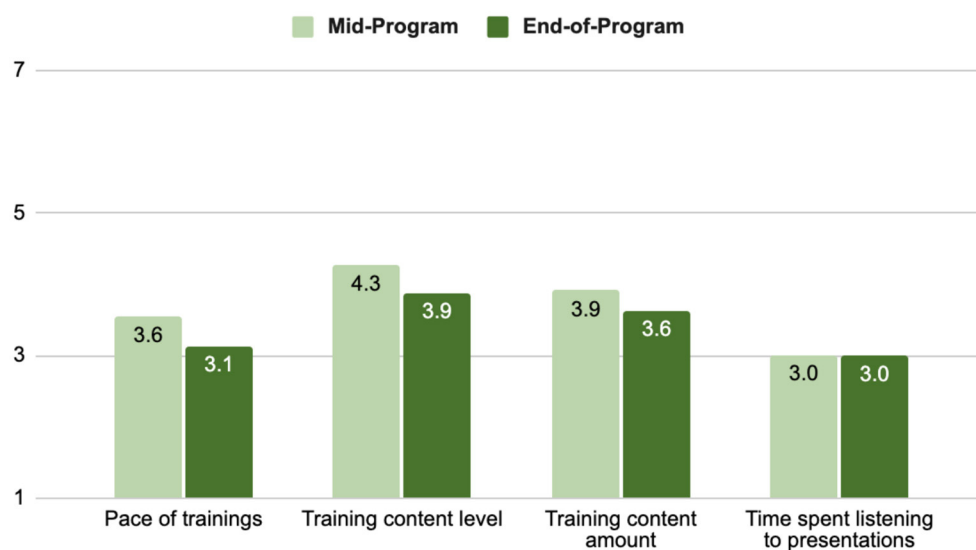


FIGURE 2

Responses regarding the pace and structure of the trainings (1 = pace too slow, level too simple, content amount too little, too much time spent listening to presentations; 4 = about right; 7 = Pace too fast, level too complicated, content amount too much, too much time spent learning by doing).

a tree distribution event. I never would've cared as deeply as I do now without knowing these injustices first, because now I can understand the significance of planting a tree."

Responses to the question "Do you have any other comments or recommendations about how to improve the Tree Ambassador Program?" included:

"I've felt very supported by my organization but I do wish there was a bit more support from the city. Reaching out to city officials to spread the word and let residents know sounds like a very reasonable thing to ask for. Private property trees are by far the easiest to get forms signed for and that's great, but I think providing Tree Ambassadors with more resources or knowledge to navigate spaces that



**FIGURE 3**  
Word cloud exhibiting the skills and knowledge learned by Tree Ambassadors during the program that can benefit the community.

don't have as much private property like commercial, industrial, apartment zones, would be very beneficial. These areas tend to lack trees and would greatly benefit from them but it's harder to navigate because of the obstacles (planting on the parkway of an apartment: technically city property but easiest and safest to get permission from property manager- can be tricky)."

"Different areas necessitate different methods. A lot of people who are recently immigrated and/or living in a rented space may view their current residence as a temporary space and therefore be disinvested in larger community needs. Trees are a long term investment, in which the immediate benefits may not be entirely obvious. If a neighborhood is seen as a transitional point, residents may be disinvested in the betterment of the community."

## Ethnographic observations

Ethnographic events spanning the 10-month period of the first training cohort—from hiring to training, and graduation—show themes that both complement and augment the findings emerging from the focus group and surveys. Specifically, as TAs gained knowledge, skills, and confidence via the program, this led them to forge new partnerships in their community and organize successful community events such as tree adoption events. Findings are presented in [Supplementary Table 3](#). Among the themes that emerged: TAs experienced significant challenges in engaging their communities in urban greening, spanning from cynicism about the City's follow-through and perceptions about the high cost of watering a tree, to the

inability to interact with people in person due to factors such as front gates and concerns around potential COVID-19 exposure. Some TAs modified their engagement methods and reported more canvassing success when canvassing focused on inviting neighbors to attend free community tree adoption events rather than on trying to convince them to sign up for a free tree at their doorstep. Findings from the ethnographic events are also incorporated into [Supplementary Table 4](#).

## Socio-ecological model

The above approach and our findings identified multiple factors that influence community-based tree stewardship. However, the nexus between tree stewardship programs, UTC co-benefits, and public health outcomes is still not clear and warrants exploration. Accordingly, using our findings from Sections Focus group, Surveys, and Ethnographic observations we developed a socio-ecological framework to better elucidate the factors associated with tree stewardship encountered by individuals intervening to address urban forest inequity in their neighborhoods. Specifically, we adapted a model frequently used in public health ([Golden and Earp, 2012](#); [Palafox et al., 2018](#)) as well as results from our evaluation to better identify factors that relate a tree planting program to positive health outcomes and is shown in [Figure 8](#).

We did this by reviewing the themes that collectively emerged from the focus group, surveys, and ethnographic observations. We evaluated the list of factors by first considering whether the presence of a given factor—e.g., high trust in local government, belief that trees cause problems, or availability of planting spaces—should be considered a support or an impediment upon a Tree Ambassador's efforts to foster tree stewardship among community stakeholders. Evaluating each factor through this lens allows for the development of interventions designed to either boost that factor as a benefit or reduce its presence as a barrier ([Golden and Earp, 2012](#)). For example, if the belief is prominent that leaf litter from trees is a problem, a Tree Ambassador's outreach can be modified to focus on how species selection (e.g., planting evergreen trees) and can avoid this problem down the line. We then categorized each factor into a level of influence ranging from individual to society level to reveal at what level interventions to address each factor should be focused. For example, individual level interventions should aim to change the knowledge and awareness of the individual, while institutional interventions should aim to create change in social relationships and organizational environments that support those individuals.

The result is a "Socio-ecological model of community-based tree stewardship" based on our approach and factors ([Figure 8](#)). [Figure 8](#) models the process of participation in urban forest management via tree adoption, committing to watering new trees, and other actions involved in planting and caring for



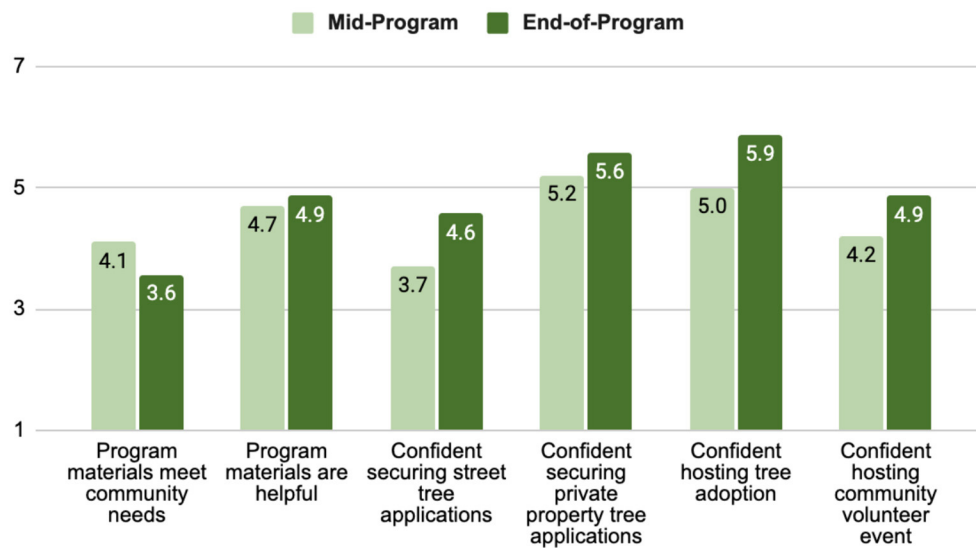


FIGURE 4

Responses regarding program materials and goals at the mid-point and end-point of the training program (1 = strongly disagree; 7 = strongly agree).

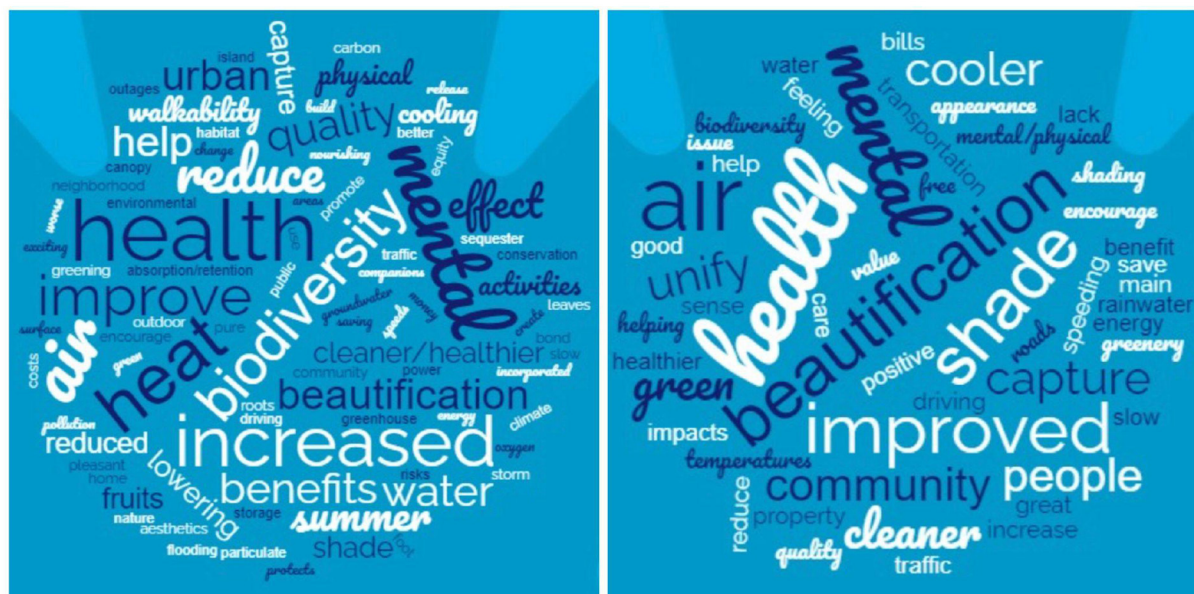


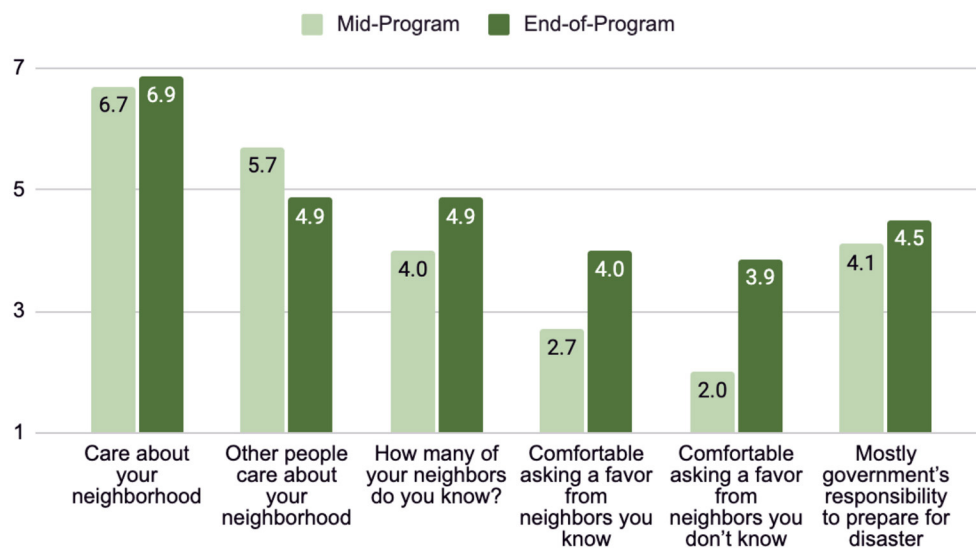
FIGURE 5

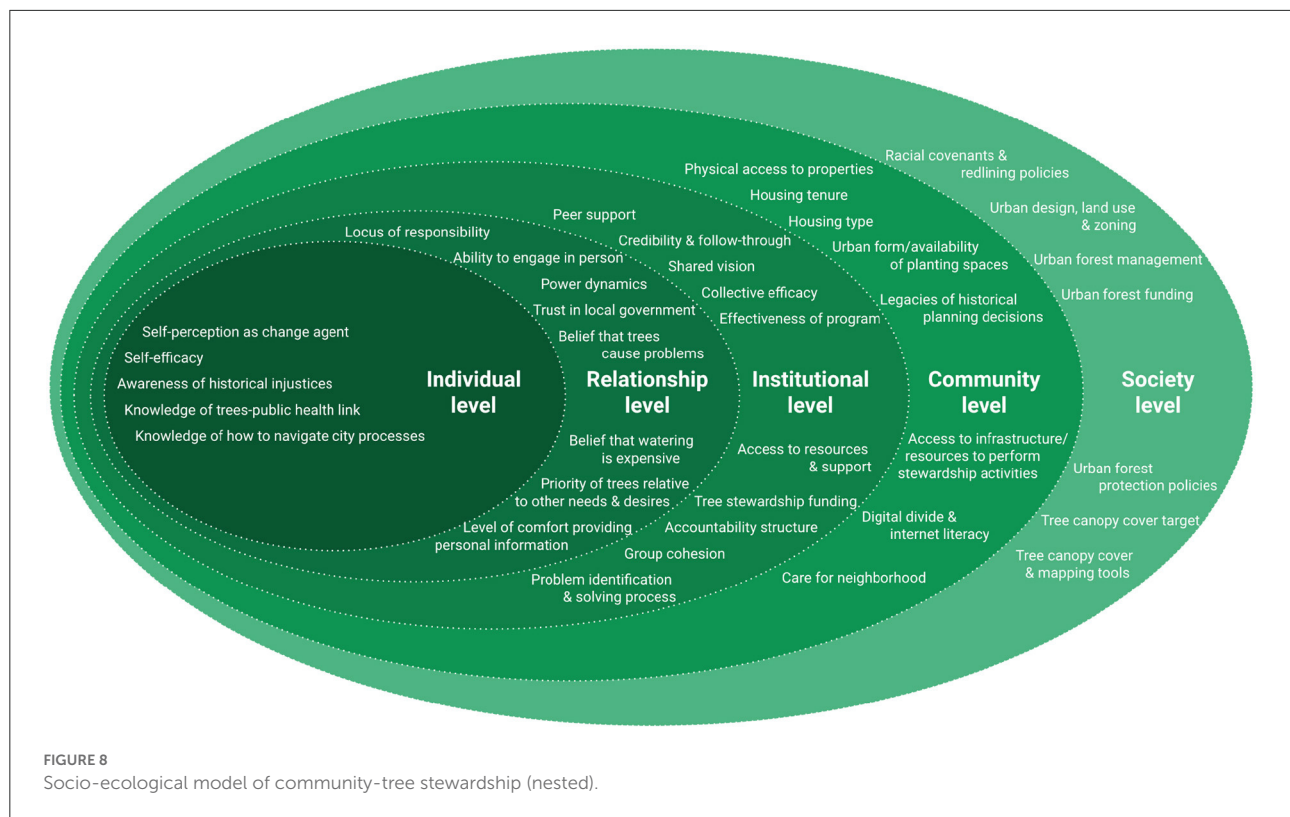
Responses to the question "List any benefits that you believe trees can bring to your neighborhood" in (left) mid-program survey and (right) end-of-program survey.

trees. Tree stewardship involves dynamic interactions between individuals and the social and political conditions and contexts that surround them. The model describes factors at each of five different levels—individual, relationship, institutional, community, and society. Community-based tree stewardship

is affected by this complex range of influences and nested interactions. The model recognizes that factors can cross between multiple levels, and we thus include nested dotted lines separating each layer of the model. They can also influence tree stewardship in different ways—either aiding or







community. These include drivers related to awareness, knowledge, and self-perception.

### Relationship level

Relationship level factors are those an individual working to affect tree stewardship may encounter as they attempt to engage with their neighbors or other members in the community. These factors may either aid or hinder their efforts and include drivers such as whether a community member prioritizes trees relative to other needs or desires for their neighborhood, and whether they are comfortable providing personal information.

### Institutional level

Institutional level factors are those that may be present or absent at the institution that is supporting an individual who is actively working to affect tree stewardship in their community—such as a non-profit or community organization, or a city agency. Collective drivers such as a shared vision, group cohesion, and the belief that the group can produce desired results are among these. Other drivers relate to support, follow-through, and processes to identify and address problems as they arise.

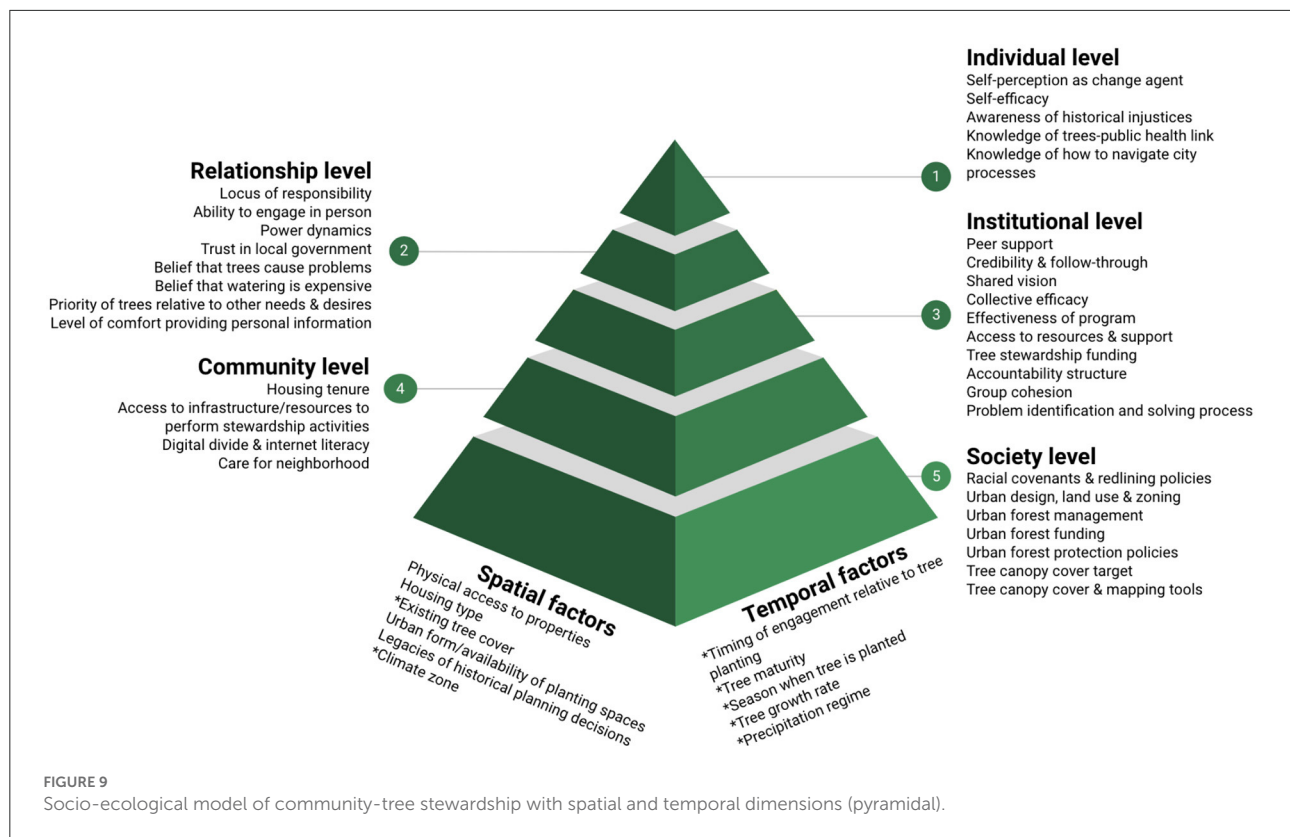
### Community level

Community level factors are neighborhood characteristics that may aid or hinder an individual's efforts to affect tree stewardship. These include physical attributes such as availability of planting spaces and access to properties to conduct canvassing. These also include indicators, such as whether a home is tenant- or owner-occupied, the level of internet literacy present in the community, and the level of care that a resident believes other community members have for the neighborhood.

### Society level

Society level factors include elements in the decision-making and information-access realm which occur at a level beyond the community—such as at the municipal, state level, or federal level. These include historical drivers such as redlining, and current drivers such as the presence of robust urban forest management and funding, public tree maintenance, UTC targets, and tree protection policies.

The nested model in [Figure 8](#) reveals the primary factors that hinder or aid tree stewardship efforts and the levels at which these occur. We offer an alternative model ([Figure 9](#)) that takes these factors and levels into account, and adds two additional dimensions: time and space. Temporal and spatial considerations also influence the success of any efforts



to advance urban forest equity. Some of the factors used in Figure 8 are moved from one of the five levels and placed under either spatial or temporal factors (for example, physical access to properties and housing type are moved from “Community level” to “Spatial factors”). We add several additional factors not captured in the nested model of Figure 8, which emerge when considering how spatial and temporal dimensions affect tree stewardship. The additional factors are marked with a \* in the figure.

Additional spatial factors include:

**Existing tree cover:** The existing UTC of a neighborhood can influence the willingness of community members to support additional UTC. Social ties and a sense of community have been shown to be stronger in apartment buildings with more vegetative cover compared to those without (Kuo et al., 1998), and these factors can in turn influence civic engagement in urban greening (Krasny and Tidball, 2015).

**Climate zone:** In LA’s semi-arid Mediterranean climate, summers are warm and dry, and rain is uncommon between late spring and fall, meaning a moisture deficit is likely to occur absent supplemental irrigation (Levinsson et al., 2017).

Additional temporal factors include:

**Timing of engagement relative to tree planting:** Engaging community members in the act of tree planting rather than after a tree has been planted enables residents to witness the difference of their efforts, boosting self and collective efficacy

while reducing barriers to continued engagement (Krasny and Tidball, 2015).

**Tree maturity:** A young tree planted in LA needs supplemental irrigation and additional care for an establishment period of 3–5 years, with the frequency of care diminishing as the tree matures (de Guzman et al., 2018).

**Season when tree is planted:** Planting a tree in the cool, wet season means less supplemental watering is needed in the first months after planting.

**Tree growth rate:** The species growth rate and the size of the tree at the time of planting influence the length of the establishment period (Watson, 2005).

**Precipitation regime:** The seasonal distribution of precipitation in a city or region determines how much supplemental irrigation a tree may need during its establishment period.

## Discussion and conclusion

There is increasing recognition of the importance of urban greening to public health in the age of climate change, and approaches are needed that can advance our understanding of the social, ecological, economic, and political mechanisms that either facilitate or hinder urban greening (Donovan et al., 2021; Sharifi et al., 2021). As we’ve demonstrated, UTC is

influenced by socio-cultural and economic processes that shape spatial outcomes, and these are often a combination of both current and historical drivers ranging from available planting spaces and funding, to social stratification (the associations between tree cover and income, race, ethnicity or education) and neighborhood succession (when a previously dominant ethnic, racial, religious, or socioeconomic group leaves a residential area and other groups fill its place) (Danford et al., 2014). These processes give rise to concerns around gentrification and displacement, issues that neighborhood improvements such as greening projects can potentially exacerbate (Checker, 2011; Wolch et al., 2014; Dawes et al., 2018; Donovan et al., 2021; Sharifi et al., 2021). Considering the long temporal periods required for the establishment of UTC, current conditions may be inherited and serve as reflections of past preferences and processes rather than current forces (Boone et al., 2010; Schwarz et al., 2015). Whether historical or present-day, many of these forces have led to systemic segregation and have important implications for health (Jesdale et al., 2013). Biophysical factors, including climate, soil type, available planting space, and topography, among others, also impact the success of tree planting programs, and the LA region is unusually diverse across all of these categories. In arid and semi-arid climates, including Southern California's Mediterranean climate, summers are typically hot and dry and trees must receive supplemental watering during the multi-year establishment period in order to survive. While watering is not the only tree maintenance activity required in the establishment period of young trees, it is an action that must be coordinated and done frequently, and it is a determining factor in the ultimate success or failure of a planting program (Jack-Scott et al., 2013; Roman et al., 2015).

In this study, we use a case study that applies and adapts a public-health based framework to better understand the use of tree planting programs as a solution to address extreme heat and subsequent public health benefits in a large semi-arid metropolitan area (Livesley et al., 2016; Santamouris et al., 2017; Kalkstein et al., 2022). We applied a socio-ecological approach used in public health disciplines to address this issue, and we developed our own alternative model to explore spatial and temporal factors as well. We did this by assuming a baseline understanding of the importance of ecological systems in providing ecosystem services and of the role that social systems play in managing natural resources (Escobedo et al., 2019). Our use of an integrated, mixed-methods approach in the City of LA reveals social and political factors and dynamics that influence urban actors engaging in urban greening programs with direct implications for public health.

We find that the Tree Ambassador Program effectively provides residents an avenue to act on their desire to serve as change agents for their communities. During the 10-month pilot program ending in April 2022, TAs planted or distributed a total of 1,929 trees and canvassed an estimated 1,244 residents. We also find that TAs face a variety of challenges, some of which are deep-rooted and intractable, as they try to convince members of

their communities to engage in tree stewardship. For instance, of the nearly 2,000 trees added to LA's urban forest through their efforts, only 53 were street trees that TAs were able to secure with agreements by nearby property owners or tenants to provide establishment-period watering. Even so, TAs used a variety of creative, community-specific strategies to get trees planted in their communities (Supplementary Table 3). TAs feel supported by the program, but there is room to refine the program and further bolster TAs' efforts in its future iterations (Figures 1, 2, 4).

Our focus group results, survey results, and ethnographic observations reveal that TAs leveraged trees as an avenue for community cohesion and understanding, and tree-centered community events provided an opportunity for TAs to celebrate the vibrancy of their community and highlight social ties and bonds. Whereas, power dynamics at the beginning of the training program favored program staff, by the end of the program those dynamics had shifted (Supplementary Table 3). Self-efficacy and collective efficacy (people's individual or shared beliefs that they can produce desired results) were evident as TAs supported one another in designing, organizing, and successfully executing community engagement and tree planting and care activities. Through the lens of the socio-ecological framework, the results indicate that the Tree Ambassador Program was effective in advancing urban forest equity at the first three levels—individual, relationship, and institutional level—while barriers at the last two levels of community and society remain significant.

Our findings corroborate that in the LA region, trees also lack protection in the face of redevelopment trends, which favor larger homes and higher ratios of hardscape, all while UTC inequity persists between higher- and lower-income neighborhoods (Pincetl, 2010; Lee et al., 2017). Current policies, funding levels, and trends compound historical contributors to low UTC. Our SES models (Figures 8, 9) indicate that there are entrenched drivers that perpetuate these conditions, but also reveal factors that can support advancing urban forest equity at the local level.

We also find that while UTC is correlated with socio-economic variables, that correlation is highly context-specific. Schwarz et al. (2015) and Volin et al. (2020) are among several studies documenting this phenomenon. Where clear relationships emerge across factors such as minority population, income, education, rentership, imperviousness, and climate zone, elsewhere those relationships do not correlate (Landry and Chakraborty, 2009; Schwarz et al., 2015; Riley and Gardiner, 2020). Our study (Table 1) adds additional evidence of this. For instance, Tree Ambassador #1 represents a foothill neighborhood that has high UTC (30%) but also has among the highest scores of pollution burden (87th percentile)—a measure that takes into account metrics including poverty, education, and public health indicators.

This is one driver behind the inequitable distribution of UTC in LA, but understanding the context specificity of how UTC



and socio-economic variables are related is critical. For example, UTC is positively correlated with the percent of Asian residents in LA but negatively correlated in Sacramento, CA (Schwarz et al., 2015). Contradictions abound in the literature, in large part because communities are highly complex, and factors such as the instability of neighborhood demographics and various legacy effects, including redlining, further contribute to these varied associations (Dawes et al., 2018; Volin et al., 2020). In cities where overall UTC is relatively high, tree equity tends to be lower, though the strength of that relationship too is variable (Volin et al., 2020). In LA, the relationship between UTC and percent Asian is positive, but it is negative and significant for both percent Black and percent Latino/a (Schwarz et al., 2015). When looking at income and educational attainment, the picture of inequity becomes clearer in LA: neighborhoods that are lower income and where educational attainment levels are low have much lower UTC than wealthier neighborhoods (McPherson et al., 2007; Riley and Gardiner, 2020).

More than half a century after the end of redlining, the legacy patterns of disinvestment are still evident today, and they are evident in our findings (Table 1, Figures 8, 9). A spatial assessment of 108 urban areas in the US, including Los Angeles, found that in addition to being hotter, in 94% of cases formerly redlined neighborhoods presently have two to three times less tree cover than their wealthier, non-redlined counterparts (Hoffman et al., 2020). Our study indicates that raising awareness of these enduring legacies of injustice can be a motivating factor for engaging in their undoing, and that tree stewardship can serve as a tangible act of addressing the causes of injustice.

Despite concerted efforts to raise UTC, achieving equitable distribution of urban trees continues to be difficult for myriad reasons. These may include lack of program oversight resulting in haphazard progress, limited funding availability, and physical and ecological constraints in environmental justice communities that are often located in more densely built-out parts of the city with limited numbers of readily plantable sites such as unplanted planting strip spaces and other sites that do not require pavement removal or other costly site modifications (Pincetl et al., 2013; Danford et al., 2014). A study that evaluated various tree planting scenarios in Boston found that focusing planting efforts mainly in environmental justice zones resulted in a lower overall UTC increase relative to planting scenarios that prioritized neighborhoods with mixed or higher socio-economic status, due in large part to site constraints such as narrow sidewalks that cannot accommodate trees, and a lack of pervious space suitable for planting (Danford et al., 2014). In LA, we found that in addition to physical constraints, distrust in local government, the belief that street tree stewardship is the responsibility of the city, and the belief that watering a tree is expensive, are also significant barriers to tree adoption and care.

As shown in Figures 3, 6, tree care, maintenance and watering are also persistent factors at the society level

that impact a Tree Ambassador's ability to organize their communities around tree planting and stewardship. In a city that is nearly 1,295 square kilometers, such management actions pose significant logistical challenges due to urban tree planting locations often being scattered over large geographic areas rather than concentrated in smaller areas, coupled with the fact that many planting sites are not served by automatic irrigation systems (City of Los Angeles Bureau of Street Services, 2015). In particular, LA's model of shared maintenance responsibilities for street trees presents additional complexities, and delivering water from tree to tree is time-intensive and requires sufficient resources to cover costs including labor, transportation, and watering infrastructure (Jack-Scott et al., 2013; Pincetl et al., 2013). Additionally, despite increasingly widespread acknowledgment that trees are critical city infrastructure, the City of Los Angeles has struggled to allocate sufficient funding to urban forest maintenance in line with industry standard best management practices since the recession that began in 2008, spending less per capita on trees than cities of comparable size, with an estimated \$70–80 million needed to bring LA up to robust urban forest management levels (Dudek for City Plants, 2018). Due to inadequate funding, the city's public tree management approach across various departments has been limited to emergency response rather than proactive enhancement, preservation, and care, and non-profit organizations must often fill in the gap for city services that are deferred or wholly unavailable (City of Los Angeles Bureau of Street Services, 2015). Of the many barriers TAs encountered in their community organizing, the "opt-in" method of requiring residents to water street trees was consistently raised, and yet an alternative vision to transfer watering responsibility to the city seems unattainable due to funding levels that are chronically insufficient.

The complexity of factors related to tree stewardship programs lead to various approaches to operating public tree-planting programs, ranging from local government-led to non-profit-led campaigns, with public-private partnerships falling within that spectrum. Whether performed by a paid workforce or volunteer residents, urban forest management demonstrates how human agency plays a direct role in the production and distribution of the services, potential disservices and benefits of urban ecosystems, including benefits to public health. How and by whom management is performed, and how resultant costs and benefits are shared and distributed is determined largely by directives made by local government and the constellation of resources that are cobbled together to try to support them. In some cases, philanthropic funds may be present—for instance, heiress Betty Brown Casey provided a \$50 million endowment to found Casey Trees in Washington D.C., while celebrity Bette Midler committed \$200 million to former New York Mayor Bloomberg to plant one million trees (Washington Post, 2001; Danis, 2007). Los Angeles has not experienced such philanthropic fortune but the City has nevertheless embarked upon ambitious



tree-planting efforts on several occasions in recent decades. In advance of the 1984 Olympics, an effort to plant and distribute one million trees was undertaken, led by volunteers (Lipkis, 1984). More recently, the launch of Million Trees LA in 2007 signaled a renewed commitment to elevating urban greening. Despite falling short of its goal and drawing criticism regarding its methods (Pincetl, 2010; Pincetl et al., 2013), in 2014 the program underwent a transformation, rebranding itself as City Plants and aligning its approach with the tree planting ethos “right tree, right place, right reason.” In its current iteration, City Plants oversees an array of urban forest programs and funding streams that serve as a critical force in greening LA, with a focus on equitable access to trees. This equity focus drove the pilot of the Tree Ambassador Program, which has received funding to continue future rounds of hiring and training. The focus on equity also drives additional programs, including City Plants’ convening of the Los Angeles Urban Forest Equity Collective, a collaborative of government, non-profit, community, and academic entities working to actively grow, protect, and prioritize an urban forest that is accessible, inclusive, deeply valued, community-driven, adequately funded, and enduring for all Angelenos (CAPA Strategies for Los Angeles Urban Forest Equity Collective, 2021a,b).

We capture the constellation of factors impacting tree stewardship in Figures 8, 9 with the intent to provide a framework to inform future UTC management activities and urban forest equity programming in Los Angeles. The nested framework (Figure 8) can be used to understand not only the relevant drivers that facilitate or hinder tree stewardship, but also to shed light on how the city and its non-profit partners can intervene in boosting factors that support increased UTC and reduce those that hinder it. The pyramidal framework (Figure 9) offers an alternative way of conceptualizing these drivers, and adds the additional considerations of how time and space impact tree stewardship. It is our hope that these frameworks are useful to decision makers, non-profit leaders, as well as individual residents; that factors will be added or removed to tailor the models to local needs; and that they will be improved upon in LA and beyond.

Our study does have some limitations. First, our sample size was low due to the exploratory nature of this new program. Thus, long-term follow up is needed not only of TA knowledge and neighborhood governance metrics, but if indeed increased UTC in the neighborhoods has measurably improved human thermal comfort and public health metrics such as morbidity and even mortality. Second, because this initial stage of the program focused primarily on tree planting in readily available sites, such as vacant street tree wells or private lots with front or back yards, we did not explore other planting options available to neighborhoods with multi-residential housing units or the use of concrete or asphalt removal to create tree planting sites. Though this method of creating tree planting sites via removal of impervious surfaces or other site modifications represents a more expensive pathway, in cities including LA, limiting

tree planting initiatives to presently-available spaces and not expanding efforts to spaces that require removal of impervious surfaces or other site modifications can hinder substantial UTC increase in impervious surface dominated neighborhoods that stand to benefit the most from additional trees (McPherson et al., 2011; CAPA Strategies for Los Angeles Urban Forest Equity Collective, 2021a). Similarly, we did not explore in detail the attitudes and perceptions of respondents to trees and UTC as well as the economic and funding limitations of TAs, property owners, renters, and other stakeholders and how this affects tree stewardship and public health outcomes (Dawes et al., 2018).

With growing recognition of the drivers behind urban forest inequity, many of LA’s tree planting programs have shifted to prioritizing low-canopy areas while continuing to face the realities of physical, social, and funding challenges entrenched in these neighborhoods. Untangling and addressing these forces is an intractable task strongly bound to socio-economics, policy, and the political economy of resource distribution. Additionally, prioritizing locations and identifying site modifications needed for large stature trees is critical, as larger trees maximize public health benefits for the same amount of establishment care resource investment. The emphasis on the number of trees planted may be less important than the size of the trees planted, given the greater shade that larger trees are able to provide, particularly when it comes to protecting frontline communities from the public health risks of urban heat. In highly impervious, densely populated neighborhoods like Westlake, where current site conditions cannot easily accommodate trees on private property or in the public right-of-way, Tree Ambassadors would need to address significant society level barriers in order to significantly move the needle on increasing UTC and addressing urban forest equity in their communities. At the individual or relationship level, this can be a monumental task (for example, leading to decision points such as trading a parking space for a tree well). This reality indicates that policy makers and society level stakeholders have considerable control over advancing urban forest equity, and that individual or community level programs will only go so far without significant society-level intervention.

Through our application of the SES framework, and in our analysis of the results, we conclude that interaction between all spheres of influence, across space and time, from the individual level to the society level, is required to advance urban forest equity in support of public health, and a singularly top-down or bottom-up approach is inadequate. The approach and SES model developed in this study used an equity-focused lens and accounted for the nexus between public health and urban forestry and its related fields. In a similar manner to the increased acknowledgment seen in recent years of the role that contact with nature plays in promoting mental health, we suggest that urban greening programs can be better aligned with optimizing climate adaptation, heat reduction, and the provision of public health benefits. Further, we suggest that increased coordination between urban ecology

and public health disciplines can serve as a tangible expression of the transdisciplinarity necessary to navigate the intractable challenges of a climate-changed era, particularly in marginalized communities, not only in LA but in other cities across the globe as well.

## Data availability statement

The original contributions presented in the study are included in the article/[Supplementary materials](#), further inquiries can be directed to the corresponding author.

## Ethics statement

The studies involving human participants were reviewed and approved by UCLA Office of the Human Research Protection Program. Written informed consent for participation was not required for this study in accordance with the national legislation and the institutional requirements.

## Author contributions

EdG: project conceptualization, methodology, investigation, data collection, editing, supervision of original manuscript, and project administration. EdG and FE: data analysis. FE: translation. EdG and RO'L: SES framework development and funding acquisition. EdG, RO'L, and FE: writing and editing of original manuscript. All authors contributed to the article and approved the submitted version.

## Funding

This project was made possible with support from U.S. Forest Service, Region 5's Urban and Community Forestry Program, the Los Angeles Center for Urban Natural Resources Sustainability, and the UC Center for Climate, Health and Equity.

## Acknowledgments

The authors thank Alyssa Thomas and Macy Dreizler with the U.S. Forest Service Pacific Southwest Research Station for

their work analyzing the survey and geospatial data. The Tree Ambassador Program was made possible via support by the Los Angeles Department of Water and Power, the California Department of Forestry and Fire Protection, the U.S. Forest Service, and Ecosia, through City Plants, the Los Angeles Conservation Corps, and Koreatown Youth and Community Center, and was administered with participation from Climate Resolve and TreePeople. The authors also wish to thank Dr. Liz Koslov and Dr. David Eisenman at UCLA.

## Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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## Supplementary material

The Supplementary Material for this article can be found online at: <https://www.frontiersin.org/articles/10.3389/frsc.2022.944182/full#supplementary-material>

### SUPPLEMENTARY TABLE 1

Socioeconomic and demographic composition of Tree Ambassadors who anonymously responded to demographic questions in the survey.

### SUPPLEMENTARY TABLE 2

Tree Ambassador mid-program assessment focus group script.

### SUPPLEMENTARY TABLE 3

Content analysis of Tree Ambassador Program ethnographic observations.

### SUPPLEMENTARY TABLE 4

Socio-ecological model of factors that influence tree stewardship.

### DATA SHEET 1

Survey instrument (Spanish).

### DATA SHEET 2

Survey instrument (English).

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## OPEN ACCESS

## EDITED BY

Eric M. Wood,  
California State University, Los Angeles,  
United States

## REVIEWED BY

Nigel G. Taylor,  
Ecological Consultant, Cambridge,  
United Kingdom  
Andres Aguilar,  
California State University, Los Angeles,  
United States

## \*CORRESPONDENCE

Wendy Katagi  
wkatagi@stillwatersci.com

## SPECIALTY SECTION

This article was submitted to  
Urban Ecology,  
a section of the journal  
Frontiers in Ecology and Evolution

RECEIVED 29 April 2022

ACCEPTED 03 August 2022

PUBLISHED 25 August 2022

## CITATION

Katagi W, Butler N, Keith A, Backlar S  
and Orr B (2022) Ecological restoration  
of the Los Angeles River provides  
natural and human benefits as part of a  
virtuous socioecological cycle.  
*Front. Ecol. Evol.* 10:932550.  
doi: 10.3389/fevo.2022.932550

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# Ecological restoration of the Los Angeles River provides natural and human benefits as part of a virtuous socioecological cycle

Wendy Katagi<sup>1\*</sup>, Nate Butler<sup>1</sup>, Anthony Keith<sup>2</sup>, Shelly Backlar<sup>1</sup>  
and Bruce Orr<sup>3</sup>

<sup>1</sup>Stillwater Sciences, Los Angeles, CA, United States, <sup>2</sup>Stillwater Sciences, Boulder, CO, United States,

<sup>3</sup>Stillwater Sciences, Berkeley, CA, United States

Ecological restoration in the Los Angeles (LA) River watershed is proceeding on multiple fronts with the support and engagement of diverse stakeholder groups. Pilot projects to restore habitat, reintroduce native species, and design science-based ecosystem enhancements have produced real benefits to nature and people and demonstrated the potential for additional benefits. The pilot projects, which are in various stages of collaborative planning and implementation, have generated increased interest and financial support to further their implementation and maximize socioecological co-benefits. This self-reinforcing positive feedback is an example of a virtuous cycle established through a combination of long-term environmental planning, community-building, and watershed-scale scientific study to gain the support of stakeholders and align ecological intervention (i.e., restoration) with the plans and policies of governments, resource managers, conservation groups, and grassroots advocacy groups. Conservation and restoration projects targeting iconic and protected focal species can be an effective means of leveraging these interests and building support. For example, the LA River Fish Passage and Habitat Structures project addresses a critical limiting factor for the recovery of endangered steelhead trout (*Oncorhynchus mykiss*) while also enhancing urban biodiversity and providing recreational opportunities and other beneficial uses (e.g., ecosystem services) for the surrounding communities. Through these efforts, our planners, ecologists, and engineers are using place-based conservation to demonstrate solutions to problems that affect people and nature in other urban landscapes. Here, we show how this work can provide socioecological benefits in disadvantaged communities and also generate public awareness and motivation to perpetuate the cycle of positive feedback.

## KEYWORDS

virtuous cycle, biodiversity, ecosystem services, conservation, urban river, restoration, steelhead



## Introduction

As global populations have moved from predominantly rural to majority urban living, urban rivers are increasingly seen as opportunities to improve ecosystem services and biodiversity (Costa et al., 2010; Schneiders et al., 2011; Everard and Moggridge, 2012; Francis, 2012). Urban rivers potentially provide numerous ecosystem services that benefit humans and their wellbeing (e.g., flood protection, cultural heritage, recreation) (Millennium Ecosystem Assessment Program [MEA], 2005a,b,c), but development has degraded rivers to the extent they often no longer can provide these services (Carpenter et al., 2009; Everard and Moggridge, 2012). Preservation and enhancement of biodiversity is also increasingly a focus of urban ecology since biodiversity increases the resiliency of ecosystems to perturbations like climate change that would reduce their ability to provide ecosystem services (Carpenter et al., 2009; Schneiders et al., 2011).

The virtuous cycle framework for conservation seeks to simultaneously address ecosystem service needs and the broader benefits of biodiversity by highlighting the relationships between places, people, and biodiversity that are essential to developing conservation projects (Morrison, 2015). Morrison (2016) describes a virtuous cycle framework in which conservation projects can be designed to promote engagement and further action by stakeholders as the benefits of conservation accrue, producing a durable, self-perpetuating cycle of improvements in both ecosystem services and biodiversity. Virtuous cycles can operate at a range of spatial scales (e.g., river reach or watershed), with virtuous cycles at different scales often supporting one another (Morrison, 2016). In this article, we present an example of a virtuous socioecological cycle fostered in the Los Angeles (LA) River watershed by numerous science-focused conservation projects that have generated burgeoning momentum and support by aligning with stakeholder priorities and addressing societal and ecological needs. We describe steps and key elements of the LA River watershed-scale virtuous cycle, with examples illustrated by several pilot projects at the reach scale. Because our example projects are still in planning and early implementation phases, their conservation benefits are envisioned but not yet fully realized.

## A river reborn—Starting a movement

A primary challenge of the virtuous cycle at any scale is developing the initial critical mass of engaged stakeholders and conservation projects such that sufficient positive outcomes are created to motivate future actions. The creation of a movement where individuals, agencies, and organizations are supportive, engaged, and inspired to take additional actions is vital to

creating a self-perpetuating virtuous cycle that can achieve positive conservation outcomes for people and nature.

Creating a virtuous cycle for people living in LA, who do not realize that the concrete drainage system running through one of the most densely populated areas of the United States was once a natural river, took decades. Among the challenges was, and still is, convincing residents that the drainage was once a free-flowing river with native fish and that it is possible to bring nature back into the built environment. To most, the LA River, which was once the sole source of water for the City of LA and habitat for the iconic Southern California steelhead trout (*Oncorhynchus mykiss*, herein steelhead), is a no man's land, fenced, and forgotten. A "River Movement" advocating the socioecological value of the LA River and the potential for its restoration began in 1986 with the founding of Friends of the LA River (FoLAR) (Friends of the La River [FoLAR], 2021). In 1996, LA County adopted the LA River Master Plan creating a vision for bike paths, parks, and other amenities along with the 51-mile LA River (Los Angeles County, 1996). The Master Plan illustrated the possibilities for improving access to nature and enhancing biodiversity in the urban riverscape and vastly increased visibility and understanding of how these improvements would benefit human wellbeing.

In the decades since the Master Plan, conservation efforts to restore habitat, reintroduce native species, and design science-based ecosystem enhancements have produced multiple socioecological benefits. Pilot projects in the watershed have generated increased interest and financial support to further their implementation and align their objectives with ecological and social priorities. This self-reinforcing positive feedback is an example of a virtuous cycle established through a combination of long-term environmental planning, community-building, and watershed-scale scientific study to gain the support of stakeholders and align habitat and flow restoration projects with the objectives of government, resource managers, and conservation/advocacy groups.

## Strategies to engage stakeholders in the virtuous cycle

### Target species restoration opens doors for community engagement

A target/focal species approach to ecosystem restoration has proven to be an effective framework on which a virtuous cycle of ecosystem restoration is being built in the LA Basin. Using a focal species approach that is founded in multidisciplinary science and explicitly strives to achieve multiple socioecological benefits at a watershed scale, steelhead serves as an umbrella species whose restoration and return to the LA Basin will require multiple beneficial improvements to the ecosystem with

intended positive outcomes for native species and habitats, as well as people. Within this framework, targeted focal species restoration efforts can be designed as multi-benefit projects that engage numerous stakeholders in the virtuous cycle while using the recognition of the focal species to generate enthusiastic support among a wide range of stakeholders (Novacek, 2008; Qian et al., 2020). This multi-benefit focal species approach is fully compatible with the current paradigm in ecological restoration that recognizes linkages among social and ecological systems and emphasizes the need for multi-benefit goals as foundational to the success of conservation, restoration, and management efforts (Wallace et al., 1996; Apitz et al., 2006; Gardali et al., 2021). In fact, it may be a particularly effective catalyst for a virtuous cycle in an urban setting because it provides a focused, recognizable foundation for ecological and social benefits that are manifested in, of, and for the city (Pickett et al., 2016). Because familiarity and positive associations with species have been found to increase individuals' willingness to pay for conservation more than ecological-scientific factors (Martín-López et al., 2007), iconic focal species that are well-known to the public and stakeholders, and indicators or keystone species are considered highly suitable targets for urban ecosystem restoration. Where they are present or potentially present, species listed as threatened or endangered are an effective choice as focal restoration species as they are especially well-known and important to many stakeholders (Qian et al., 2020). Such species are the subject of legal protections, inherently generating support from regulatory agencies and conservation organizations, opening doors to funding, and building a virtuous cycle.

The Southern California steelhead is listed under the federal and state Endangered Species Acts and is the subject of a federal Restoration Plan (National Marine Fisheries Service [NMFS], 2012), making it an ideal focus for regional river restoration and conservation efforts. This species is the current focus of restoration efforts in the LA River, including the Los Angeles River Fish Passage and Habitat Structures (LAR FPHS) project being implemented under the watershed-wide LA River Fish Passage Program (LAR FPP), and related efforts in the lower mainstem LA River and the Arroyo Seco, a headwater tributary. The focus on steelhead has helped generate support and involvement by the public and other groups including the National Marine Fisheries Service, California Department of Fish and Wildlife (CDFW), Arroyo Seco Foundation (ASF), the LA Mayor's Office, CalTrout, Trout Unlimited, and others.

## Aligning federal, state, and local policies, programs, and plans

Alignment of restoration with environmental regulations, adopted plans, strategic initiatives, programs, and projects at all levels of government, is critical to fostering the conditions for a virtuous cycle. When a conservation project is undertaken,

regulatory and resource agencies [e.g., U.S. Army Corps of Engineers (USACE), CDFW] are one of the categories of stakeholders that must be engaged to promote a virtuous cycle since their approval or participation (e.g., permits) is needed to advance the project. Alignment helps the agencies achieve their goals and motivates them to support and advance the project. Direct involvement in and approval for a conservation project by these agencies also communicates to other stakeholders that the project is beneficial to a resource, building the momentum and broader support for conservation projects that promotes the virtuous cycle (Morrison, 2016).

The LAR FPHS project engages the support of regulatory and resource agencies by incorporating alignment with policies, programs, and plans such as the City of LA Biodiversity Report (City of LA, 2020) and the LA River Ecosystem Restoration Integrated Feasibility Report (USACE, 2015) along with highlighting how it advances the objectives of the LA River Ecosystem Restoration and Recreation Project (LARERRP), the City of LA, and the USACE (Stillwater Sciences, 2022). Alignment can also lead to agency actions that advance the project. The LARERRP was designated as a P3/Alternative Delivery pilot project by the USACE due to its alignment with USACE objectives (City of LA, 2021). Wider alignment also promotes the virtuous cycle by providing tangible examples of how agencies can work together to further their different goals, facilitating future collaborative efforts, and reducing the likelihood of conflicts that would impede future projects.

## Enhancing ecosystem services provides societal benefits

Conservation objectives aimed at preventing extinction and advancing the recovery of endangered species appropriately and justifiably focus on benefits to nature (e.g., habitat improvement, population expansion) without an explicit focus on societal benefits, but conservation projects in a virtuous cycle, especially those in urban environments, must consider how outcomes will affect adjacent communities (Morrison, 2015, 2016). Conservation projects are better able to motivate communities, Tribal Nations, and other local stakeholders to participate in the virtuous cycle by understanding the ecosystem services the project site provides to those living near it and incorporating enhancement of those ecosystem services into a conservation project. Ecosystem services are used to help muster support for conservation by assigning a quantitative (e.g., monetary) or qualitative (e.g., cultural identity) value to conservation outcomes and justifying conservation objectives relative to society (Costanza et al., 1997; Bullock et al., 2011; Seppelt et al., 2011). While the monetary valuation of ecosystem services can provide useful information to guide conservation, it is also important to incorporate ecosystem services that are valued by local communities but cannot easily be assigned a monetary value into conservation projects to generate the

enthusiasm to engage these communities in the virtuous cycle and foster community support of elected officials who support conservation.

In the urban environment along with the LA River where access to nature is sparse, conservation projects often include enhanced access to nature and recreation since those are key ecosystem services local communities frequently list as important to them. Channel redesigns for the LAR FPHS project specifically took into consideration how the conversion of barren concrete to vegetation along with the channel enhances access to nature along with the river and improves the kayaking experience (Stillwater Sciences, 2022). The LARERRP also enhances access to nature and recreation for local communities along with the river by including new parks, vegetating barren concrete, and constructing wetlands along with the LA River (City of Los Angeles [City of LA], 2016).

## Regional conservation projects promoting the virtuous cycle

### Steelhead recovery projects across the Los Angeles River watershed

Efforts to facilitate steelhead recovery along with the length of the LA River have built support for restoring the river and its tributaries by highlighting how conservation projects for steelhead can provide multiple benefits. The LAR FPP consists of a series of fish passage and habitat structure design pilot projects to restore fish migration from the ocean to spawning habitat in the upper tributaries. As the first of several projects under this program, the LAR FPHS project (Stillwater Sciences, 2021, 2022) and the Conceptual Ecological Model and Limiting Factors Analysis for Steelhead in the LA River watershed (Limiting Factors Analysis, or LFA) (Stillwater Sciences, 2020) have been especially influential in catalyzing a movement by bringing together stakeholders to form a collective vision for steelhead recovery in the LA River watershed.

The LAR FPHS project demonstrates how a target species conservation approach for steelhead can also be developed as a multi-benefit project that simultaneously advances the goal of restoring steelhead to the LA River while being consistent with the goals of local communities, conservation organizations, and numerous agencies (Figure 1). The LAR FPHS project advances the local, reach-scale, and watershed-scale virtuous cycles of LA River conservation by ensuring that its own goals align with the goals of related agency-developed plans. The broad alignment of the LAR FPHS project with these goals and the support from elected officials, such as Mayor Eric Garcetti, incentivizes these agencies to take action to advance the LAR FPHS project and its conservation outcomes. As a pilot project, it also highlights how a multi-benefit conservation

project that advances multiple conservation outcomes can be scaled up and used as a template for other projects within the watershed. One reach of the LAR FPHS is moving into final design and construction, while the overarching LAR FPP creates more momentum for related watershed-wide projects engaging regulatory agencies and community.

The LFA provides the foundational science-based framework for the steelhead recovery efforts in the watershed, including the LAR FPHS, and recommends studies, conservation projects, and planning efforts that should be implemented to advance steelhead recovery within the watershed (Stillwater Sciences, 2020). Its recommendations to build multi-benefit conservation projects by coordinating steelhead-focused planning and conservation projects with watershed-wide initiatives are key to contextualizing how steelhead recovery efforts provide value to a wide range of stakeholders—engaging more groups in a movement and developing funding partnerships. Planning and implementing the steelhead actions in coordination with other plans, projects, and initiatives are key to developing approaches to river-riparian restoration and enhancement that capitalize on synergies and multi-benefit strategies throughout the watershed and the region and perpetuate the virtuous cycle.

### Conservation projects within the Arroyo Seco

Vital to steelhead recovery efforts in the LA River watershed is the Arroyo Seco, whose headwaters have cool stream habitat suitable for trout and steelhead. Adopted stakeholder-based watershed plans, including the Arroyo Seco Watershed Assessment, have led to conservation projects like the Central Arroyo Seco Stream Restoration that restore more natural stream conditions for native fish and enhance ecosystem services (e.g., recreation) for the surrounding communities (Arroyo Seco Foundation [ASF], 2008a; ASF, 2011; CDM, 2011). Native arroyo chub (*Gila orcuttii*), a key indicator species for river and riparian health, was reintroduced to this 20-acre restoration area in 2008, and early data indicated they were persisting in the stream and enhancing the local biodiversity (U.S. Army Corps of Engineers [USACE], 2011). CDFW cites it in management recommendations for the species as an example of successful native fish restoration (Moyle et al., 2015).

These projects and activities have advanced the virtuous cycle by generating engagement and enthusiasm for conservation along with the Arroyo Seco and the LA River watershed, as evidenced by ASF's volunteer Trout Scouts, its growing newsletter circulation, and public comments advocating for conservation projects in the watershed (USACE, 2015; Sierra Institute, 2019). ASF and the City of Pasadena also have fostered involvement in the virtuous cycle by annually welcoming hundreds of watershed stewards to assist with native

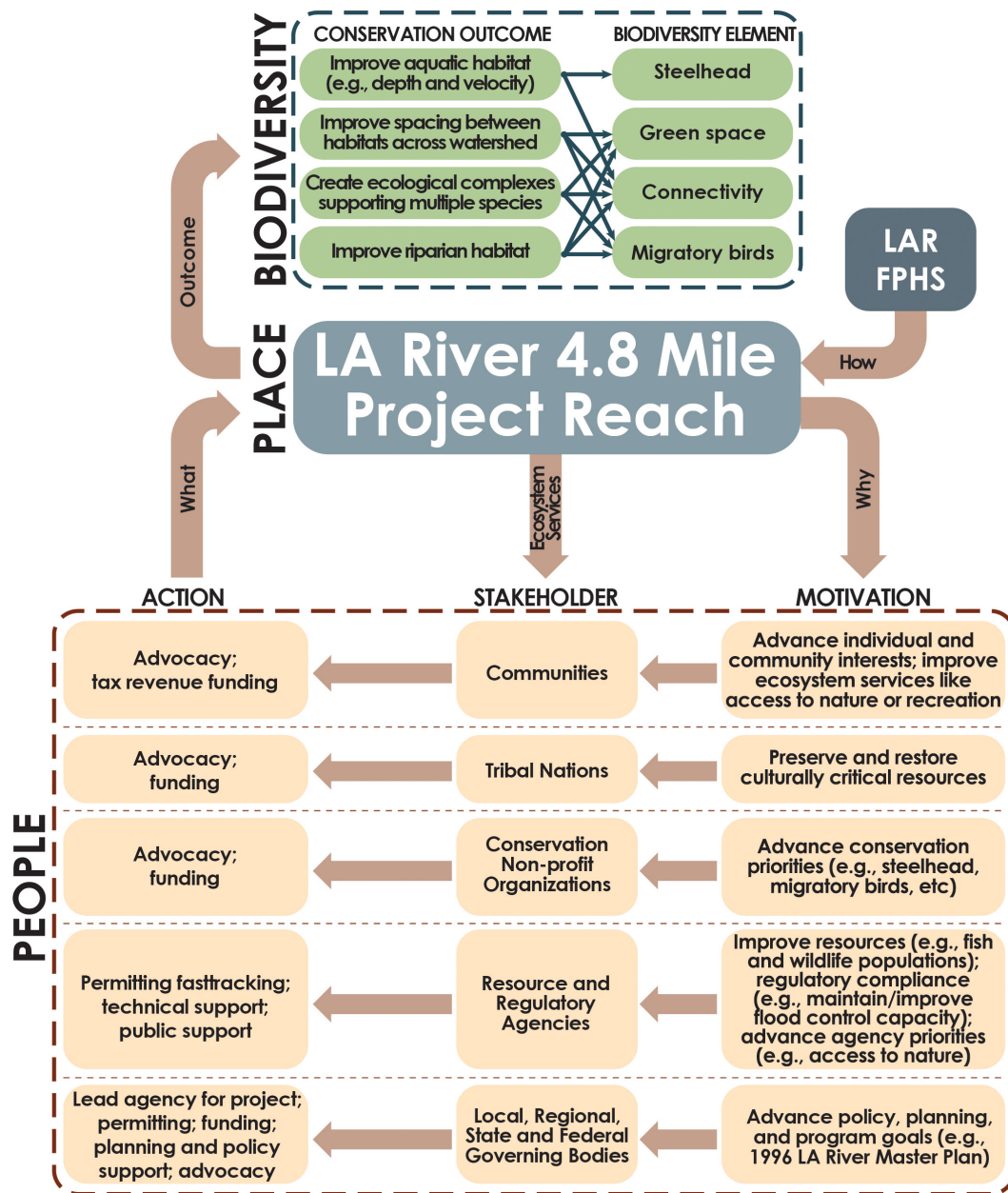


FIGURE 1

The LA River Fish Passage and Habitat Structures (LAR FPHS) project advances a virtuous socioecological cycle in the 4.8-mile project reach of the LA River between the Glendale Narrows and Washington Boulevard. The LAR FPHS is designed to provide motivations for a wide range of stakeholders to take actions to support conservation in the LA River that advances multiple social and ecological priorities and ecosystem services of stakeholders and promotes urban biodiversity. In this diagram, the arrows flow initially from the LAR FPHS project to the "Place" box because it is "how" the virtuous cycle is started. Next, the arrows flow from the "Place" box to the "Motivations" column in the "People" box to highlight how the LAR FPHS project is designed to provide specific motivations for "why" individual stakeholders want to engage in the virtuous cycle, so stakeholder support is not necessarily dependent on the stakeholders valuing the planned conservation outcomes or biodiversity enhancements of the project. Arrows continue to flow from "Stakeholders" column of the "People" box to the "Action" column to emphasize "what" actions engaged stakeholders can take to support the LAR FPHS in this reach of the LA River. Ecosystem services flowing from the "Place" box to the "Stakeholder" column of the "People" box shows how ecosystem services inherently are provided to stakeholders by the river reach, and enhancements in ecosystem services from the conservation project will provide direct positive benefits to stakeholders that contribute to perpetuating the virtuous cycle. Additionally, there are conservation outcomes flowing from the "Place" box to the "Biodiversity" box that show the conservation outcomes the LAR FPHS is designed to achieve (left column of the top "Biodiversity" box) and how these outcomes enhance various biodiversity elements (right column of the top "Biodiversity" box) within the project reach. Please note that the virtuous cycle shown is a simplified summary that only highlights some of the key components (e.g., stakeholders, conservation outcomes) in this conservation project and the reach of the LA River. An organically self-perpetuating virtuous cycle would expand to engage more stakeholders and produce more conservation outcomes across the LA River watershed.



plantings and other conservation activities. ASF Trout Scouts conduct educational hikes to foster watershed stewardship and recognition that suitable steelhead habitat exists in the upper Arroyo Seco. Stakeholders frequently cite the successes of the Central Arroyo Seco Stream Restoration as one motivation for continuing to push for linked conservation projects and funding opportunities to recover endangered steelhead in the Arroyo Seco and LA River mainstem (Arroyo Seco Foundation [ASF], 2008b; U.S. Army Corps of Engineers [USACE], 2011; USACE, 2014; State of California Wildlife Conservation Board [WCB], 2019; Brick, Arroyo Seco Foundation, personal communication, June 15, 2022; WCB, 2022).

## Urban orchard restoration in the lower Los Angeles River

Currently under construction, the Urban Orchard Community Park in South Gate is located along with the lower LA River, connected to the river bikeway and many other parks and recreational opportunities described in the Lower LA River Revitalization Master Plan. A grassroots, community-driven park and orchard developed by neighborhoods, community leaders, schools, and elected officials in partnership with the Trust for Public Land, Urban Orchard is a 7-acre native habitat park with a 1-acre native wetland and trout stream that will provide an oasis for residents to engage in nature-based play, farming fruits and vegetables, celebratory gatherings, and interpretive learning (outdoor classroom) about local flora and fauna (City of South Gate, 2019). The project design, its inclusion of ecosystem services for local residents, and its success at engaging the community, has already become a model for future parks and open space projects throughout the watershed, thus promoting the virtuous cycle.

## The virtuous cycle creates its own funding support

The restoration actions described above—from planning to implementation phases—bolster confidence in and support for still greater funding opportunities as the restoration activities align with the programs and strategic goals of funding entities. Each stage of a restoration project contributes to the overall progress of watershed-wide recommendations. For example, the State of California has adopted strategic planning recommendations and goals that guide grant funding opportunities for restoration projects. Likewise, state, federal, and regional entities have adopted similar recommendations that tier off or build upon such restoration strategies—such as conserving 30% of California's land and coastal waters by 2030 (California Natural Resources Agency, 2022). When watershed leaders, such as the City of LA and our LA River and Arroyo Seco restoration teams, intentionally

seek funding together for restoration projects under these guidelines, there is greater alignment with stakeholder goals, and systemic momentum is built into the watershed-based restoration process. As the LAR FPHS project has moved through conceptual design to final design, implementation of the first construction phase of the program has attracted even stronger support for implementation of the funding. Other benefits include growing funding awards for the upper tributaries, such as the Arroyo Seco restoration and fish barrier removals. Similarly, funding opportunities in the lower LA River watershed for restoration and fish passage projects have advanced project concepts in tandem. Funders leverage unanimous local support and progress in watershed-based projects, providing funding incentives for construction phases and linked restoration projects.

## Discussion

Virtuous cycles promoted by conservation projects influence broader regional conservation outcomes and improvements in biodiversity. As conservation projects such as the LAR FPHS, Urban Orchard, and Central Arroyo Seco Stream Restoration produce improvements in ecosystem services/beneficial uses, local community members are more likely to become engaged stakeholders who take action to promote the local virtuous cycles and/or the broader LA River watershed-scale virtuous cycle. As an example, tangible increases in green space and recreation opportunities within a 10-min walk or a 10-min drive provided by conservation projects in the LA River watershed (Figure 2) are anticipated to motivate local residents to become stakeholders and participate in advocacy, action, and/or funding that lead to a watershed-wide self-perpetuating virtuous cycle (Nguyen et al., 2018).

Tangible improvements in one part of the watershed also bring potential stakeholders in other parts of the watershed into the cycle as people and organizations see and experience what is possible along with an urban river. Numerous public comments on the LARERRP, including Arroyo Seco stakeholders, highlight how stakeholder engagement in the Arroyo Seco is extending into other portions of the watershed (USACE, 2015). Successes realized in the LA River watershed may also provide an example for virtuous cycles elsewhere. The following sequential steps have proven successful in generating a self-perpetuating socioecological cycle: (1) develop a science-based understanding of the system (the biodiversity), (2) strive to understand the role and importance of the system to society and stakeholders (the place and its people), (3) identify data gaps, limiting factors, or critical needs for the ecosystem and people, (4) plan and implement conservation projects that align with stakeholder priorities and societal needs, and (5) leverage



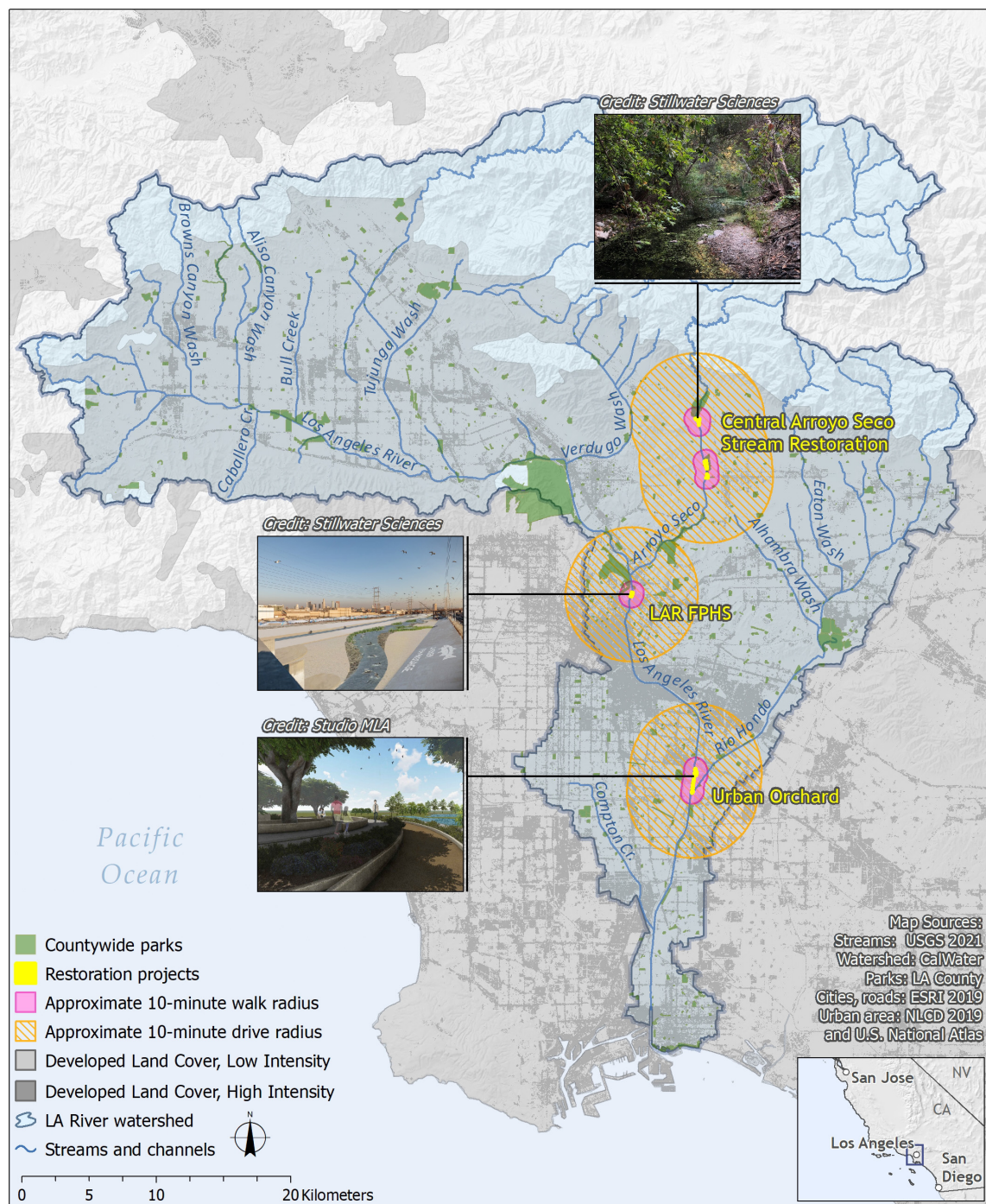


FIGURE 2

Conservation projects often improve ecosystem services for people outside of the direct project footprint, leading to further engagement by existing stakeholders and potentially encouraging new stakeholders to realize the value of joining the virtuous cycle to promote future conservation projects. As an example, the LA River Fish Passage and Habitat Structures (LAR FPHS), Central Arroyo Seco Stream Restoration, and Urban Orchard conservation projects in the LA River watershed enhance green space and improve access to nature and recreational opportunities within a 10-min walk (i.e., 0.8 km) or 10-min drive (i.e., 4.8 km) of those projects in a highly urbanized environment. Such enhancements promote stakeholder engagement in the virtuous cycle by those who want to advance these ecosystem services (e.g., local communities, City of LA, Mountains Recreation and Conservation Authority). Please refer [Supplementary Figure 1](#) for a visual comparison of pre-project and current or planned post-project improvements for the three example conservation projects. Magnitude of urbanization represented by the intensity of developed land cover using the National Land Cover Database classification of impervious surface percentage. Not depicted here are ecological and recreational benefits associated with enhanced connectivity, both aquatic and terrestrial, within and along with the river channel.

successful outcomes and alignments in efforts to plan and build new projects.

The City of LA is moving forward to restore habitat along with an 11-mile stretch of the LA River from Griffith Park to downtown LA. The LAR FPP is an example of a bold, transformative vision. Ultimately, the LA River can be transformed into an urban green space as iconic as Griffith Park in LA, Golden Gate Park in San Francisco, or Central Park in New York. Community engagement, political will, and funding fueled the creation of and desire for projects that will enhance biodiversity and improve the quality of life for people including disadvantaged communities and Tribal Nations. The virtuous cycle initiated by LA River conservation projects continues to raise awareness that healthy urban ecosystems are a cornerstone of the livability and socioecological wellbeing of LA and are demonstrating that science-based solutions benefiting wildlife and people are not only possible but they are also within our grasp.

## Data availability statement

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

## Author contributions

WK, AK, and BO conceived the manuscript. AK, NB, and WK organized the text and manuscript. WK, NB, AK, and SB contributed to the manuscript writing. NB, AK, and BO conceived the figures. BO provided an editorial review. All authors contributed to the article and approved the submitted version.

## Funding

Funding to Stillwater Sciences for the work described in this manuscript was provided by the California Wildlife Conservation Board, the Santa Monica Mountains Conservancy, the Mountains Recreation and Conservation

Authority, the State Water Resources Control Board, the City of Los Angeles, and the Rivers and Mountains Conservancy.

## Acknowledgments

Stillwater Sciences supported the authors' work on this manuscript. We also thank our partners in conservation throughout the LA Basin and our colleagues at Stillwater Sciences for inspiration and support. Special thanks to Yuliang Jiang and Anna Ballasiotes for creating the figures. Partners in the work described in this manuscript include the City of Los Angeles, Arroyo Seco Foundation, Council for Watershed Health, Friends of the Los Angeles River, Trout Unlimited, California Department of Fish and Wildlife, National Marine Fisheries Service, Bureau of Reclamation, County of Los Angeles, US Army Corps of Engineers, Regional Water Quality Control Board—Los Angeles, and US Fish and Wildlife Service.

## Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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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.2022.932550/full#supplementary-material>

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## OPEN ACCESS

## EDITED BY

Sophie S. Parker,  
The Nature Conservancy, United States

## REVIEWED BY

Wei Chen,  
Northwest A&F University, China  
Seth Riley,  
National Park Service, United States

## \*CORRESPONDENCE

Daniel S. Cooper  
dcooper@rcdscmm.org

## SPECIALTY SECTION

This article was submitted to  
Urban Greening,  
a section of the journal  
Frontiers in Sustainable Cities

RECEIVED 19 April 2022

ACCEPTED 02 August 2022

PUBLISHED 30 August 2022

## CITATION

Cooper DS, Katz ND, Demirci B and  
Osborn FM (2022) Lessons from the  
Santa Monica Mountains: Continuing  
the cycle of conservation.  
*Front. Sustain. Cities* 4:923946.  
doi: 10.3389/frsc.2022.923946

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# Lessons from the Santa Monica Mountains: Continuing the cycle of conservation

Daniel S. Cooper<sup>1,2\*</sup>, Nurit D. Katz<sup>3</sup>, Brianna Demirci<sup>1</sup> and  
Fiona M. Osborn<sup>4</sup>

<sup>1</sup>Resource Conservation District-Santa Monica Mountains, Topanga, CA, United States, <sup>2</sup>Department of Ornithology, Natural History Museum of Los Angeles County, Los Angeles, CA, United States,

<sup>3</sup>Department of Sustainability, University of California, Los Angeles, Los Angeles, CA, United States,

<sup>4</sup>Department of Geography, University of California, Los Angeles, Los Angeles, CA, United States

Spanning more than 73 km across two counties at the western border of the Los Angeles metropolitan area, the Santa Monica Mountains represent both a major landform as well as a unique urban-adjacent open space for millions of residents throughout southern California. Critically, they are essential for the maintenance of high levels of biodiversity within a global biodiversity hotspot that includes a major metropolis. The Los Angeles County portion of the Santa Monica Mountains (LASMM), spanning approximately 62 km from the Los Angeles River at the eastern edge of Griffith Park to the Los Angeles – Ventura County Line, contains substantial public open space, protected from encroaching development in the growing metropolis. In order to understand how these protected areas were established, we gathered information regarding over 3,000 parcels of public open space and their acquisition dates and owners, and examined the history of land conservation in the LASMM to determine the roles and relationships of key stakeholders. These stakeholders have included residents, activists, scientists, legislators, non-governmental organizations (NGOs), and land management agencies. We suggest that there is a virtuous cycle, or positive feedback loop over time, as open space protection is informed by, and influences, advocacy, land use policies, and habitat conservation. This interplay of stakeholders has been refined over several decades, and may offer lessons for other regions working to produce similar results in durable open space conservation.

## KEYWORDS

land acquisition, virtuous cycle framework, urban biodiversity, open space, conservation

## Introduction

The acquisition and conservation of undeveloped land, is critical to the maintenance of biodiversity. Particularly in urbanizing areas, parkland managed by public agencies represent a means of long-term conservation of resources. In a review of land conservation in the southern California city of Thousand Oaks, Towne (1998) described

nine “keys to successful open space conservation,” including “community initiative and support,” “open space conservation policies,” and “diverse implementation techniques,” including the creation of agencies dedicated to land acquisition and management (see also [Petrillo, 2008](#)). The policies of individual cities that contain undeveloped land within their borders are reflected in higher levels of biodiversity, while those that fail to acknowledge biodiversity and natural areas show the opposite pattern ([Cooper et al., 2021](#)).

The Santa Monica Mountains of Los Angeles County (LASMM) represent an ideal case study by which to understand the process of acquiring and protecting open space for public good and resource conservation, which has been ongoing in the area for more than a century ([Li and Pei, 2019](#)). While the Santa Monica Mountains span Los Angeles and Ventura counties, we focus on the Los Angeles County portion of the range, a 63km long expanse extending from the Arroyo Sequit watershed west of Malibu, east through Topanga Canyon and Sepulveda Pass to Griffith Park ([Figure 1](#)). While such popular natural areas as Griffith Park, Topanga State Park, and Malibu Lagoon might seem to most residents and visitors to have “always been here” for their enjoyment, the creation of most local parks and protected parcels of land is usually the result of their acquisition by a public agency (or shifting to another public entity, such as city land absorbed by a state park) or a donation by a private individual to a state conservancy or non-profit group. In recent decades, some efforts to preserve remaining open space threatened by development are successful only after a protracted battle involving grassroots activism organized by local residents.

We examine the complex web of interests involved in land conservation in the LASMM, and explore how these stakeholders continue to work together to support conservation of these resources. We show how early open space acquisitions, while slow to accumulate, gathered momentum after the 1960s, leading to a virtuous cycle today, where land is seen more often as a public good to be protected and fought for, rather than as a blank slate for urban development.

## Methods

### Setting

The California Floristic Province is an internationally recognized biodiversity hotspot ([Myers et al., 2000](#)). As a large area of undeveloped land in Southwestern California, the LASMM supports an ecosystem rich in California-endemic flora and fauna along its entire length ([Tiszler and Rundel, 2007](#); [Cooper, 2017](#)), and is characterized by a mix of scrub and oak-covered canyons and ridges in the larger patches of undeveloped land, with slivers of vegetation between houses in more densely-populated areas. The Santa Monica Mountains are divided at their western end by the Los Angeles/Ventura

County line. The Los Angeles County portion of the Santa Monica Mountains (LASMM), spanning ~62 km from the Los Angeles River at the eastern edge of Griffith Park to the Los Angeles – Ventura County Line. 32,000 acres of protected open space in the LASMM are contained within the boundaries of incorporated cities of varying size and population density. The majority of these open space holdings (c. 25,000 acres) are within the City of Los Angeles.

Land conservation in the LASMM was initiated by philanthropy as early as the late 1800s, with a donation (to the city) of the 4,000 acre Griffith Park in what had been the northern edge of the city of Los Angeles ([Eberts, 1996](#)). Yet, the years 1900–1950 saw just six other park acquisitions here (totaling just under 400 acres), and open space was frequently used for generally unsuccessful afforestation attempts such as planting groves of eucalyptus ([Godfrey, 2013](#)) rather than for outright conservation. Starting in the 1950s, however, acquisitions (by the state of California) of sprawling cattle ranches would become Leo Carrillo State Park (2,264 acres) and Topanga State Park (now 11,439 acres), and launched an era of widespread and significant open space protection in the LASMM that continues today. Currently, the Los Angeles portion of the LASMM alone supports more than 70,000 acres of land classified as “open space,” a category of protected area ([CPAD, 2020](#)), owned by 36 entities (refer to lists in [Supplementary Tables S1, S2](#)).

### Data analysis

To understand temporal patterns of land acquisition, we first established a study area boundary, drawing a broad perimeter around the topographical unit of the Santa Monica Mountains/Simi Hills, and then used the California Protected Areas Database ([CPAD, 2020](#)) to identify the open space parcels within the Los Angeles County portion of this area (the Ventura County portion of the range to the west exists under a different land-planning regime, as most land use decisions involving undeveloped land, such as zoning, are made at the county level). We chose to focus on Los Angeles County in order to better examine the relationships between stakeholders, not all of which operate across county lines. This resulted in a preliminary list of 3,091 individual parcels within the study area totaling 72,581 acres. To focus and simplify our analysis, we removed parcels under 1 acre in size, and removed those located in the west San Fernando Valley portion of the eastern Simi Hills (2,809 acres), which are generally treated as separate from the Santa Monica Mountains (e.g., Chatsworth Hills, Santa Susana Pass), thus restricting our analysis to the main body of the Santa Monica Mountains as commonly understood. We then aggregated multiple separate parcels by park/preserve name, resulting in 249 “sites,” and further refined our list of sites by removing school properties, urban parks (i.e., with lawn and little/no



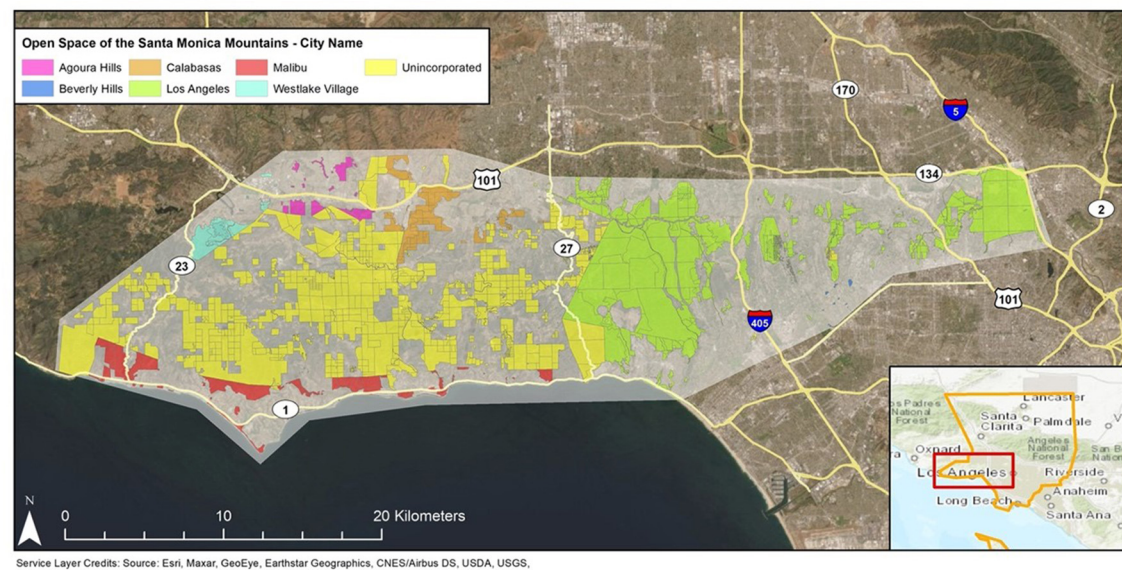


FIGURE 1

Map of protected open space of Santa Monica Mountains within Los Angeles County, by city (shading added to improve visibility). Griffith Park is the large green area at the far east, and Topanga State Park dominates the green area in the center (both within the city of Los Angeles). The surrounding cities of West Hollywood, Burbank, Glendale, Santa Monica, and Hidden Hills contain small portions of the Santa Monica Mountains topographically, but these areas are largely urbanized and do not contain protected open space, so are not reflected on the map.

natural habitat), and golf courses, restricting our analysis to areas dominated by natural open space. We then researched the most recent owner and “creation date” of each remaining site (where this information was not included in CPAD; usually the date of acquisition by the last conservation entity/agency to manage it) using online searches, which included reviews of city documents, meeting minutes and newspaper articles (keywords included “purchased,” “bought,” “acquired,” “saved,” etc.). This resulted in a final list of 91 sites located in ten incorporated cities (as well as in unincorporated Los Angeles County) where both the year of creation and the landowner is known.

## Results

Our review of protected open space in the LASMM revealed several distinct patterns involving elected officials, agencies, scientists/conservationists, and the public. In the political realm, local elections within the neighborhoods of the LASMM have consistently promoted candidates with a strong record of land protection, starting in the 1960s with Los Angeles city councilmember Marvin Braude and later Paul Koretz, County Supervisors Edmund D. Edelman and later Zev Yaroslavsky, and U.S. House members Thomas Rees and later Anthony Beilenson (Table 1). Over years and in some cases decades, these officials were essential to directing local, regional and national attention (and funds) toward land conservation in the LASMM. Their efforts resulted in multiple park bonds passed through the 2000s,

even as the acreage of undeveloped land available for purchase began to decline, making each acquisition more expensive (McGreevy, 1999; Pincetl, 2003). While land acquisition for open space exceeded 5,000 acres per decade between 1960 and 2010, that of the most recent decade (2010–2020) dropped by roughly half that of the prior one, perhaps signaling an eventual limit to how much land can be realistically acquired for public open space (Figure 2).

As these representatives worked within government to secure bond funding for park creation and management, they did so with the strong support of the earliest non-governmental organizations, including volunteers from a local task force of the Sierra Club launched in the early 1970s (Guldimann, 2018). These groups organized such events as a 5,000 person march in 1971 along the crest of the LASMM to push for the creation of a national park here, which was realized less than a decade later with the creation of Santa Monica Mountains National Recreation Area in 1978 (Woo, 2008). Today, agencies serving as de facto land trusts including Santa Monica Mountains Conservancy and the Mountains Recreation and Conservation Authority, as well as California State Parks, and a federal park unit, Santa Monica Mountains National Recreation Area, have matured in their working relationship to efficiently direct funding toward land purchases to piece together the remaining undeveloped land in the range. These groups and agencies continue to use organizing principles like the Backbone Trail, a single, continuous hiking trail from Point Mugu east to Will Rogers State Park in Pacific Palisades, or the Big Wild, a gateway

**TABLE 1** Notable examples within each major stakeholder category that drive the virtuous cycle of conservation in the Santa Monica Mountains.

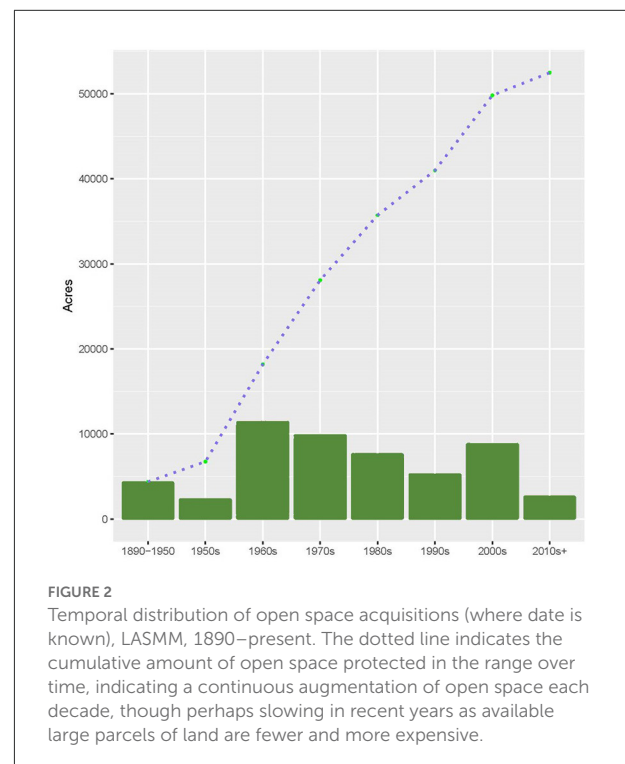
Ecological Research	Stunt Ranch Santa Monica Mountains Reserve (1995)
	RCD-SMM (1999)
	Puma Project/NPS (2002)
	La Kretz Field Station (2013)
Activist Groups and NGOs	Friends of the SMM, Parks and Sea Shore (1964)
	Sierra Club SMM Task Force (1972)
	Save Open Space (1990)
Supportive Legislators	City: Marvin Braude (1965)
	US: Thomas Rees (1966)
	County: Edmond D. Edelman (1975)
	US: Anthony Beilenson (1977)
	County: Zev Yaroslavsky (1994)
	US: Brad Sherman (1997)
	City: Paul Koretz (2009)
	State: Richard Bloom (2012)
	County: Sheila Keuhl (2015)
State Park Bonds	1964 (\$150M)
	1974 (\$250M)
	2000 (\$1.3B)
	2002 (\$2.6B)
	2018 (\$4.1B)
Land Acquisitions	Griffith Park (1896)
	Will Rogers SHP (1944)
	Pt. Mugu SP (1967)
	Cold Creek Canyon Preserve (1970)
	Topanga SP, Malibu Creek SP (1974)
	SMMNRA (1978)
	Paramount Ranch (1980)
	Jordan Ranch/Palo Comado (1994)
	Stunt Ranch Santa Monica Mountains Reserve (1995)
	Ahmanson Ranch/ULV (2003)
	King Gillette Ranch (2005)
	La Kretz Field Station (2013)
	Wallis Annenberg overcrossing (2022)
Land Management Entities	Santa Monica Mountains Conservancy (1980)
	TreePeople Land Trust (formerly Mountains Restoration Trust) (1984)
	Mountains Restoration and Conservation Authority (1985)
	California State Parks
	National Park Service
	University of California (UCLA)

(Continued)

**TABLE 1** Continued

Conservation Regulations and Ordinances	State: California Environmental Quality Act (1970)
	State: California Coastal Commission (1976)
	County of Los Angeles: Santa Monica Mountains Zone Local Coastal Program (1986, updated 2014)
	County of Los Angeles: Environmental Review Board of the Santa Monica Mountains (1992)
	City of Los Angeles: Mullholland Scenic Parkway Specific Plan (1992)
	County of Los Angeles: Santa Monica Mountains North Area Plan (2002, updated 2021)
	City of Malibu: Local Coastal Program (2002)
	County of Los Angeles: Oak Woodland Ordinance (2010- draft)
	City of Los Angeles: "Own a Piece of LA" Ordinance (2022)
	City of Los Angeles: Wildlife Ordinance (2022 – draft)

The legislators included were recognized in reports and media articles for their efforts to support conservation in the Santa Monica Mountains. See [Supplementary Table S1](#) for full list of Protected Areas and dates of first acquisitions.



park concept that includes the 10,000 acre Topanga State Park and other public lands at Encino Reservoir and Rustic, Sullivan, and Mission canyons.

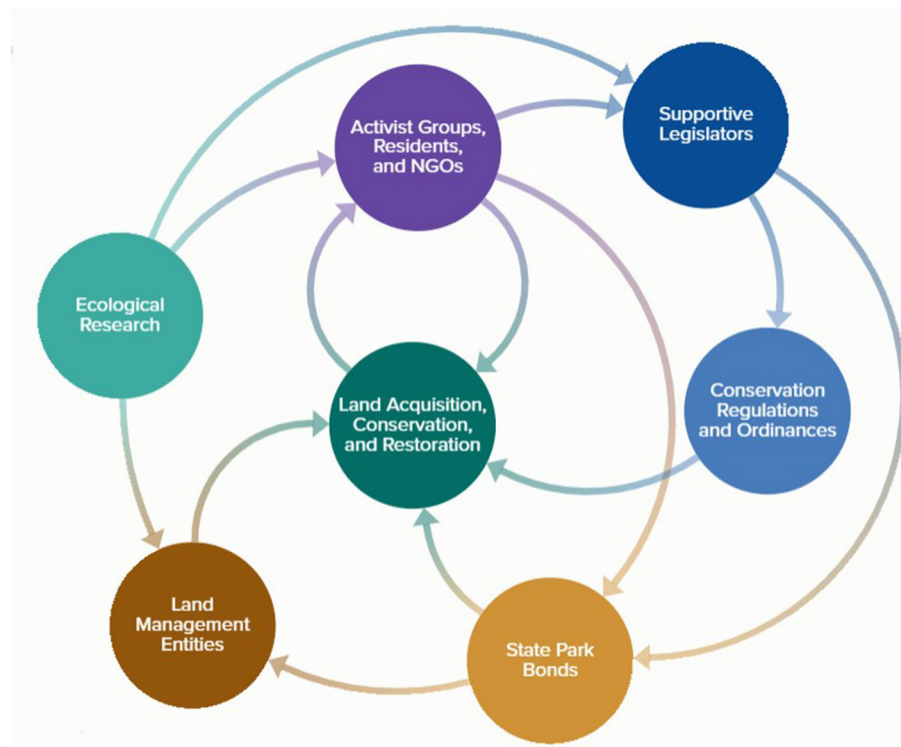


FIGURE 3

The virtuous cycle of land acquisition in the Santa Monica Mountains, showing the relationship between governmental officials and organizations, NGOs, and other stakeholders involved in open space prioritization and acquisition. Ecological research informs conservation priorities and educates stakeholder groups. Local activists and NGOs rally public support and identify threats and opportunities. NGOs endorse and support politicians and bond measures. Bond measures fund grants for local land acquisition and support management. Officials at multiple levels of government draft laws and bond measures for land protection. Multiple agencies own and manage land. A few purchase and set aside open space for public use. Access to open space inspires activists and NGOs to conserve more. See Table 1 for detail on each category.

A surge in ecological research occurred in the LASMM starting in the 1990s, with studies of habitat connectivity (Swenson and Franklin, 2000), urban-edge wildlife response (Sauvajot et al., 1998), large mammal movement and wildlife corridor use (e.g., Riley et al., 2003, 2021), rare fishes and stream ecology (Dagit et al., 2009), and detailed vegetation mapping of the entire range (AIS ESRI, 2007). This research has been led by a diverse group of scientists from more than a dozen agencies and NGOs, and has informed projects such as a wildlife crossing (vegetated bridge) over the 101 Freeway to assist in the genetic exchange of mountain lions (see Riley et al., 2021), which is under construction, at the cost of tens of millions of dollars (Anaya-Morga, 2021). In this way, ecological research has both aided—and reflected—the public's understanding of the importance of connecting and conserving these pieces of land.

Finally, the continued refinement and enforcement of laws and regulations aimed at conserving, rather than facilitating development of, raw land at both the municipal and county level across the LASMM seeks to ensure that these conservation acquisitions are encouraged (e.g., Santa Monica Mountains Conservancy given “first right

of refusal” on vacant, city-owned lands in the LASMM; Catanzaro, 2022). Since the mid-1970s, the California Environmental Quality Act (1970)<sup>1</sup> has required environmental review of most projects larger than a single-family home, and the California Coastal Act (1976)<sup>2</sup> strictly regulates development within five miles of the coast (which represents roughly half the area of the LASMM). Much of the land in the LASMM is located outside incorporated cities, in unincorporated portions of Los Angeles County; here, the administration of open space falls under the purview of the county's Department of Regional Planning, whose additional regulatory overlays include Local Area Plans and an Environmental Review Board (the latter staffed by local scientists and representatives of NGOs) to assess and reduce the impact of proposed development on open space within the LASMM.

1 California Public Resources Code §21000 et seq.

2 Public Resources Code Division 20 California.

## Discussion

The iconic public lands of the Santa Monica Mountains are visited and enjoyed by people from both the surrounding region and around the world, particularly at such tourist destinations as Griffith Park/ “Hollywood Sign” and Malibu Beach. Conservation efforts here have paid off for many species groups, as reflected in the persistence of high diversity in herptiles (Delaney et al., 2021) and breeding birds (Allen et al., 2016), as well as rare plants (Cooper, 2011); other groups, such as rare amphibians (Halstead et al., 2022), large mammals (Riley et al., 2006), and raptors (Cooper et al., 2020) have suffered extirpations and loss of genetic diversity and may require human intervention to persist long-term. And while individual activists or politicians such as Anthony C. Beilenson and Susan B. Nelson have been dubbed by media “the father (or mother) of the Santa Monica Mountains,” no single individual or group is responsible for the acquisition and continued protection of open space. Many stakeholders worked collaboratively and in tandem over time. The activities of each inform the others, encouraging more land to be acquired and protected each year. We depict this cycle in Figure 3.

Maintaining this virtuous cycle of land acquisition for conservation and public enjoyment will depend on supporting productive relationships between the public and government. The success of the LASMM over the past decades may serve as a model for other areas of California and beyond, as human needs are balanced with those of the natural environment. However, the acquisition-conservation model must also be sustainable, as protected areas may not remain protected forever, given the demands of forces such as recreation and the perceived need for housing. Today, large areas of open space in the LASMM—particularly those close to dense urban areas—remain off-limits to many residents (see Wolch et al., 2005; Byrne et al., 2009). Access to open space is also hampered by early decisions to permanently close access to open space to the public, not for protection of wildlife and biodiversity, but for security concerns (e.g., nearly 1,400 acres of open space in the eastern LASMM are fenced off, with entry strictly controlled by the Los Angeles Department of Water and Power, though a recently-opened perimeter path around Hollywood Reservoir provides some access). Because public support for land and wildlife conservation actions appears to be linked to one’s own activities in nature (including bird-watching; see Cooper et al.,

2015; Rutter et al., 2021), the dearth of accessible open space in some areas may eventually cause drag on the virtuous cycle of land conservation by impacting residents’ sense of connection to and willingness to advocate for continuing to conserve open space in the LASMM.

## 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

DC conceived of the paper and wrote the majority of the manuscript. NK assisted in the analysis, provided editing, and improved the graphics. BD and FO assisted in GIS data collection and spatial analysis. All authors contributed to the article and approved the submitted version.

## Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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## Supplementary material

The Supplementary Material for this article can be found online at: <https://www.frontiersin.org/articles/10.3389/frsc.2022.923946/full#supplementary-material>

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## OPEN ACCESS

## EDITED BY

Sophie S. Parker,  
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## REVIEWED BY

Joscha Beninde,  
University of California, Los Angeles,  
United States  
Stephanie Pincetl,  
University of California, Los Angeles,  
United States  
Amy Collins,  
University of California, Davis,  
United States

## \*CORRESPONDENCE

Amanda J. Zellmer  
zellmer@oxy.edu

## SPECIALTY SECTION

This article was submitted to  
Urban Greening,  
a section of the journal  
Frontiers in Sustainable Cities

RECEIVED 26 May 2022

ACCEPTED 22 September 2022

PUBLISHED 14 October 2022

## CITATION

Zellmer AJ and Goto BS (2022) Urban  
wildlife corridors: Building bridges for  
wildlife and people.  
*Front. Sustain. Cities* 4:954089.  
doi: 10.3389/frsc.2022.954089

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# Urban wildlife corridors: Building bridges for wildlife and people

Amanda J. Zellmer<sup>1,2\*</sup> and Barbara S. Goto<sup>2</sup>

<sup>1</sup>Department of Biology, Occidental College, Los Angeles, CA, United States, <sup>2</sup>Arroyos & Foothills  
Conservancy, Pasadena, CA, United States

Urbanization is rapidly expanding across the globe, leading to increasing threats to wildlife in and around cities. Wildlife corridors are one strategy used to connect fragmented wildlife populations; however, building wildlife corridors in urban areas remains a challenge because of the number of barriers between habitat patches and the extensive number of property owners and stakeholders involved. Successful urban wildlife corridor conservation thus requires a collaborative approach and a cohesive plan that transcends municipal boundaries. Here we demonstrate how urban wildlife corridor conservation can provide a unique opportunity to build bridges not only for wildlife but also among scientists, non-profits, government agencies, and communities. Our case study centers on the conservation of a network of wildlife corridors in one of the world's megacities, Los Angeles, and the positive feedback loop sparked by collaboration between research and non-profit work. We discuss the benefits of and challenges to building complex collaborations for the purpose of strengthening urban resilience and redesigning sustainable cities.

## KEYWORDS

habitat connectivity, urban ecology, urban wildlife, conservation, Los Angeles

## Introduction

With the continued growth in cities worldwide, urban ecosystems are rapidly expanding (Grimm et al., 2008). More than 55% of the world's population already live in urban areas and this is expected to grow to over 68% by 2050 (World Urbanization Prospects: The 2018 Revision, 2018). This continued growth increasingly threatens wildlife living in and around cities, such as through isolation in urban green spaces due to habitat fragmentation, increased mortality as a result of vehicle-wildlife collisions, increased exposure to toxins and poisons, exposure to diseases, and competition with introduced species (Kowarik, 2011). Managing urban wildlife populations is not only important for the conservation of these species, but also for the people living in cities. Urban expansion increases the risk for more human-wildlife conflicts (Woodroffe et al., 2005; Skogen et al., 2008), necessitating mitigation to prevent conflict. At the same time, urbanization also reduces opportunities for positive human-wildlife

interactions as species become extirpated from isolated habitat patches. This is a concern for city inhabitants because positive human-wildlife interactions can have numerous psychological benefits (Curtin, 2009). Moreover, people living in biologically impoverished areas are subject to increasing “extinction of experience” with nature (Miller, 2005), which in turn may impact their engagement with conservation action (Morrison, 2015, 2016). As such, the loss of wildlife from cities may lead to a breakdown in the virtuous cycle, the positive feedback loop where the societal benefits of biodiversity conservation catalyze increased conservation action (Morrison, 2015, 2016). Because access to nature remains deeply inequitable across cities (Williams et al., 2020), these losses of urban wildlife will disproportionately impact low-income communities and communities of color. Thus, it is essential that we develop successful strategies for urban wildlife conservation that strengthen the virtuous cycle between biodiversity conservation and the people inhabiting cities, both for the sake of wildlife and humans.

Wildlife corridors are frequently used in conservation as a tool to connect wildlife populations that have become isolated because of human-mediated habitat fragmentation (Bennett, 1999). Despite early debate (Beier and Noss, 1998; Haddad et al., 2000), success of wildlife corridors has been documented for several species, with increased movement between isolated populations, increased genetic admixture (Gilbert-Norton et al., 2010; Resasco, 2019), and, when used in conjunction with other mitigation measures, reduced human-wildlife conflicts such as vehicle-wildlife collisions (Rytwinski et al., 2016). Moreover, corridors will be increasingly essential for wildlife to be able to respond to changing climates (Rudnick et al., 2012; Costanza and Terando, 2019; Littlefield et al., 2019; Jennings et al., 2020; Schloss et al., 2022).

Although there is some evidence of the success of wildlife corridors in urban areas (Shwartz et al., 2014; Adams et al., 2017), within cities it is rarely possible to connect habitat patches with a single bridge. Habitat fragments are frequently separated by multiple roads, multiple land parcels with different owners, and may even be separated across different jurisdictions. Furthermore, land ownership and usage within urban areas can change rapidly and unexpectedly. As a result, approaches to wildlife corridor design that are recommended in more rural locations may not be appropriate or successful in urban areas. For instance, recommendations for corridor design include maximizing width of the corridor and exclusion of human development and activity from the corridor (Bond, 2003). Yet, in urban areas, such recommendations are often impossible to achieve. Thus, traditional wildlife corridors, such as a bridge between two conservation landscapes (Beier and Loe, 1992), may not be adequate for protecting connectivity in cities.

These challenges are further compounded by more general difficulties of conservation within urban areas (Shwartz et al.,

2014), which continue to limit the success of redesigning cities to be more wildlife-inclusive (Kay et al., 2022). First, in a landscape so heavily dominated by humans, it can be especially challenging to balance the, sometimes, competing needs of people and wildlife (Goswami and Vasudev, 2017; Turo and Gardiner, 2020). Second, biodiversity conservation already requires successful collaboration between multiple stakeholders (Gavin et al., 2018), but this challenge is only intensified in urban settings where, due to smaller parcel sizes, there are far more stakeholders. Third, disciplinary silos between researchers, conservation practitioners, land planners, and policymakers create additional barriers for urban conservation (Kay et al., 2022). Finally, as urban sprawl expands and begins to connect disjunct cities, reducing the remaining open spaces between jurisdictional boundaries (Kraas, 2008), enacting a cohesive conservation plan becomes even more arduous. Conservation within urban areas thus requires a unifying framework to bridge efforts across jurisdictions and stakeholders.

Establishing priority areas for urban wildlife corridors, where efforts are coordinated to preserve multiple pathways and stepping-stones of connectivity, may be one way to facilitate conservation in urban areas. Utilizing a combination of approaches, including green infrastructure, backyard habitat restoration, land acquisitions, and conservation partnerships, urban wildlife corridor conservation has the potential to enhance wildlife connectivity while simultaneously building bridges between the vast network of stakeholders in cities (Figure 1). Although the need for connectivity has long been a primary recommendation for urban conservation (Soulé, 1991), there are few case studies where the process of urban corridor conservation has been fully documented, especially when considering connectivity across multiple land parcels and jurisdictions.

Here, we present a case study of urban wildlife corridor conservation along the eastern edge of the Rim of the Valley Corridor, a series of mountains and open space encircling part of the Metropolitan Los Angeles Area (Figure 2; NPS, 2015), as a framework for addressing the challenges of biodiversity conservation in urban areas. We first defined an urban wildlife corridor priority area to create a database for researching and monitoring connectivity. To track wildlife presence within the priority area, we established a transect of remote-triggered camera traps and collated community science observations of all terrestrial mammal species. We then compared all vacant privately owned land parcels to prioritize conservation needs and make evidence-based decisions for land acquisitions. To encourage backyard restoration on developed land parcels, we initiated a native plant distribution project with local volunteers. Finally, with our database, we established an extensive outreach and education program to build community across stakeholders within and adjacent to the corridor. We use our results to discuss the potential for conserving wildlife corridors in urban areas and highlight remaining challenges.

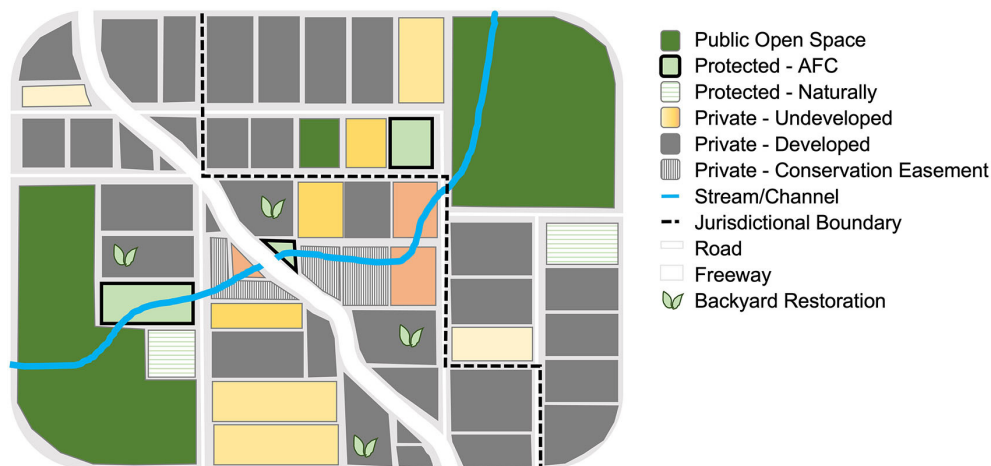


FIGURE 1

Illustration of the complexity of conserving urban wildlife corridors. Land parcels are shown on a street map with roads (white lines), jurisdictional boundaries (black dashed lines) and an urban stream channel (blue line). The culvert under the freeway (thick white line) would be assessed as a potential wildlife passage corridor. Land parcels are colored based on their ownership and status. Preserved public open space includes parks and undeveloped natural habitat (green). Privately owned undeveloped parcels are split into parcels that have been acquired and therefore protected (light green, black border), parcels that have been protected naturally due to land ordinances (green horizontal stripes), and parcels that have been ranked for corridor conservation need ranging from high (orange) to low (light yellow). Parcels in orange would be ground-truthed to assess conservation potential. Privately owned occupied parcels include developed parcels (grey) and parcels with conservation easements (grey vertical stripes). Properties on which landowners have participated in backyard restoration projects are also indicated (green leaf).

## Methods

### Study area

The Greater Los Angeles Area provides an ideal opportunity to highlight the need for and strengths of an urban wildlife corridor framework. Los Angeles is located within the California Floristic Province, an area that is recognized as one of the world's biodiversity hotspots (Cincotta et al., 2000; Myers et al., 2000). Moreover, the topographical complexity of the Greater Los Angeles Area creates numerous physical environments, which gives rise to an impressive amount of biological diversity, with many endemic and endangered species (Dobson et al., 1997).

At the same time, the region has experienced rapid urbanization as a result of population growth and economic expansion (Syphard et al., 2005) and as of 2020 was home to over 18.7 million people ([www.census.gov](http://www.census.gov)). The layout of the Greater Los Angeles Area exhibits a unique example of the impacts of urban sprawl, encompassing at least 177 communities (Scott, 1995) connected by a vast network of major highways (Fraser et al., 2019). The urban sprawl extends these densely populated areas right up to large undeveloped natural areas, including the Angeles National Forest.

The impacts of urban habitat fragmentation on wildlife populations within Los Angeles has been extensively documented (Riley et al., 2006; Delaney et al., 2010; Ernest et al., 2014; Poessel et al., 2014; Benson et al., 2016; Fraser et al., 2019). Notably, Los Angeles is one of only two megacities in the

world that is home to large predatory cats, with the well-known P-22 mountain lion (*Puma concolor*) living in Griffith Park and other GPS collar tracked mountain lions in fragmented habitat patches nearby (Riley et al., 2014b, 2021). Despite the resilience of these large cats, there remains an urgent need for re-establishment of connectivity (Benson et al., 2016). Recent studies have documented that mountain lions within these habitat patches show evidence of inbreeding (Huffmeyer et al., 2022) and experience mortality risks associated with humans (Benson et al., 2020). Mountain lions provide just one example of the many species that would benefit by improving connectivity among fragmented habitat areas within the Los Angeles metropolis.

Landscape connectivity across California (Spencer et al., 2010) and in Southern California (Beier et al., 2006) has long been a priority for conservation. Notably, however, many efforts to establish priority areas for region-wide connectivity initially excluded highly urbanized areas. More recently, there has been increased effort at setting priorities for conservation of connectivity within the Greater Los Angeles Area. In 2008, the National Park Service, as directed by Congress through the Consolidated Natural Resources Act of 2008 (P.L. 110-229-May 2008), began a study to assess the significance of the Rim of the Valley Corridor, which generally includes the mountains and foothills encircling the San Fernando, La Crescenta, Santa Clarita, Simi, and Conejo Valleys in California (NPS, 2015).

The eastern edge of the Rim of the Valley Corridor is of particular interest because it weaves through Los Angeles. Here,



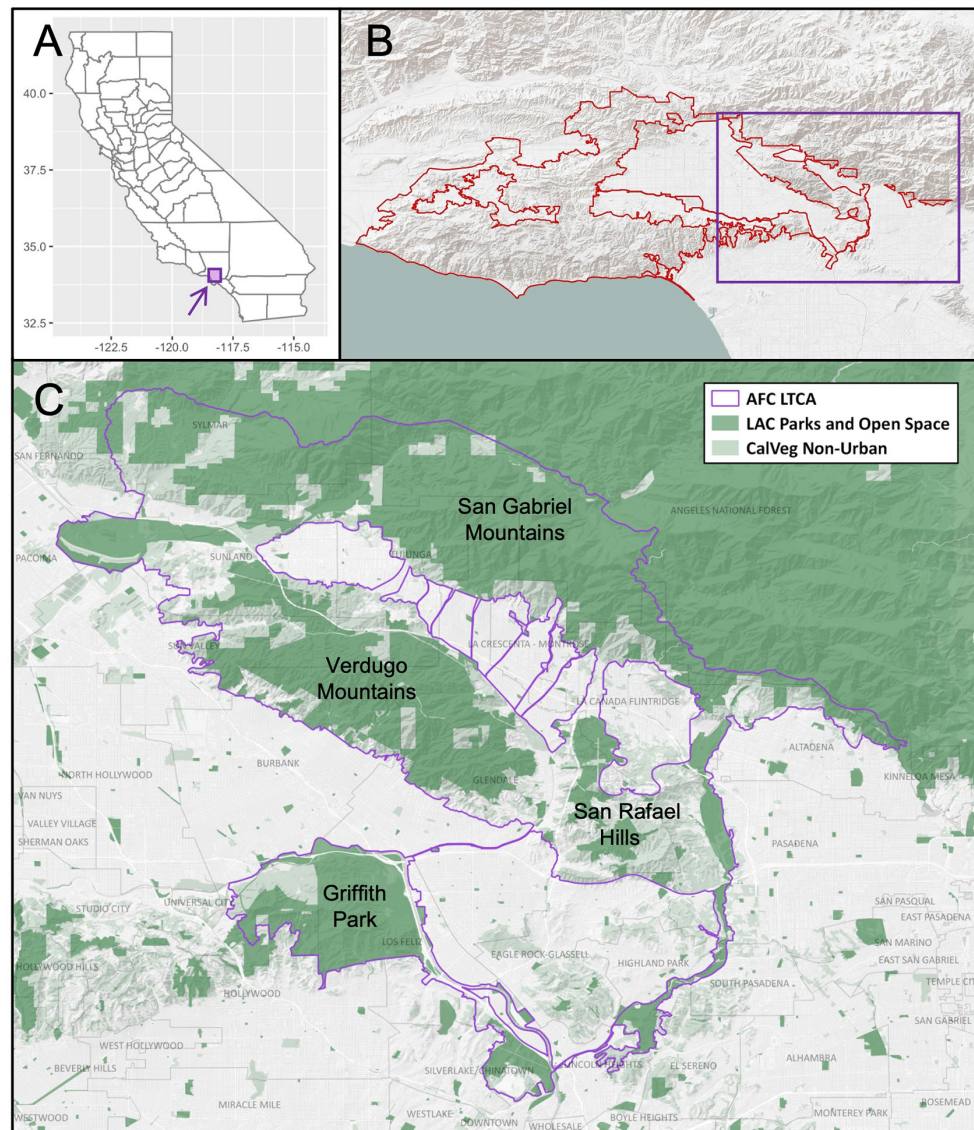


FIGURE 2

Map of the Arroyos & Foothills Conservancy's Long Term Conservation Area (AFC LTCA) on the eastern edge of the Rim of the Valley Corridor, a case study of an urban wildlife corridor. (A) Location of the study area (purple box) within California. (B) Location of the study area (purple box) in reference to the Rim of the Valley Corridor (red outline). (C) Extent of the LTCA in the Greater Los Angeles area (purple outline). Existing public parks and open space (LA County) are shown in dark green. Remaining land is classified into urban (white) and non-urban (light green; CalVeg). Key areas of protected open space needing connection include the Verdugo Mountains, San Gabriel Mountains, the San Rafael Hills, and Griffith Park.

large blocks of natural land, including the Verdugo Mountains, are entirely surrounded by development (Figure 2), yet still are home to wide ranging species, such as mountain lions (Riley et al., 2021). This region is also important for migrating birds (Terrill et al., 2021) and other species that require migration or dispersal corridors, such as monarch butterflies (*Danaus plexippus*). The Arroyos & Foothills Conservancy (AFC) - a land trust dedicated to conservation, restoration, and education - began to lead a collaborative effort to study,

monitor, and acquire properties within this region in 2012 in an effort to preserve and restore connectivity. In 2017, AFC formed a partnership with Occidental College to begin a research program studying connectivity throughout the region, primarily for medium to large terrestrial mammals. Through this collaboration, we combine research, land acquisition, restoration and stewardship, and outreach and education in order to conserve wildlife corridors within an urban area (Figure 3).

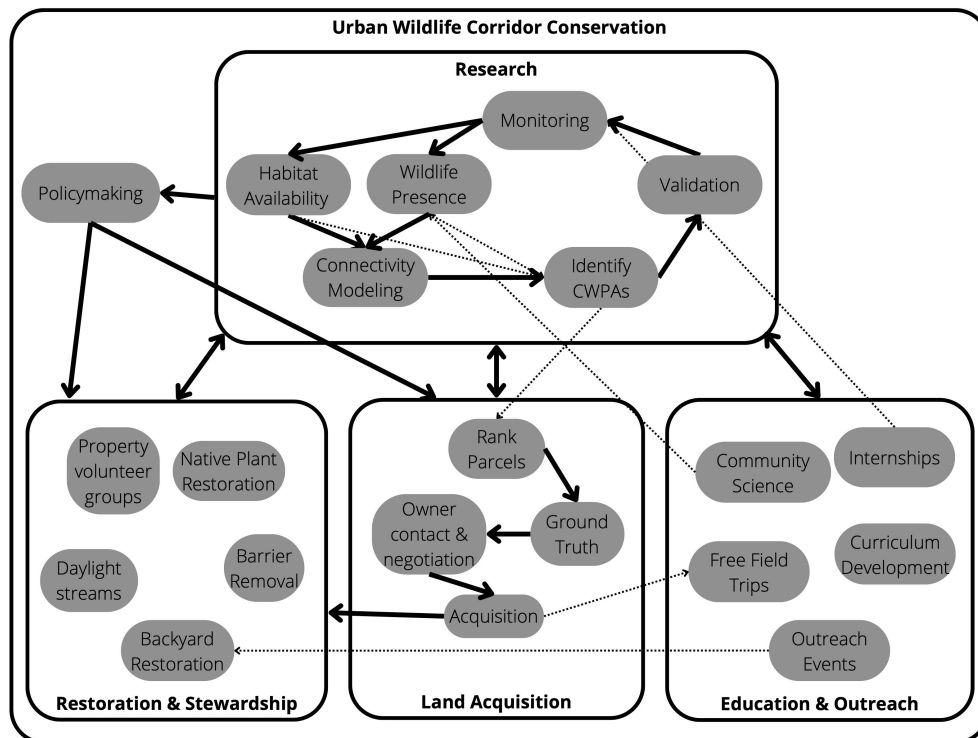


FIGURE 3

Graphical abstract of urban wildlife corridor conservation strategy. Urban wildlife corridor conservation requires a multi-pronged, collaborative approach including research, land acquisitions, restoration and stewardship, and outreach and education. CWPAs refers to critical wildlife passage areas.

## Urban wildlife corridor assessment

We began by designating a priority area for urban wildlife corridor conservation, which we call the Long-Term Conservation Area (LTCA). The LTCA was initially established within a buffer around each of the undeveloped, publicly owned protected habitat patches in the eastern edge of the Rim of the Valley Corridor and the narrowest connections between these protected spaces. The LTCA was revised periodically to incorporate new data, including expanding the extent of the focal area to reach additional protected spaces as well as adding additional routes of connectivity as new data came available.

Within the LTCA, we investigated the potential for wildlife passage between preserved open spaces using a combination of GIS data, wildlife observations, and scouting potential routes of wildlife movement. To begin, we used the CalVeg layer (USDA, 2004) and Google Earth to identify the most continuous and direct routes for wildlife movement between preserved open spaces. Specifically, we created a polygon shapefile of Critical Wildlife Passage Areas (CWPAs) outlining all areas where undeveloped habitat appeared, based on our remotely sensed data, to be physically connected to preserved open spaces, or where there was potential for connectivity (provided some

remediation was completed). Once these areas were visually identified from the maps, we began the process of ground-truthing by collating wildlife observations (described below), walking the potential routes, evaluating habitat quality (e.g., availability of habitat, availability of water sources, low noise and light pollution), and identifying both conduits (e.g., underpasses, culverts, channels) and barriers (e.g., fencing, roads, structures) to wildlife movement. With these data, we then refined the CWPAs shapefile to focus on regions that were deemed most suitable to wildlife movement. In addition, we created a polyline shapefile of potential routes for wildlife movement through the CWPAs. The CWPAs and potential routes were continually re-evaluated as new information was gained or as conditions changed.

To assess wildlife presence within and adjacent to the LTCA, we monitored terrestrial mammals using remote-triggered camera traps and community science data. We established a transect through the LTCA where we deployed cameras beginning in October 2018. The cameras were set up in month-long sampling periods four times per year through January 2022 following the Urban Wildlife Information Network (UWIN) protocol (Magle et al., 2019) and opportunistically at other times of the year. Cameras were also deployed opportunistically



when invited by community partners or at the discretion of landowners. The make and model of the cameras included Browning Dark Ops Elite HD, Browning Strikeforce HD, and Reconyx Hyperfire. Photos were uploaded to the UWIN Database and classified by at least two people, including both researchers and trained volunteers (Katrak-Adefowora et al., 2020). Photos with inconsistent classifications between the first two observers were validated by a third person. Beginning in 2021, we pre-filtered out empty images and images with humans using the machine learning photo detection pipeline, Megadetector (Beery et al., 2019), using a detection confidence cutoff of 0.8.

In addition, we compiled community science observations of terrestrial mammals within and adjacent to the priority conservation area by establishing a project on the iNaturalist database ([www.inaturalist.org/projects/la-wildlife-connectivity](http://www.inaturalist.org/projects/la-wildlife-connectivity)). We downloaded all observations with a spatial locality accuracy of 1 km or less between January 2012 and August 2022. We included only research grade observations. In addition, we began monitoring social media sites including Facebook groups, Ring Neighbors, and Nextdoor in 2019 and recorded all wildlife observation posts with spatial locality information. We quantified the total number of observations of each terrestrial mammal species recorded within the LTCA and within CWPAs.

## Parcel evaluation

After establishing the perimeter of the LTCA, we collated data on all land parcels within the perimeter. We downloaded parcel information for all properties within the LTCA from the Los Angeles County assessor's office and created a parcel database. Parcel information included parcel boundaries, landowner information, and information about publicly owned parcels.

We then compiled spatially-explicit data on key features relating to the suitability of habitat for wildlife movement (Duttweiler, 2021 unpublished), including functional landscape connectivity, presence of rare or sensitive plant or animal species, presence of rare or sensitive terrestrial communities, presence of critical or wetland habitat, presence of rivers or streams, presence of desirable vegetation, coincident with areas prioritized by others, coincident with CWPAs, and proximity to roadway bridges, drainages, and other protected area.

To assess conservation needs and prioritize land acquisitions, we developed a Parcel Evaluation Tool (PET) (Duttweiler, 2021 unpublished), which allowed us to compare land on a parcel-by-parcel basis. We used the PET to evaluate all privately owned, vacant land parcels as well as some privately owned developed parcels with sufficient habitat in a CWA where conservation easements may contribute to conservation of connectivity. Parcels were ranked based on weighted scores for the factors described above. Land parcels were then ranked

into four quartiles. Highly ranked properties identified by the PET along with opportunistic properties were ground-truthed to validate our rankings and assess conservation potential. To accomplish this, we gathered more fine-scale habitat and wildlife data and solicited expert review from wildlife biologists, land development experts, and urban planners. In addition, we reached out to individual landowners, gathered wildlife observation data using remote-triggered camera traps, talked to neighbors, and walked the communities where these parcels are located. Parcels were then classified with a conservation status: Protected, Protected Naturally, Protected by Others, Monitor, Revisit, Passed, Under Contract, Target—Owner Contacted, Target—Owner Not Contacted. Once a property was identified as a target for pursuit, we began the process of land acquisition, negotiating conservation easements, or supporting others in acquiring the property for conservation. As properties were acquired and new information became available, we reassessed parcel prioritization to monitor changes in conservation needs across the LTCA.

## Restoration and stewardship

For acquired properties, we began habitat restoration and monitoring of wildlife with remote-triggered camera traps. We also recruited neighborhood volunteers to help monitor and manage each property. We then developed an adaptive management plan for each property including restoration as well as fire fuel reduction and any other projects identified by our Friends groups and stewardship advisors as beneficial to the overall biodiversity of the property.

In addition to acquiring and conserving properties, we facilitated backyard restoration within and adjacent to the LTCA. One example of this is the Monarch Recovery Program. We distributed native milkweed plants (*Asclepias fascicularis* and *A. eriocarpa*) to volunteers, who then planted their milkweed plants in their backyards or neighborhoods. Volunteers were then instructed to upload the coordinates of their milkweed plants into an online ArcGIS database. We mapped the distribution of the planted milkweed to evaluate the spatial extent of the plantings. After 6 months, volunteers were asked to report the status of their milkweed plants.

## Collaboration, outreach and education

To enable conservation across the LTCA, we shared our data with stakeholders. Data were shared through various approaches including meetings with stakeholders, public comments on policy recommendations, presenting at conferences, and publishing in scientific journals. We provided data by request, including connectivity assessments and species observations.

In addition, we developed an outreach and education program to disseminate the results of our research and to empower residents within and adjacent to the urban wildlife LTCA to contribute to conservation. We hosted field trips with local schools and organizations on AFC owned properties within the corridor. We hosted volunteer events to enable restoration on acquired properties. Wildlife Movement curriculum was developed to teach students about the importance of the corridor for wildlife movement and made available to teachers. We attended community events and contributed to exhibits to share the results of our research with the broader community. We established an internship program for high school and college students for hands-on training in urban wildlife corridor conservation and research. Finally, we trained volunteers to assist in urban wildlife corridor conservation. We held frequent meetings to bring researchers, conservation practitioners, volunteers, and interns together to discuss research, learn new approaches to conservation, and collaboratively set goals for conservation and research within the corridor.

## Results

### Urban wildlife corridor assessment

Within our study area, we identified a total of  $2.43 \times 10^5$  km<sup>2</sup> of land as a priority area for urban wildlife corridor conservation (LTCA; [Figure 2](#)). The LTCA traversed 22 jurisdictions within Los Angeles County. To date, we have identified 17 CWPAs within the LTCA and 156 potential routes for wildlife movement.

Camera trap and community science observations confirmed numerous wildlife species are present within this key urban wildlife corridor. Between September 2018 and February 2022, we documented 25,333 photos with positive wildlife detections of at least 63 wildlife species on our camera traps across 43 unique sites within the LTCA. This includes at least 19 terrestrial mammal species ([Table 1](#)), as well as many bird, amphibian, and reptile species. The community science dataset included an additional 8,360 mammal observations. Of those observations, 3,933 were within the LTCA and 1,644 were within CWPAs, representing 31 species ([Table 2](#)).

### Parcel evaluation

Within the LTCA, we identified a total of 28,994 unique land parcels. We evaluated 5,461 of these parcels with the PET, with 2,013 parcels within or overlapping CWPAs. To date, we have reviewed a total of 734 parcels for conservation potential. Of these parcels, we assigned 728 a conservation status. AFC acquired and preserved 29 parcels, supported the acquisition of 23 others, and is currently under contract to acquire 14 more.

**TABLE 1** Number and percentage of sites at which each terrestrial mammal species was observed along a transect of remote-sensored wildlife cameras between September 2018 and February 2022.

Scientific name	Common name	#	%
<i>Canis latrans</i>	Coyote	35	81
<i>Lynx rufus</i>	Bobcat	28	65
<i>Odocoileus hemionus</i>	Mule deer	26	60
<i>Sylvilagus bachmani</i> or <i>S. audubonii</i>	Rabbit	24	56
<i>Mephitis mephitis</i>	Striped skunk	20	47
<i>Otospermophilus beecheyi</i>	California ground squirrel	20	47
<i>Procyon lotor</i>	Raccoon	20	47
<i>Sciurus niger</i>	Fox squirrel	17	40
Various species*	Rodent	15	35
<i>Dipodomys spp.</i>	Kangaroo rat	10	23
<i>Didelphis virginiana</i>	Virginia opossum	9	21
<i>Sciurus griseus</i>	Western gray squirrel	8	19
<i>Urocyon cinereoargenteus</i>	Gray fox	5	12
<i>Ursus americanus</i>	Black bear	5	12
<i>Neotamias merriami</i>	Merriam's chipmunk	1	2
<i>Puma concolor</i>	Mountain lion	1	2

\*Four distinct rodent morphologies were observed but could not be confidently identified to species.

The total number of cameras (#) and percent of cameras (%) on which each species was observed is listed.

Additionally, we deemed nine properties as Protected Naturally due to land ordinances or access issues, although this status is continually re-evaluated.

### Restoration

As of July 2022, a total of 371 volunteers signed up to adopt milkweed plants. Of those volunteers, 209 adopted milkweed plants with the remaining on a waitlist. A total of 172 volunteers reported their results in the online portal representing 172 unique locations where milkweed was planted. The planted milkweed covered the full extent of our LTCA and extended outside the LTCA as well ([Figure 4](#)). Of the 172 initial respondents, 83 responded to our second survey to report the status of their milkweed plants after approximately 6–7 months.

### Collaboration, outreach and education

We invited a total of 40 different types of stakeholders to participate in urban wildlife corridor conservation

**TABLE 2** Terrestrial mammal species observed by community scientists within AFC's Long Term Conservation Area (LTCA) and within the Critical Wildlife Passage Areas (CWPAs).

Scientific name	Common name	LTCA	CWPAs
<i>Otospermophilus beecheyi</i>	California ground squirrel	722	495
<i>Sciurus niger</i>	Fox squirrel	705	374
<i>Odocoileus hemionus</i>	Mule deer	619	204
<i>Canis latrans</i>	Coyote	513	122
<i>Sylvilagus audubonii</i>	Desert cottontail rabbit	447	263
<i>Sciurus griseus</i>	Western gray squirrel	188	19
<i>Thomomys bottae</i>	Botta's pocket gopher	182	48
<i>Lynx rufus</i>	Bobcat	161	32
<i>Mephitis mephitis</i>	Striped skunk	89	18
<i>Urocyon cinereoargenteus</i>	Gray fox	66	0
<i>Procyon lotor</i>	Raccoon	48	9
<i>Ursus americanus</i>	Black bear	40	12
<i>Didelphis virginiana</i>	Virginia opossum	33	8
<i>Puma concolor</i>	Mountain lion	24	4
<i>Scapanus latimanus</i>	Broad-footed mole	17	7
<i>Neotamias merriami</i>	Merriam's chipmunk	14	0
<i>Sylvilagus bachmani</i>	Brush rabbit	12	5
<i>Mus musculus</i>	House mouse	8	7
<i>Rattus norvegicus</i>	Brown rat	7	2
<i>Tadarida brasiliensis</i>	Mexican free-tailed bat	6	1
<i>Microtus californicus</i>	California vole	5	2
<i>Rattus rattus</i>	Black rat	4	4
<i>Lasiurus cinereus</i>	Hoary bat	3	3
<i>Neotoma macrotis</i>	Big-eared woodrat	3	2
<i>Parastrellus hesperus</i>	Canyon bat	2	0
<i>Bassariscus astutus</i>	Ringtail	1	0
<i>Chaetodipus californicus</i>	California pocket mouse	1	0
<i>Neotoma fuscipes</i>	Dusky-footed woodrat	1	1
<i>Nyctinomops macrotis</i>	Big free-tailed bat	1	0
<i>Oryctolagus cuniculus domesticus</i>	Domestic rabbit	1	1
<i>Peromyscus maniculatus</i>	Deer mouse	1	0

Number of community science observations for each species from iNaturalist and social media posts from Facebook, Nextdoor, and Ring Neighbors.

through stakeholder meetings or outreach and education events (Table 3).

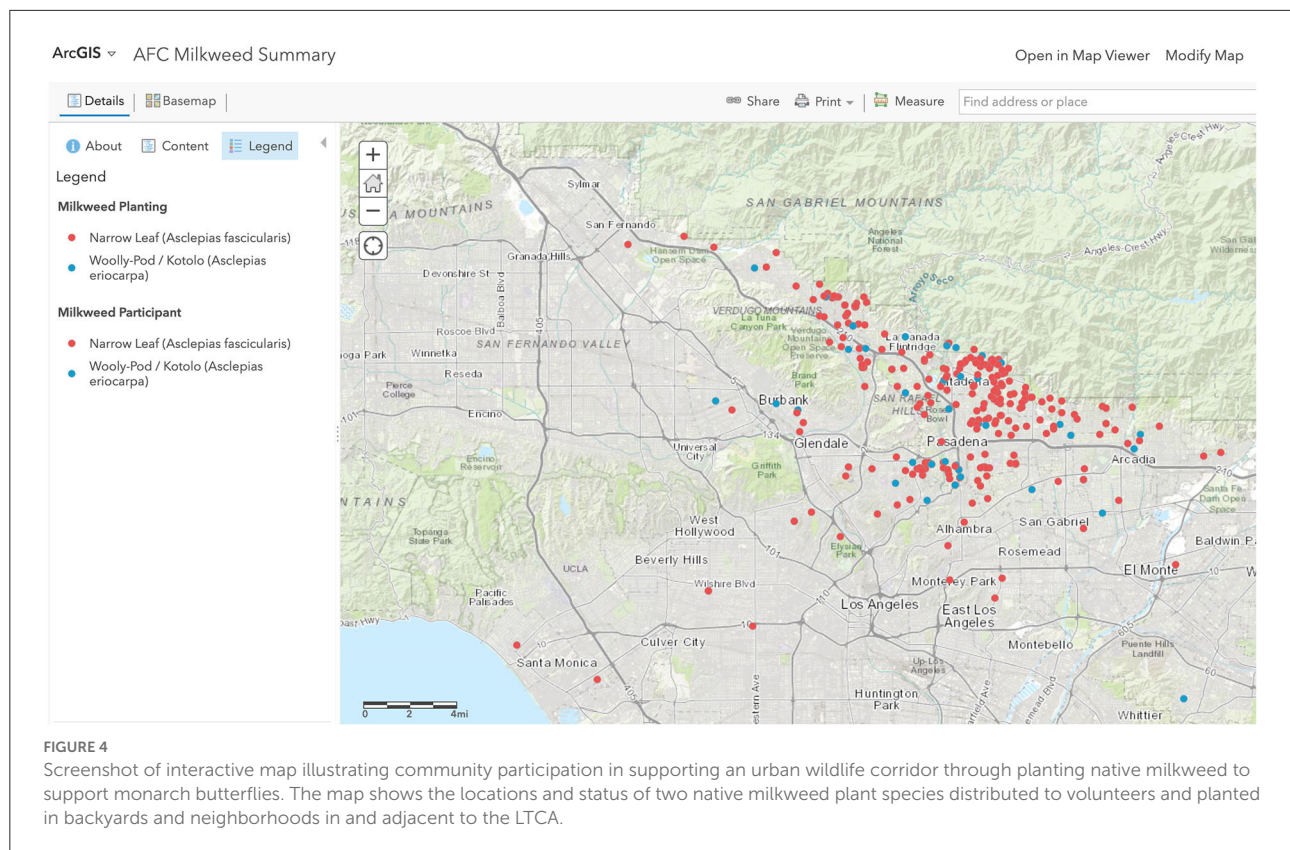
## Discussion

Wildlife corridors are increasingly being used to connect fragmented populations but how this is accomplished in urban areas, where conservation of connectivity requires coordination across multiple barriers and land parcels as well as the collaboration of numerous stakeholders, remains a challenge. We set out to use an evidence-based framework for conserving urban wildlife corridors in the Greater Los Angeles Area.

We found that the eastern edge of the Rim of the Valley Corridor alone includes over 28,000 land parcels and extends through a total of 22 jurisdictions. These results demonstrate the extraordinary number of properties and landowners as well as the extensive research and data needed to coordinate conservation of wildlife corridors within an urban area. As such, the results highlight the need for a cohesive framework for conservation. By establishing a priority area for conservation of landscape connectivity (LTCA) and building a database of all land parcels within that area, we were able to build a structure for gathering data, prioritizing conservation needs, and integrating research, conservation, and policy. With our framework in place and these data in hand, we were able to efficiently contribute to conservation of habitat for wildlife movement within one of the world's megacities.

Central to our approach to urban wildlife corridor conservation is the preservation and restoration of remaining undeveloped land through acquisitions and conservation easements to prevent further loss of connectivity. Notably, we identified over 5,460 privately owned vacant lots within the priority corridor area. While these lots vary in the quality and potential for supporting wildlife movement, future development of some of these properties would weaken connectivity throughout the region. By comparing land parcels based on their potential to contribute to wildlife connectivity, we were able to monitor and identify priority parcels for conservation. To date, we have acquired or assisted in the acquisition of 52 parcels. These parcels constitute a total of 1.19 km<sup>2</sup> of land, with 19 parcels within or overlapping CWPAs. Some of the parcels that were conserved fall within narrow strips of remaining habitat that connect two protected open spaces and had these parcels not been conserved, connectivity would have been severed. However, additional acquisitions and easements will be needed to guarantee preservation of structural connectivity throughout the region for the long term.

In addition to land acquisitions, we identified numerous potential routes along which connectivity will need to be restored, particularly across roads and fenced areas. A number of solutions have been suggested for mitigating the impacts of roads on wildlife connectivity, with wildlife bridges being a primary example (Riley et al., 2014a). However, as the construction of wildlife bridges remains an expensive option, lower cost options will be necessary to restore connectivity across the many different roads bisecting cities. One opportunity for restoring connectivity is through remediation of channelized stream beds, which have the potential to provide physical connections between protected open spaces, undercutting barriers such as roads and bypassing urban development. Previous research has shown that some species in Southern California will utilize culverts and undercrossings depending on the design of the undercrossing as well as the availability of habitat (Ng et al., 2004); however, there is little research on the extent to which wildlife will then continue on to use channelized



stream beds for moving between habitat patches. Although we documented the presence of multiple species within and along urban channels using camera traps and community science observations, further research is needed to fully evaluate the extent to which channels are utilized for wildlife movement and which channel designs best improve functional connectivity. Habitat restoration, fence removal, and setbacks could make urban channels more conducive to wildlife movement. However, additional mitigation strategies will be needed in areas where there are no suitable underpasses or culverts.

Simply building structural connectivity between habitat patches will not be enough to guarantee functional connectivity for wildlife (Baguette and Van Dyck, 2007), and this may be especially a concern in cities where corridors may have degraded habitat or the presence of humans may reduce permeability to wildlife movement. Evaluation of the effectiveness of wildlife corridors is essential for providing feedback to policymakers and practitioners as to whether these corridors are functioning as intended (Caro et al., 2009; Cushman et al., 2013; Brodie et al., 2016; Bond et al., 2017). Our results demonstrate that numerous wildlife species continue to persist throughout our proposed corridor region (Tables 1, 2). More importantly, our research suggests that wildlife, and even mountain lions and other species that typically avoid urban areas, are present within our defined CWPAs (Table 2). These results bolster previous

research on wildlife movement within Los Angeles (Ng et al., 2004; Riley et al., 2021), providing additional evidence for the need to conserve connectivity in this region. Yet, for many species and across many parts of the world, urban biodiversity remains understudied (Magle et al., 2012; Collins et al., 2021). A dedicated focus on modeling and assessing how wildlife use urban wildlife corridors would help fill important gaps in urban biodiversity research while also providing expertise needed for conservation planning.

In addition to creating corridors for wildlife, building a framework for monitoring and protecting habitat for wildlife corridors in an urban area provides an opportunity to build bridges between all the stakeholders involved in urban conservation. First, our analysis allowed us to build a database of all landowners within and adjacent to the LTCA. With this database, we were able to reach out to landowners and host neighborhood events to bring stakeholders together and foster community around a common goal. These conversations allowed us to hear the needs and concerns of local landowners and residents and share data about wildlife movement. Creation of volunteer-driven community stewardship groups centered around protected wildlife corridors created a connection between the land being conserved and the neighbors who live there. In addition, it helped foster crucial relationships among landowners. For instance, volunteers in our Monarch Recovery



**TABLE 3** List of stakeholders invited to participate in urban wildlife corridor conservation.**Stakeholders**


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AFC Property stewardship groups
Boy & Girl scouts
Businesses
Camps—youth
Churches
City planning commissions
City & Town councils
Community farms
Community gardens
Community groups
Community scientists
Conservation clubs
Conservation corps
Conservation organizations
Corporate sponsors
County supervisors
Disadvantaged communities
Docents
Environmental groups
Federal agencies
Funders
Heritage/Historical organizations
Hikers
Indigenous peoples
Landscapers
Land trusts
Native plant nurseries
Neighbors of acquired parcels
Parks & Recreation organizations—county, city
Rock climbers
Rotary clubs
Schools—elementary, middle, high, community colleges, colleges, universities
State agencies
Trail builders
Transportation agencies
Unified school districts
Utilities
Volunteers
Wildlife photographers
Youth education organizations

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Program were not only able to meet each other at milkweed distribution events but more importantly were able to visualize their connection to others and to the overall corridor through our online portal (Figure 4) while at the same time contributing

to research on the success of the corridor. By prioritizing these relationships among stakeholders, we were better able to build trust, a key factor in success of conservation programs (Young et al., 2013).

Second, our urban wildlife corridor framework helped us connect research and data with conservation and policy. Specifically, to enable an evidence-based approach to urban wildlife corridor conservation, we established a collaboration between researchers at an academic institution (Occidental College) with conservation practitioners at a non-profit land trust (AFC). Through this collaboration, we were able to jointly determine research needs within the wildlife corridor, negotiate permissions for land access for wildlife research, and provide recommendations for land acquisitions based on data collected within the LTCA. This collaboration was a stepping-stone to more broadly sharing data across researchers, conservation practitioners, and policymakers across the LTCA. Open access data can serve as a tool to help bridge the gap between science and action, by providing conservation practitioners and policymakers with access to relevant data in near real time (Sullivan et al., 2014). As such, we provide wildlife observation data and landscape connectivity models by request to municipalities, state agencies, and other conservation partners. We also provide municipalities with focused prioritization data for open space parcels within their jurisdiction. Through these collaborations, we were able to overcome disciplinary silos, a key barrier to urban conservation (Kay et al., 2022).

Moreover, designating a priority area for urban wildlife corridor conservation forced us to think across jurisdictional boundaries. Our analysis identified 22 jurisdictions within our priority wildlife corridor area. Wildlife do not necessarily recognize human political boundaries as they move across the landscape (Peters et al., 2018) and as a result may experience inconsistent protection of their corridors (Titley et al., 2021). Across these 22 jurisdictions, we identified variation in habitat protections. For instance, within our proposed corridor one key ordinance that is relevant to conservation of wildlife corridors is the regulation of hillside development. Properties in steep hillsides may be prohibited from development, resulting in conservation by default. However, each jurisdiction defined the hillside areas subject to these regulations differently. The cities of Burbank and Los Angeles use a map to zone hillsides while the County of Los Angeles defines a “Hillside Management Area” as land with a natural slope gradient of 25% or steeper. Some jurisdictions impose additional restrictions designed to protect ridgelines and views, along with protections for specific plant species such as oak trees. As a result, protections may be lost as the corridor passes between neighboring jurisdictions, making a comprehensive assessment essential. By having an urban wildlife corridor framework that assembles and analyzes habitat protections across all jurisdictions simultaneously, we were able



to create and execute on an action plan more knowledgeably and effectively.

## Remaining challenges and recommendations

While an urban wildlife corridor framework can enable conservation within cities, there remain challenges that need to be addressed. Primarily, there is a need for leadership in not only proposing urban wildlife corridors but also in providing and funding the structures necessary for collaboration. At the same time, this leadership needs to be collaboratively driven to maintain equity and prevent unintended power dynamics, such as negative interdependence (Trif et al., 2022), in managing conservation across the corridor. Establishing a collaborative network to lead this effort could serve as a platform for data sharing, peer networking, and more consistent land stewardship across multiple jurisdictions. The nascent collaboration created through our study highlights one path that can be used to achieve meaningful connections for empowering multiple stakeholders in an urban wildlife corridor, but further collaboration and leadership are needed to fully realize the benefits for conservation.

While utilizing an urban wildlife corridor framework has enabled conservation within the Rim of the Valley Corridor, how this approach will apply to other cities and other urban ecosystems remains to be tested. Substantial variation exists across cities in terms of size, density of human populations, the distribution of greenspace or natural areas, as well as the species present, and these differences can have a significant effect on the presence of wildlife (Fidino et al., 2020). In addition, policies and land practices across cities may impact the ability to establish successful wildlife corridors within different urban areas. Our approach relies heavily on conserving land that is undeveloped in order to preserve remaining habitat connectivity; however, in many urban areas, undeveloped vacant parcels may be scarce. Alternative approaches such as land ordinances, incentives, and conservation easements may prove more fruitful in such urban areas.

Additionally, human-wildlife conflicts pose a particularly difficult problem for urban conservation (Dickman, 2010), and wildlife movement through corridors can increase conflicts in some cases (Buchholtz et al., 2020). Human-wildlife conflicts, such as pet depredation by wildlife, occurs more frequently in areas with dense human populations (Poessel et al., 2017). As a result, some people may have negative perceptions of wildlife or even fear some species, which may create stakeholder disagreement. As such, extensive consideration must be placed into the shape and design of urban wildlife corridors to prevent unwanted consequences. For instance, mitigation structures designed to route wildlife to safer road crossings have been shown to significantly reduce wildlife-vehicle collisions (Rytwinski et al., 2016). Additionally, outreach and education

may help to alleviate fears and address misconceptions. For example, increasing connectivity may instead reduce the risk of human-wildlife conflict because wildlife are able to move more freely to access resources, thereby reducing their need to forage or hunt within backyards and neighborhoods. Shifting attitudes about wildlife from conflict to coexistence (Dickman, 2010; Buijs and Jacobs, 2021) will be essential for successful conservation of urban wildlife corridors. More research will be needed on the best approaches for reducing human-wildlife conflict in urban areas.

Similarly, there remains a challenge in building wildlife corridors within urban areas where there are competing needs between wildlife and humans. Building urban wildlife corridors increases greenspace within cities, which has numerous benefits for the human inhabitants within cities. Access to greenspace supports recovery from stress, child development, and other physical health and psychological benefits (Kowarik, 2011; Scott et al., 2018), and greenspace can mitigate impacts of climate change and urban heat islands (Park et al., 2017). However, wildlife may be less likely to use corridors if there is too much human activity within these spaces (Bond, 2003). As such, a careful balance needs to be struck. There remains a lack of research on the ability of infrastructure and management to alleviate the impacts of human activity on wildlife (Sweeny and LaClair, 2000). Thus, future research is needed to determine how best to design urban corridors to allow for human access to these spaces while simultaneously minimizing the impact of human activity on wildlife movement.

Furthermore, it is important to consider issues of environmental justice when establishing urban wildlife corridors. Access to greenspace and nature within cities is not equitably distributed, with low-income communities on average being located farther from parks and natural spaces within the city (Williams et al., 2020). In fact, in many areas, city inhabitants are located more than a 20-min walk from the nearest park (Williams et al., 2020). These inequities are also linked to systemic racism and historic practices such as redlining, which have continued consequences for the distribution of biodiversity and wildlife within cities (Schell et al., 2020; Vasquez and Wood, 2022). Urban wildlife corridors have the potential to transform equity in access to nature by building habitat connectivity through areas impoverished of nature. Yet because wealth has been associated with biodiversity conservation (Leong et al., 2018), there remains a risk that urban wildlife corridors will be inequitably conserved across cities. As such, care must be taken to assure that wildlife corridors are established in an equitable manner within cities.

At the same time, building of wildlife corridors in areas impoverished of nature may have the unintended consequence of eco-gentrification. While preserving and restoring habitat within an urban area has many benefits for both wildlife and human health, it may also have an unintended impact of displacement of residents as property values rise due to environmental remediation and investment (Wolch et al.,

2014; Rice et al., 2020). As a result, care must be taken to consider placement of greenspaces so that they can improve the local environment without displacing low-income communities. Calling attention to and organizing community discussions around the issue of eco-gentrification is the first step in addressing this potential threat to the success of urban wildlife corridors (Mayayo, 2019).

Finally, funding conservation of wildlife corridors in urban areas is particularly challenging. Since urban wildlife corridors require preservation and acquisition of stepping-stones through the urban environment, many small land parcels may need to be restored or purchased. However, these smaller land parcels are also more difficult and expensive to fund. First, it is much easier to describe to funders how hundreds of acres of open space will function ecologically, than to help them visualize how a one-acre parcel is essential in a series of yet-to-be-acquired habitat fragments. Second, the acquisition cost per acre is higher in urban areas than more rural areas (Nolte, 2020). Third, acquisition, restoration, and management of multiple land parcels is more expensive and time consuming than a single large land parcel. Creative approaches to funding conservation within urban areas, such as incentives (Ring et al., 1998) or Conservation Subdivisions (Carter, 2009), will need to be developed.

## Summary

Conservation within urban areas is essential for many wildlife species and for improving equity in access to nature, but many challenges exist. Our approach provides one example of how urban wildlife corridor conservation can be achieved through research, land acquisitions, collaboration, restoration and stewardship, and outreach and education. We demonstrate that urban wildlife corridors can provide a framework for conservation in cities that helps to overcome some of the challenges to urban conservation. By explicitly weaving together natural and urban spaces, urban wildlife corridors bridge important gaps between researchers and practitioners, numerous stakeholders, neighboring jurisdictions, complementary datasets, as well as between humans and wildlife. Ultimately, utilizing a collaborative urban wildlife corridor framework for conservation in cities can increase the efficiency of conservation efforts, help redesign cities to be more wildlife-inclusive, and build crucial connections among stakeholders to enable further action.

## 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/s.

## Author contributions

All authors contributed to all aspects of the work, from the design and conception of the ideas to the writing and editing the manuscript. All authors contributed to the article and approved the submitted version.

## Funding

Funding for camera trap research was provided by Disney Conservation Fund and Occidental College. Additional project funders include the Cygnet Family Foundation, Michael J. Connell Foundation and Southern California Edison. The Monarch Recovery Project was supported in part by the Santa Monica Mountains Conservancy (3810-P68-2142) and Pasadena Community Foundation. Open access publication was funded by Occidental College.

## Acknowledgments

We thank the supporters and staff of AFC, the Occidental-AFC camera team, Mark Duttweiler, Roshni Katrak-Adefowora, Maggie Swomley, and Auxenia Privett-Mendoza for their leadership in studying urban wildlife corridors in Los Angeles. We thank the United States Geological Survey for camera trap supplies. We thank the Urban Wildlife Information Network and the Lincoln Park Zoo Urban Wildlife Institute for database support and methodological design review. We thank Scott Harris, Kat Superfisky, and Sophie Parker for insightful discussions that inspired this work.

## Conflict of interest

Author BG was employed by Arroyos & Foothills Conservancy.

The remaining author declares that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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## OPEN ACCESS

## EDITED BY

Sophie S. Parker,  
The Nature Conservancy, United States

## REVIEWED BY

Manoj Kumar Jhariya,  
Sant Gahira Guru  
Vishwavidyalaya, India  
Samraj Sahay,  
University of Delhi, India

## \*CORRESPONDENCE

Erica L. Wohldmann  
erica.wohldmann@csun.edu

## SPECIALTY SECTION

This article was submitted to  
Urban Greening,  
a section of the journal  
Frontiers in Sustainable Cities

RECEIVED 11 May 2022

ACCEPTED 05 October 2022

PUBLISHED 26 October 2022

## CITATION

Wohldmann EL, Chen Y, Schwarz K,  
Day SD, Pouyat RV, Barton M and  
Gonez M (2022) Building soil by  
building community: How can an  
interdisciplinary approach better  
support community needs and urban  
resilience?  
*Front. Sustain. Cities* 4:941635.  
doi: 10.3389/frsc.2022.941635

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# Building soil by building community: How can an interdisciplinary approach better support community needs and urban resilience?

Erica L. Wohldmann<sup>1\*</sup>, Yujuan Chen<sup>2,3</sup>, Kirsten Schwarz<sup>4,5</sup>,  
Susan D. Day<sup>6</sup>, Richard V. Pouyat<sup>7</sup>, Michelle Barton<sup>8</sup> and  
Manny Gonez<sup>3</sup>

<sup>1</sup>Department of Psychology, California State University, Northridge, Northridge, CA, United States,

<sup>2</sup>Department of Agricultural and Environmental Sciences, Tennessee State University, Nashville, TN, United States, <sup>3</sup>Department of Policy and Research, TreePeople, Beverly Hills, CA, United States,

<sup>4</sup>Department of Urban Planning, University of California, Los Angeles, Los Angeles, CA, United States,

<sup>5</sup>Department of Environmental Health Sciences, University of California, Los Angeles, Los Angeles, CA, United States, <sup>6</sup>Department of Forest Resources Management, University of British Columbia,

Vancouver, BC, Canada, <sup>7</sup>Scientist Emeritus, United States Department of Agriculture Forest Service, NRS-08, Newark, DE, United States, <sup>8</sup>LA Sanitation and Environment, Los Angeles, CA, United States

Given the interrelated problems of climate change, energy and resource scarcity, and the challenge of supporting critical natural systems in cities, urban dwellers may be exceptionally vulnerable to the impacts of climate change. While a number of programs and policies have been developed and implemented to help reduce the environmental and social impacts of climate change on communities, we argue that effective and sustainable programs must not only consider how the changing environment impacts communities, but also how communities interact with and impact the environment. Specifically, drawing on a case study of the needs assessment of the Healthy Soils for Healthy Communities Initiative conducted in Los Angeles (LA) County, CA as a model for a Virtuous Cycle Framework, we attempted to better understand how urban residents interact with land, green spaces, and soil as a means of finding ways to address some of the environmental and health disparities that many urban residents experience, while also exploring ways to improve soil health to support its capacity to provide essential ecosystem services (e.g., carbon sequestration, water filtration, food and biomass production). A unique feature of our approach is that it involved an interdisciplinary and multi-level partnership composed of a well-established environmental organization dedicated to urban forestry, environmental justice, and climate resilience, university faculty researchers who study human behavior and human-nature relationships, government partners, and, most importantly, community members, among others. The first step in understanding how community members interact with their environment involved collecting survey and focus group data from residents of LA County to assess attitudes, beliefs, and behaviors around land and soil. Results were used to explore strategies for deepening community engagement, addressing

knowledge gaps, and shaping policies that would benefit not just people who live/work in LA, but also the soil and other natural systems that rely on soil. This article integrates our previously published survey and focus group findings with new results that pertain specifically to the Virtuous Cycle Framework, and demonstrates how the data are being used to inform our community-based interventions (e.g., policy change, public education and community engagement, and demonstration projects).

#### KEYWORDS

soil education, community science, community engagement, soil science, urban soil management, climate resilience

## Introduction

The Virtuous Cycle was first proposed by Morrison (2015) as a socioecological systems framework for conservation. The framework is centered around an intervention, grounded in a specific place, that aims to improve conservation outcomes. The intervention has beneficial outcomes for nature, and when those benefits are recognized by people, communities reinforce positive conservation outcomes through policies and actions that promote sustained positive change. Thus, the Virtuous Cycle Framework envisions a positive feed-forward loop.

However, we know that community-based environmental interventions, including policies, can have unintended consequences, or are not universally beneficial, recognized, or even desired by communities and, therefore, fail to impact human behavior. For example, if an intervention or policy enacted is not supported by the community in which it is embedded, it is unlikely that the environmental benefits will be fully realized or reinforced by the community, weakening the positive feedback loops that contribute to long-term environmental change. Going further, while community support is essential for long-term change, we believe interventions that are proposed, planned, implemented, and evaluated by the communities in which they are situated—and are supported by community leaders and/or backed by policy—produce even stronger reinforcing positive feedback loops among people and nature, resulting in highly resilient and sustainable socio-ecological systems. In this paper we argue that interventions that are driven by community goals and values not only strengthen the Virtuous Cycle, but also increase the likelihood of being accepted and adopted. We draw on a case study of the needs assessment of the Healthy Soils for Healthy Communities Initiative as a model for a Virtuous Cycle Framework to explore how to protect people and the planet through better soil management practices in Los Angeles (LA) County, California, which is the most populous county in the United States (Chen et al., 2021; Schwarz et al., 2022).

Urban populations continue to increase with well over half of the global population living in urban areas. In the

United States, 86% of people lived in urban metro areas in 2020, and LA County currently houses over 10 million people (United States Census Bureau, 2021). Although urbanization, if planned strategically and managed properly, has the potential to reduce poverty and inequality by providing more opportunities for employment, education, and better access to medical facilities, important environmental and social challenges must be addressed as urban populations increase. For example, large urban agglomerations pose significant challenges for natural resources, and can lead to a degradation of the environment and, consequently, human health and well-being (e.g., Khan et al., 2021). Thus, there is an urgent need to protect and enhance environmental quality in highly urbanized settings in order to secure an improved quality of life for the increasing numbers of urban dwellers. Urban systems are socio-ecological in nature, however, and understanding human perspectives on conservation interventions and the resulting outcomes is essential.

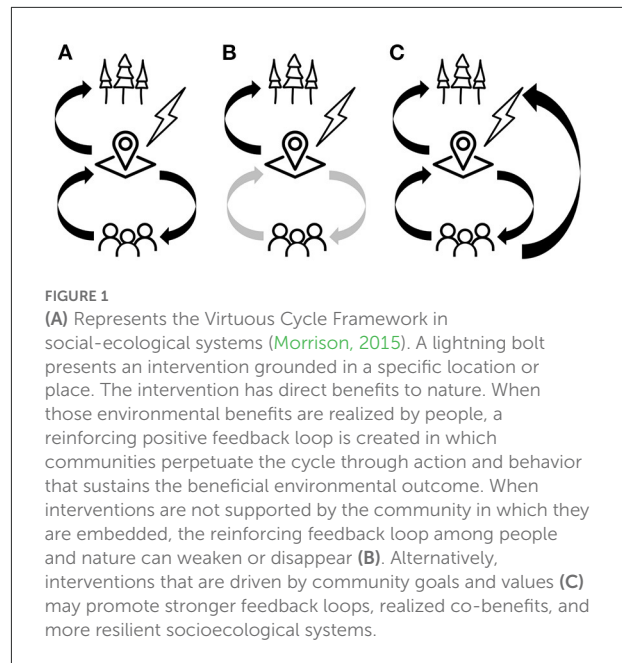
Although often overlooked, the impacts of urbanization on soil contributes to many environmental and social challenges. For instance, the conversion of land from primarily agricultural and forest uses to urbanized landscapes significantly modifies soils via scraping, redistribution, compaction, and management (Pouyat et al., 2020), especially with respect to their ability to store carbon and mitigate the release of greenhouse gasses (Trammell et al., 2018). Urban soils also receive inputs such as irrigation, fertilizers, pesticides, and construction debris that can alter function. In urban environments, the proportion of soil covered by impervious surfaces can be very high, resulting in elevated air temperatures compared to the surrounding rural landscape, an effect commonly known as the urban heat island effect (Arnfield, 2003). Additionally, soils covered by impervious surfaces are “sealed” from water and gas exchanges and lack organic matter inputs (Pouyat et al., 2020).

Changes in climate, including higher temperature and more intense precipitation events, may further impact urban soils and their ability to provide important ecosystem services including climate regulation, stormwater management and filtration, provisioning of habitat for various organisms, and social services

such as pollution mitigation and food security. Soils perform these functions directly, but also support vegetation, including urban forests and food crops. In turn, plant types can drive changes in urban soil characteristics such as nitrogen and carbon accumulation (Setälä et al., 2016; Kotze et al., 2021).

The ecosystem services of soils are vital for reducing the vulnerability of densely populated areas to natural disasters, as well as for improving the health and quality of life of urban residents, especially those living in disadvantaged communities, as they are usually the most vulnerable to climate impacts and frequently have limited adaptive capacity to climate change. For example, those living with poverty or in marginalized communities may be at higher risk for urban heat exposure (Voelkel et al., 2018). In addition, ozone exposure has also been found to be significantly higher in community parks located in disadvantaged communities (with majority Latino use) compared to affluent community parks (Winter et al., 2019), while urban tree canopy, which could mitigate these effects, is positively related to household income (Schwarz et al., 2015). Additionally, many studies have shown that high concentrations of heavy metals are more often found in soils located in low-income areas (e.g., Montañón-López and Biswas, 2021). Moreover, because urban areas are major contributors to air pollution, urban residents, especially vulnerable populations, are often exposed to unhealthy air, an effect that is amplified by the micro-climatological effects of buildings and other infrastructure and the associated decrease in vegetation due to limited soil resources (Lane et al., 2022). Thus, understanding ways to address these disparities and increase resilience in impacted neighborhoods is key. In this article, we argue that public engagement strengthens the Virtuous Cycle Framework and, thus, is an important and necessary part of the equation.

Guided by the Virtuous Cycle Framework, in 2020, through a partnership composed of NGOs, universities, governmental agencies, and community groups, TreePeople, launched the “Healthy Soils for Healthy Communities” initiative (Chen et al., 2021). Our team defined healthy soils as the capacity of soil to function as a living ecosystem that offers a range of services that support and sustain life, and is the foundation for healthy environments that foster robust socio-ecological systems. One of the main goals of this initiative was to conduct a needs assessment of LA County’s soils using online surveys, focus groups, and an in-depth understanding of the literature (Chen et al., 2021; Schwarz et al., 2022). Through this assessment, we learned that LA County residents value green space, and actively maintain their green spaces. In addition, LA County residents are accustomed to composting, and the majority regularly use the “green bin” (i.e., the curbside residential yard waste bin) for their green waste, or they allow green waste to compost in some form on their property. However, despite the fact that interest in gardening and composting is high, knowledge about factors that affect soil health was generally low. Furthermore, although most LA residents expressed concern about soil contamination



and pollution, very few had ever tested their soils, and those who had, only tested for nutrient deficiencies, not heavy metals or pollutants (Schwarz et al., 2022).

Another important outcome of the needs assessment related to the process of working with an interdisciplinary and multi-level team. Engaging the community and associated stakeholders in regard to healthy soils was identified as a key to achieving our goals of creating an effective and sustainable strategy for changing attitudes, behavior, and social norms through public engagement. In fact, one primary purpose of the Healthy Soils for Healthy Communities initiative was to create an intentional and meaningful interaction between scientists (i.e., active researchers), the public (i.e., people who operate primarily outside of the practice of science, including the “general public” and highly specialized publics, such as policy makers, business leaders, community leaders, and others with extensive expertise in non-science domains), and practitioners (i.e., those with expertise in soil and soil-related education) to provide opportunities for mutual learning. The process involves raising awareness, providing education, and enabling the community to both advocate for and work toward building healthy soils in the region, which are all important aspects of the positive feedback loops in the Virtuous Cycle Framework (see Figure 1).

Without interventions that are informed and supported by impacted communities, a healthy soils initiative is far less likely to elicit the reinforcing positive feedback loops represented in the Virtuous Cycle Framework. In fact, several lines of research have demonstrated that the deficit model of communication, which presumes the public lacks knowledge, and that scientists need to supply that knowledge, is ineffective (Besley et al.,

TABLE 1 Use of public green spaces (percentage of respondents as a function of home ownership status among residents).

How often do you or members of your household use a public green space...?	Never	Rarely	Sometimes	Frequently	Daily
Home owners	6.8%	18.9%	28.7%	36.0%	9.6%
Renters	5.3%	11.7%	26.4%	44.9%	11.7%

2013). Despite this, “informing the public” and/or “defending science from misinformation” continue to be at the top of scientists’ most prioritized communication goals, predictors of valuing outreach, and desires for communication training (Besley et al., 2013; Besley, 2015; Dudo and Besley, 2016). The perceived importance of “informing publics” is so ingrained that its prioritization is generally unaffected by other attitudinal, behavioral, or demographic factors (Dudo and Besley, 2016). Community engagement, often proposed as an alternative to the deficit model, presents an opportunity to identify key interventions and feedback that are likely to sustain a Virtuous Cycle, according to the described framework.

In our project, key interventions were identified by the community, and based on these interventions, our team developed three demonstration projects. The first project was aimed at increasing tree canopy cover and enhancing stormwater mitigation through soil best management practices. The second project involved community-based soil sampling to better understand soil contamination and pollution in disadvantaged communities. The third project established an urban carbon farm to explore the carbon sequestration potential of soils in LA.

Here, we present how the process, results, and interventions contribute to the Virtuous Cycle Framework, as well as the potential that our process has to be applied in different regions in order to achieve more climate-resilient urban communities. We also document some of the lessons gleaned from the process of working with an interdisciplinary and multi-level stakeholder model for community engagement and discuss considerations for moving forward, including outcomes of this project that are currently in process and recommendations for future interventions.

## Materials and methods

### Study area

LA County, CA, United States covers 4,058 square miles (10,510 sq km), and has a population of approximately 10.04 million people (United States Census Bureau, 2021), making it the most populous county in the nation. County-wide, the average tree canopy cover is 18% (LA County Tree Canopy Advanced Viewer; <https://www.treepeople.org/los-angeles-county-tree-canopy-map-viewer/>), although tree canopy is greater in wealthy neighborhoods, such as Beverly

Hills (35%), than in less wealthy neighborhoods, such as Irwindale (6%) and Compton (11%). Because of the size of LA County, it was divided into eight geographic regions using the California County Department of Public Health service areas (Schwarz et al., 2022).

### Online surveys

We disseminated four separate online surveys (in both English and Spanish) to residents, educators, policymakers, and soils-related professionals across LA County (Chen et al., 2021; Schwarz et al., 2022). However, this article focuses on new analyses conducted using variables not previously analyzed or reported. Specifically, an ANOVA was conducted to determine whether home ownership predicted use of public green spaces. In addition, a correlational analysis was conducted to examine the relationship between frequency of self-reported use of public green spaces and concern about the soil quality in those spaces. Further, correlational analyses were conducted to examine the factors that are associated with the likelihood of soil testing. These analyses center around the Virtuous Cycle Framework and, most critically, how community feedback about soil-related concerns fed into the interventions we developed through the demonstration projects.

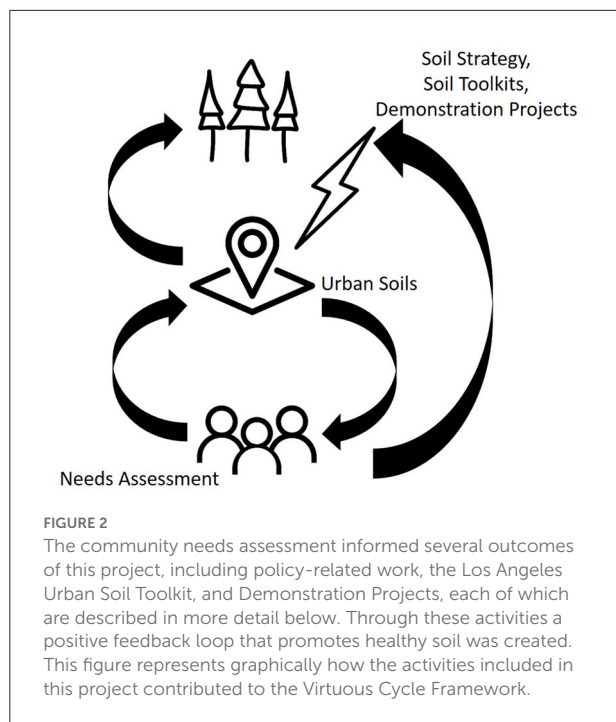
## Results

We found that residents who rent their homes are significantly more likely to use public green spaces than those who own,  $F_{(2,1037)} = 5.97$ ,  $p = 0.003$  (see Table 1).

Further, there was a significant positive correlation between use of public green spaces and concern for the soil in those green spaces,  $r(1040) = 0.17$ ,  $p < 0.001$ , with residents who use public green spaces being more concerned about the soil quality than those who do not.

One of the key strategies for improving soil health in urban areas involves soil testing. However, Schwarz et al. (2022) described two findings that we believed might be correlated. Specifically, Schwarz et al. (2022) found that very few residents had conducted soil testing. In addition, the authors found that concern for soil contamination and pollution was high. In the present study, we conducted a correlational analysis to test whether concern about soil contamination and pollution was associated with soil testing, but did not find a significant





relationship. That is, concern about soil contamination and pollution was not correlated with soil testing. What was correlated with the likelihood of soil testing, however, was knowledge about factors that contribute to soil health including soil pH, bulk density, permeability, chemistry, and biodiversity,  $r(1042) = 0.26, p < 0.001$ . In addition, knowledge about how to compost was significantly correlated with soil testing,  $r(1042) = 0.24, p < 0.001$ . Thus, the more knowledgeable residents reported being about these topics, the more likely they were to conduct soil testing.

## Discussion

There are a number of ways that results from this needs assessment can be used to inform and impact policies and/or practices to improve community resilience, public education and community engagement, as well as to demonstrate potential solutions that address the identified needs, all of which contribute to the Virtuous Cycle Framework (see Figure 2). Here, we describe some of the specific outcomes that this project has already produced, including the Healthy Soils Strategy for the City of Los Angeles and the Soil Toolkit. In addition, we discuss how the results were used to create demonstration projects, which included opportunities to interact with and learn about healthy soil, and to actively contribute to potential solutions (e.g., through community-based soil sampling and testing). We would like to note that while our needs assessment was the first of its kind for urban soils, other examples of urban soil programs do exist in other cities, most notably the Urban

Soil Initiative (USI) in New York City (<https://urbansoils.org>). In fact, a partnership between the LA Healthy Soils Initiative and the USI to create and hold joint workshops and other community programming was another outcome of this study.

## Policy change: LA City healthy soil strategy

The Healthy Soils for Healthy Communities Initiative has informed and guided the City of LA's work on healthy soils. In 2021, the City of Los Angeles published the Healthy Soils Strategy for the City of Los Angeles (<https://lacitysan.org/san/sandocview?docname=cnt067543>). This strategy document was prepared by the LA Sanitation and Environment (LASAN) Healthy Soils Team and the Healthy Soils Advisory Panel (HSAP) composed of academics, researchers, local nonprofits, and experts in soil health. The HSAP provided significant guidance on the effort and contributed extensively to the strategy document, ensuring that the document was comprehensive and had buy-in from experts and relevant community representatives. The document details relevant urban soil topics and provides strategies and supporting actions that LASAN, City departments, community groups, and residents can take to conserve and properly manage healthy soils. Each chapter of this strategy document has a specific focus, for example, about the ecosystem services that soil provides, the importance of composting, ways to test for and report contamination and pollution, and opportunities to benefit from and learn more about soil. Each chapter also includes strategies and supporting actions that can be taken to achieve healthy soils goals. The variety of actions proposed encourage involvement at all levels within the community. Some of the actions represent interventions in soil health, for example, incorporating compost into compacted soils. Other actions, such as facilitating community-based soil testing, are interventions that help explicitly identify who might benefit from a specific intervention and thus strengthen information feedback to the community. Several project team members serve on the HSAP. In this way, we can ensure that future policy work is guided by community needs and research and, in turn, increase the adoption and implementation of new strategies by community members. In this way, this project contributes to the positive feedback loop in the Virtuous Cycle Framework.

## Public education and community engagement: Los Angeles urban soil toolkits

As part of the needs assessment, we developed the Los Angeles Urban Soil Toolkit (in English and Spanish), which

was meant to serve as a beginner's guide to improving and sustaining the health of LA's urban soil. The objective was to incorporate what was learned from our community engagement efforts and, thereby, transfer science into practice, as well as to provide technical support for communities to actualize what they expressed wanting. The toolkit was intended to be a useful source for information and resources about soil health, including how soils impacts our environment. It is currently being used as an educational resource for TreePeople's public education and community engagement activities, and there are plans to develop two additional soil toolkits that can be used by educators and community leaders, which the needs assessment suggested were the groups that expressed the strongest interest in learning more about soils. We anticipate that education about soils will not only promote soil testing, but will also increase community awareness of the feedback loops stemming from soil interventions. These toolkits can be targeted to help residents recognize when they are seeing the effects of the soil interventions, which serves to elicit a positive feedback loop that contributes to the Virtuous Cycle. A similar project not related to our initiative is one involving the Windy City Harvest Model (Chicago Botanic Garden, 2021), which used a hands-on tool kit for public gardens to connect people to soil and plants (<https://www.usbg.gov/urbanagriculturetoolkit>). Their toolkit provides an array of information from building effective partnerships, to farm design and operations, to fundraising.

## Demonstration projects

To address the identified needs, we developed an overall framework for the continuation of this initiative, which proposes to establish an overall strategy for a Los Angeles Urban Soil Collaborative. The strategy will be developed through community, government, NGOs, academia, and private sector participation. One of the demonstration projects involved community-based soil sampling, which aimed to deliver a powerful tool to help communities, researchers, and policymakers chart the potential for soil restoration or improvement. Soil sampling within and by the community is the first step in generating neighborhood-specific information on the spatial distribution of soil-related hazards, and optimizing remediation efforts through the targeted use of best management practices. This intervention helps explicitly define who will benefit from soil management interventions, a critical characteristic of the Virtuous Cycle. It can also support more climate-resilient futures by empowering communities who have suffered a disproportionate burden of toxic exposure with the tools and information necessary to promote healthier urban ecosystems.

## Closing the loop: Recommendations for future interventions

One of the key strategies for improving soil health in urban areas involves soil testing. However, our survey results suggested that only a proportion of LA residents have tested their soil. As stated previously, the most common testing examined only pH and NPK. However, it is well-known that harmful levels of lead and arsenic can be found across parts of Los Angeles due to the operations of Exide Technologies, a former battery recycling plant in Vernon, CA that was responsible for widespread soil contamination. While respondents were keenly aware of the potential for harmful levels of contaminants, they found it challenging to identify ways in which they might engage in this part of the cycle. In fact, the survey results suggested residents were very concerned about soil contamination, and the results of the focus groups aligned with the survey results. That is, community members expressed a high level of concern about the potential that their neighborhoods may have been contaminated by heavy metals (Schwarz et al., 2022). Additionally, based on the results from focus groups, there was a strong desire and consensus around future work needing to effectively engage and center communities, working to build trust and address past harm. Without the community engagement efforts and, more specifically, the online surveys and focus groups, we would not have identified the community's desire for access to soil testing that was not controlled by either the private sector or government—institutions in which the community lacks trust.

One way for cities like LA to establish trust and encourage residential soil testing could be to follow the lead of, for example, the New York City Urban Soils Institute. More specifically, this organization offers individual soil testing packages for home and community gardeners, as well as more specialized soil and site assessments from their own soil specialists or member academic soil scientists (<https://urbansoils.org/soil-assistance>). They offer low-cost tests that measure nutrients, physical properties, and trace, as well as free consultation for collecting and processing samples.

Other ways to improve soil health include education about factors that influence soil health, and strategies for improving soil health (e.g., composting). Schwarz et al. (2022) found that only about 8% LA residents reported being highly knowledgeable about factors that influence soil health (e.g., soil pH, permeability, composition), and only 15% reported being highly knowledgeable about composting. New analyses reported here suggest knowledge on these topics may be an important predictor of soil testing, as these factors were significantly correlated. Specifically, we found that people who are highly knowledgeable about one or both of these topics are also more likely to have their soil tested than those who are not

knowledgeable. Future research could explore this relationship more systematically.

Taken together these results suggest public education and engagement around soil health and composting could be one way to increase soil health. Further, residents' concern for soil contamination in public green spaces could be addressed by (1) conducting widespread soil testing with community involvement, and (2) posting signage or providing other similar forms of indirect education that identify the links between interventions and wellbeing.

While a sustained healthy soil engagement and education effort that raises awareness is necessary to achieve long-lasting change, several key actions must be taken to gain community and stakeholder support—engage residents directly in the virtuous cycle—and encourage behavior and mindset change. First, developing a public-private partnership is essential. This project was successful because of its focus on maintaining a true collaboration between nonprofit organizations and the City of Los Angeles. Specifically, TreePeople, LA Compost, and Kiss the Ground—all LA-based non-profits, worked together to inform the original research questions, distribute the online surveys and recruit participants for the focus groups. In addition, they have continued to collaborate on demonstration projects, for example, developing an urban carbon farm situated at a public park that is being used for additional research, public education, community engagement, and as the site for one of the demonstration projects. Furthermore, this collaboration was composed of a multi-disciplinary team including nonprofits, scientists, and government agencies. TreePeople conducted the needs assessment report as part of the community engagement process while building strong partnerships with soil scientists. This diverse effort allowed for a more effective and well-rounded project and deepened the community connections that are essential to create and sustain a Virtuous Cycle.

## Conclusion

A comprehensive community engagement approach allows interventions in urban natural systems to support a Virtuous Cycle that increases the likelihood of long-term sustainable improvements in environmental outcomes. Such a community-based approach with an interdisciplinary team can help identify key areas where initiatives can be tailored to support either feedback or interventions that will strengthen the Virtuous Cycle. Without this information, feedback information may not be directed to appropriate sectors of the community or may be misinterpreted, weakening the sustainability of such initiatives. Many urban residents are uniquely vulnerable to climate-related health and other impacts. Targeted interventions based on the Virtuous Cycle

Framework may increase the likelihood of success in mitigating these impacts.

## 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 UCLA and CSUN Institutional Review Board. The patients/participants provided their written informed consent to participate in this study.

## Author contributions

EW, YC, KS, RP, and SD designed the study. EW conducted analyses and interpretation of the online survey. EW, YC, KS, and SD wrote the manuscript with substantial contributions from RP, MG, and MB. All authors contributed to the article and approved the submitted version.

## Funding

Funding for the research was provided by Accelerate Resilience LA, a sponsored project of Rockefeller Philanthropy Advisors.

## Acknowledgments

The authors would like to thank the survey and focus group participants for their contribution and the Healthy Soils for Healthy Communities Steering Committee and Project Team for their collaboration.

## Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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## OPEN ACCESS

## EDITED BY

Zoltan Elek,  
Hungarian Academy of Sciences (MTA),  
Hungary

## REVIEWED BY

Jana Růžicková,  
ELKH-ELTE-MTM Integrative Ecology  
Research Group, Hungary  
Zsolt Végvári,  
Hungarian Academy of Science,  
Hungary

## \*CORRESPONDENCE

Eric M. Wood  
ericmwood@calstatela.edu

## SPECIALTY SECTION

This article was submitted to  
Urban Ecology,  
a section of the journal  
Frontiers in Ecology and Evolution

RECEIVED 31 May 2022

ACCEPTED 06 October 2022

PUBLISHED 28 October 2022

## CITATION

Vasquez AV and Wood EM (2022) Urban  
parks are a refuge for birds in park-poor  
areas.  
*Front. Ecol. Evol.* 10:958572.  
doi: 10.3389/fevo.2022.958572

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# Urban parks are a refuge for birds in park-poor areas

Amy V. Vasquez<sup>1,2</sup> and Eric M. Wood<sup>1\*</sup>

<sup>1</sup>Department of Biological Sciences, California State University Los Angeles, Los Angeles, CA, United States, <sup>2</sup>Department of Earth and Planetary Sciences, Johns Hopkins University, Baltimore, MD, United States

Urban parks provide amenities that support both human and animal communities. However, parks are often unevenly distributed within cities. One metric used to assess the distribution of parks to the public is termed the Park Score. The Park Score is an approach to measure access, acreage, investment, and amenities, and is designed to understand a city's needs for greenspace, with a major focus on public health. In addition to issues related to public health, a disparity in the distribution of urban parks may pose a barrier for wildlife, such as birds. Yet, this remains unclear. We designed a study to quantify the role of parks in providing a refuge for birds across a park-needs gradient in Greater Los Angeles (LA), a metropolis with one of the lowest park scores in the United States. We had two objectives to address our goal. First, we quantified patterns in habitat features and avian communities within and adjacent to parks. Second, we analyzed relationships among habitat features within and adjacent to parks on avian abundance. We sampled birds and habitat features at 48 parks across a park-needs gradient in L.A. from October to March of 2017/2018 and 2018/2019. We found three lines of evidence supporting the refugia effect of parks. First, habitat features within parks were similar between low- and high-needs areas of LA, and this likely influenced avian abundance patterns, which were also alike. Second, avian communities were generally similar across the park-needs gradient, where parks in high-needs areas harbored birds affiliated with forest and shrub ecosystems. Third, bird abundance patterns were related to numerous habitat features within parks, regardless of where parks occurred in the city. The patterns we uncovered were opposite to what is found in residential areas (i.e., luxury effect), suggesting that parks provide important habitat for birds, whether in high- or low-needs sections of LA. Our results stress the role of parks as refugia in park-poor areas because they provide habitat in otherwise inhospitable urban conditions. Continued investment in park development in high-needs areas can thus potentially be a win-win when considering the benefits to people and birds.

## KEYWORDS

avifauna, biodiversity, LiDAR, park score, remote sensing, socioeconomic, luxury effect

## Introduction

Urban ecosystems are densely populated, human-dominated environments embedded within a mosaic of natural and anthropogenically modified landscapes (Cadenasso and Pickett, 2008; Grimm et al., 2008). Cities and other urban environments are the primary living areas of humans, containing approximately 55% of the world's population (United Nations, 2018). In addition to providing conditions amiable to people, urban ecosystems also support varying levels of biodiversity (Aronson et al., 2014; Lepczyk et al., 2017). For example, cities tend to harbor an unusually high diversity of plants and a lower, more homogenous diversity of wildlife (Helden and Leather, 2004; Alvey, 2006; Colding and Folke, 2009; Beninde et al., 2015; Talal and Santelmann, 2019). As cities continue to develop and sprawl to accommodate increasingly dense human populations, there is a growing concern about the degradation of green spaces within the urban landscape (Vallejo et al., 2009; Wu, 2010; Xu et al., 2018). Given the loss and fragmentation of habitat, and the increase in land-cover change across the globe, biodiversity must either adapt or risk extirpation in the face of urbanization (McKinney, 2002; Seress and Liker, 2015; La Sorte et al., 2018; Young et al., 2019).

Urban planners often intentionally, or unintentionally, design and include features that benefit biodiversity and the environment. One such feature that is prominent in urban areas is urban parks. The collection of managed amenities in parks, such as trees, shrubs, and lawn cover, is often positively correlated with wildlife (Hermy and Cornelis, 2000; Khera et al., 2009; Nielsen et al., 2014). Nevertheless, parks are usually unevenly distributed across cityscapes. One metric that cities in the U.S. use to assess the value of their parks to the public is the Park Score (Trust for Public Land, 2021). The Park Score measures access, acreage, investment, and amenities, and is designed to understand a city's needs for greenspace, with a major focus on public health (Trust for Public Land, 2021). Cities with high park scores often have parks distributed equitably across the urban environment, which carries over to benefit the human population. On the other hand, cities with low park scores face the opposite patterns, with large swaths of a metropolis being park-poor, often in lower-income residential communities (Trust for Public Land, 2021). The negative effects of low park scores are correlated with a host of public health issues in low-income communities ranging from higher rates of diabetes and obesity to increased crime and lack of access to nature (Lovasi et al., 2013; Han et al., 2018). Further, given the disparity in habitat conditions across socioeconomic gradients in urban areas (e.g., Wood and Esaian, 2020), cities with low park scores likely also face considerable challenges in providing habitat for wildlife throughout their boundaries.

The 'luxury effect' is a socio-ecological hypothesis that states that the amount and diversity of vegetation and wildlife in the urban environment follow general wealth patterns (Hope et al., 2003; Leong et al., 2018; Schell et al., 2020; Magle et al., 2021). Evidence of the luxury effect has been found in many cities (Luck et al., 2009; Clarke et al., 2013; Jenerette et al., 2013; Avolio et al.,

2015; Schwarz et al., 2015) and in a variety of green spaces across the urban landscape, such as community gardens (Clarke and Jenerette, 2015) and residential areas (Wang et al., 2015). The luxury effect has similarly been shown to predict patterns of wildlife diversity in cities, with low-income areas being less biodiverse than wealthier counterparts (Kinzig et al., 2005; Strohbach et al., 2009; Lerman and Warren, 2011; Davis et al., 2012). While the luxury effect is not present in every city, often as a result of distinct development and social histories (Kendal et al., 2012; Chamberlain et al., 2019), the phenomenon is typically linked with the segregation of greenspaces (e.g., Venter et al., 2020), which is characterized by the park-score metric (Trust for Public Land, 2021). While parks are public features of cities, concerted efforts in investment are required at the city and community levels to develop and maintain parks. Thus, the luxury effect may also explain patterns of urban park biodiversity. However, this remains untested.

The overarching goal of our study was to understand the role of parks in providing a refuge for birds throughout Greater Los Angeles, California (LA) across a park-needs gradient. LA has one of the lowest park scores of the major cities in the United States (Trust for Public Land, 2021). With a clear understanding of the hurdles this poses to its population, the city (and region) has been investing heavily to meet this challenge (City of Los Angeles Department of Recreation and Parks, 2019). Nevertheless, there remains a lack of information on whether parks provide suitable habitat for birds in LA, especially when considering the variation in parks across the cityscape (Trust for Public Land, 2021). Given that avifauna varies strongly in residential areas across an income gradient throughout LA (Wood and Esaian, 2020), we sought to examine whether parks can buffer the negative effects of urbanization in areas of the metropolis that comparably lack green space. Thus, we designed a study set along a park-needs gradient to understand how variations in park features and urban habitat surrounding parks influence their avifauna. To address our overarching goal, we had two objectives.

First, we analyzed variations in habitat features and avian communities in parks across a park-needs gradient. We predicted a refugia effect of parks, following from refugia effects in the conservation literature (Rojas et al., 2022), where parks with higher stressors surrounding their boundaries would have a higher abundance of birds and a distinct avifaunal community than those with low stresses. In our system, we assumed that neighboring stresses of parks were related to the amount and extent of urbanization, e.g., high impervious surface cover. Since high-park needs areas of LA are generally situated in low-income areas that tend to be less vegetated (Avolio et al., 2018), we expected that parks in these areas would be more beneficial for birds than parks in low-needs areas (high income), where birds may utilize the largely vegetated residential areas (Wood and Esaian, 2020). Further, we predicted that parks surrounded by higher impervious surface cover would harbor a greater abundance of synanthropic species than birds that typically reside in natural

areas (Aronson et al., 2016), a pattern found in protected areas across the United States (Wood et al., 2014, 2015).

Second, we analyzed relationships among habitat features within and adjacent to parks on avian abundance. Our intention with the relationship analysis concerning refugia effects was generally to understand whether habitat features within or adjacent to parks are influential in describing avifaunal patterns and whether these patterns vary across the park-needs gradient. Given how parks may functionally act as ‘islands’ in the cityscape, we predicted that larger parks near natural areas would have a greater abundance of birds following from other urban systems and also from the predictions of island biogeography (Donnelly and Marzluff, 2004; Molles and Sher, 2018; La Sorte et al., 2020). Additionally, we predicted that synanthropic birds affiliated with urban habitat features would be positively related to impervious surfaces, both within and adjacent to parks, and other features resembling dense urban form, e.g., less tree cover surrounding parks (Johnston, 2001). Further, we predicted that birds affiliated with shrubs, trees, and other natural amenities would be positively related to similar features within and adjacent to parks (Wood and Esaian, 2020).

## Materials and methods

### Study area and sampling design

We studied habitat and bird communities in 48 urban parks throughout LA. The LA metropolitan area has a population of over ten million people and spans an area of approximately 10,510 km<sup>2</sup> (U.S. Census Bureau, 2019). The region is characterized by a Mediterranean climate and experiences hot and dry summers contrasted with cool and wet winters. LA, which primarily covers the San Fernando and San Gabriel Valleys, the LA Plain, and the foothills of various hills and mountains ranges of the region, was formerly a diverse mosaic of wetlands, riparian forests, oak (*Quercus* spp.) and walnut (*Juglans* spp.) woodlands, coastal sage scrub, chaparral, and grassland ecosystems before intense development throughout the 20th century (Stein et al., 2007). The region is now mainly urbanized, dominated by a composite of large, medium, and small municipalities with extensive suburbs, numerous urban cores, and few natural green spaces distributed throughout. The greater metropolitan area is primarily bounded by the Pacific Ocean to the South and West and the Transverse and Peninsular Mountain Ranges to the North and East.

We initially selected a random sample of 60 managed urban parks set throughout LA. We identified these 60 parks using a polygon shapefile of the parks and open spaces in Los Angeles County, which we acquired from the Los Angeles County GIS repository (Los Angeles County, 2016a). To categorize the 60 urban parks based on park needs, we utilized the ‘Park Needs Assessment Detailed (Hosted–Public)’ geodatabase (Los Angeles County, 2016b). The geodatabase provides a spatial layer intended to highlight the 2016 needs assessment, that quantified the needs

for parks and recreation resources and estimated the potential cost of meeting the need across the County (Los Angeles County, 2016a). The needs assessment grouped locations of L.A. into six categories: very high, high, moderate, low, very low, and not participating. The majority of Los Angeles County falls under the categories of very high (32.2%), high (20.4%), and moderate needs (26.2%). Low (16.5%), and very low needs (4.6%) make up a considerably smaller percentage (Los Angeles County, 2016b). We used a spatial join to merge the 60 managed urban parks with the needs assessment spatial layer. We only retained parks with at least 20% of tree cover in the final sample because we assumed that parks composed primarily of grass, bare ground, or impervious surfaces, which typically were those dominated by ball fields or courts, would have fewer birds due to lack of habitat. Thus, all parks in the study were generally typical of urban parks in LA, with grassy fields, play areas, and trees (Figure 1). The final sample consisted of 48 parks: 10 in very high and 10 in high, which we merged into a ‘high needs category’ (20), 11 in moderate, and 17 in low (Figure 2; Supplementary Figure S1).

### Area search bird surveys

We surveyed parks using an area search method to quantify bird abundance (Loyn, 1986). We favored area search surveys over point counts because nearly all parks were small enough to be sampled in their entirety. We surveyed each park three times over two field seasons with one visit during the winter of 2018/2019 and two additional visits during the winter period of 2019/2020. We surveyed during the winter months from late October to late March as it is a time of year when wintering migratory birds are abundant in southern California (Garrett et al., 2012; Higgins et al., 2019). Surveys typically involved an observer walking on a set route throughout parks, identifying and counting each bird that was seen or heard within park boundaries.



**FIGURE 1**  
An example of an urban park included in this study (Villa Parke, City of Pasadena, Los Angeles County, California, United States). Photo credit, E. Wood.

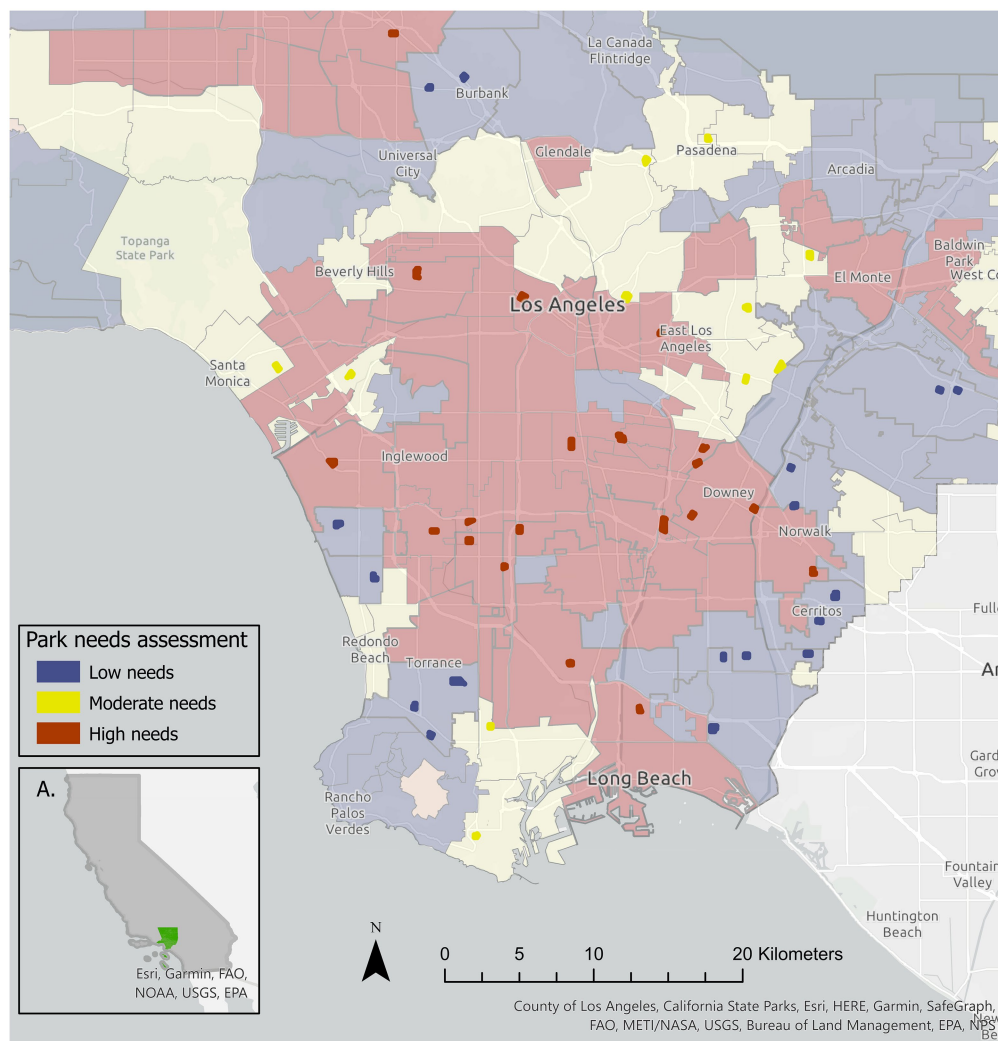


FIGURE 2

Locations of the 48 urban parks included in this study. The red, yellow, and blue color gradient indicates designations of Greater LA categorized by park need based on the Los Angeles Countywide Comprehensive Parks and Recreation Needs Assessment report (LA County, Department of Los Angeles County, 2016b). We grouped very high, and high needs parks into a 'high needs' group (red,  $n=20$ ), parks in We then included parks categorized within sections of the city as 'moderate' (yellow,  $n=11$ ) and 'low' (blue,  $n=17$ ) needs.

We used the Gaia GPS® mobile app to record the initial survey routes that we followed during each subsequent park visit. Care was taken to not double-count birds, especially flocking birds that would frequently move throughout a park during a survey. If an observer encountered a flock of birds, we recorded the number of species and abundance. If a similar composition of birds within a flock was encountered within 100-m of our previous observation, we omitted those from the survey to avoid double counting birds. Birds flying over parks or outside park boundaries were not recorded, as we were only interested in analyzing bird communities within park boundaries during the duration of a survey. We counted raptors and waterbirds in our survey, but they were not included in our analysis (see bird guilds below). Surveys began within an hour after sunrise and were concluded by 1,100h. to capture the prime activity of birds. The length of each survey

varied between parks to account for the variability in park size and tree cover. Larger parks with greater tree cover generally took longer to survey.

## N-mixture abundance calculations and bird guilds

To account for detection probability, which is a concern with wildlife count data (MacKenzie et al., 2017), we calculated N-mixture models, which are hierarchical models that incorporate spatial replicates of raw abundance count data (Royle and Nichols, 2003). The model estimates a detection probability for a given species, which is then utilized to adjust abundance estimates from a predicted model using an appropriate error distribution (e.g.,



**TABLE 1** Common, scientific names, and the American Ornithological Society (AOS) alpha 4-letter bird codes (Chesser et al., 2021) for 33 species included in the *N*-mixture modeling analysis.

Common name	Scientific	AOS	Naïve	Mean abundance	SE	Detection probability	SE
Rock pigeon	<i>Columba livia</i>	ROPI	0.48	15.8	0.77	0.46	0.02
Band-tailed pigeon	<i>Patagioenas fasciata</i>	BTPI	0.13	0.78	0.16	0.43	0.07
Eurasian collared-dove	<i>Streptopelia decaocto</i>	EUCD	0.38	2.70	0.28	0.53	0.04
Mourning dove	<i>Zenaida macroura</i>	MODO	0.67	24.8	5.34	0.12	0.03
Anna's hummingbird	<i>Calypte anna</i>	ANHU	1	32.5	5.12	0.11	0.02
Allen's hummingbird	<i>Selasphorus sasin</i>	ALHU	0.98	15.3	1.99	0.33	0.04
Acorn woodpecker	<i>Melanerpes formicivorus</i>	ACWO	0.06	0.23	0.09	0.37	0.14
Nuttall's woodpecker	<i>Dryobates nuttallii</i>	NUWO	0.38	1.30	0.57	0.20	0.09
Cassin's kingbird	<i>Tyrannus vociferans</i>	CAKI	0.67	3.82	1.25	0.20	0.06
Black phoebe	<i>Sayornis nigricans</i>	BLPH	1	10.6	4.84	0.24	0.11
Say's phoebe	<i>Sayornis saya</i>	SAPH	0.50	1.22	0.41	0.28	0.09
California scrub-jay	<i>Aphelocoma californica</i>	CASJ	0.31	1.31	0.35	0.27	0.07
American crow	<i>Corvus brachyrhynchos</i>	AMCR	0.83	17.2	1.66	0.23	0.02
Common raven	<i>Corvus corax</i>	CORA	0.52	6.69	1.24	0.20	0.04
Ruby-crowned kinglet	<i>Regulus calendula</i>	RCKI	1	10.6	2.04	0.31	0.06
Bushtit	<i>Psaltiriparus minimus</i>	BUSH	0.79	22.2	1.68	0.24	0.18
Northern mockingbird	<i>Mimus polyglottos</i>	NOMO	0.65	2.59	0.38	0.42	0.06
European starling	<i>Sturnus vulgaris</i>	EUST	0.79	22.2	2.43	0.22	0.02
Western bluebird	<i>Sialia mexicana</i>	WEBL	0.67	6.17	0.99	0.29	0.05
American robin	<i>Turdus migratorius</i>	AMRO	0.10	0.88	0.23	0.30	0.08
House sparrow	<i>Passer domesticus</i>	HOSP	0.65	10.7	0.75	0.42	0.03
House finch	<i>Haemorhous mexicanus</i>	HOFI	0.98	83.3	11.5	0.14	0.02
Lesser goldfinch	<i>Spinus psaltria</i>	LEGO	0.71	14	1.28	0.29	0.03
Lark sparrow	<i>Chondestes grammacus</i>	LASP	0.15	1.86	0.24	0.47	0.05
Dark-eyed junco	<i>Junco hyemalis</i>	DEJU	0.38	4.08	0.62	0.29	0.04
White-crowned sparrow	<i>Zonotrichia leucophrys</i>	WCSP	0.50	7.96	0.81	0.32	0.03
California towhee	<i>Melospiza crissalis</i>	CALT	0.25	1.05	0.39	0.22	0.08
Brewer's blackbird	<i>Euphagus cyanocephalus</i>	BRBL	0.27	12.2	0.91	0.32	0.02
Orange-crowned Warbler	<i>Vermivora celata</i>	OCWA	0.85	4.79	0.89	0.31	0.06
Yellow-rumped Warbler	<i>Setophaga coronata</i>	YRWA	1	65.8	6.33	0.35	0.01
Black-throated gray Warbler	<i>Setophaga nigrescens</i>	BTYW	0.38	0.98	0.44	0.22	0.10
Townsend's Warbler	<i>Setophaga townsendi</i>	TOWA	0.77	4.40	1	0.27	0.61

We also display the naïve detections (proportion of parks with a detected species), mean predicted abundance, and detection probability (*p*) derived from the intercept-only *N*-mixture analysis.

Poisson, Royle and Nichols, 2003). We fitted the intercept-only *N*-mixture model, using the 'pcount' function in the R package 'unmarked,' for 33 bird species (Table 1). We then estimated the posterior distribution of latent abundance from the *N*-mixture models for the 33 candidate bird species at each park using empirical Bayes methods from the unmarked package (function, 'ranef'; Fiske and Chandler, 2011). Therefore, when we present the results of bird abundance, we refer to the estimated abundances from the *N*-mixture models. We assumed observer bias in our survey was minimal, as one observer (AV) collected nearly all data (92% of observations), with EW and a handful of students occasionally completing surveys. Further, because we sampled over two seasons, we varied detection probability by season to account for potential year-to-year effects. Lastly, a critical assumption for estimating detection probability within a season is

'closure' (MacKenzie et al., 2017). While birds move frequently during the non-breeding period, we assumed that the focal species of this study were present and available during the winter months for detection throughout our surveys.

To focus our analysis on bird species that may have variable responses to park and urban habitat features, we created four bird habitat guilds. These included birds affiliated with forest and open woodland, shrublands (shrub), grassland, or urban ecosystems (urban) during the breeding season, assuming their habitat associations would be similar during the winter months (Supplementary Table S1; Clark, 2017; Billerman et al., 2021). The 'urban' birds are species often categorized as synanthropes (Supplementary Table S1). We also created a migratory and resident bird guild, which included species that either depart the L.A. area during the summer for breeding duties or stay within the

region (Supplementary Table S1). For each guild, we summed the estimated abundance of each bird within a guild to quantify a guild-specific estimated abundance value, which we used as dependent variables. We also summed the total migratory and resident bird groups for a total abundance group within each park (Supplementary Table S1).

## Habitat variables, remote sensing and spatial analysis

We used data from remote sensing platforms coupled with spatial processing to characterize habitat features within and adjacent to parks. We used a pixel-based image classification to derive habitat features within parks and within a 0.8 km (0.5 miles) buffer around each park. The purpose for characterizing habitat features in the 0.8 km buffers surrounding parks was to capture adjacent landscape characteristics and their effect on park avian communities, standard practice when performing landscape-extent analyses (Jimenez et al., 2022). We used a 2016 National Agriculture Inventory Project (NAIP) 4-band image data acquired from the Los Angeles County GIS Database for the classification (Los Angeles County, 2009). The image was taken 2 years before our sampling. However, we assumed that any potential differences in built structures surrounding parks, or infrastructure within parks, e.g., tree removal, would be negligible over the 2 years. Before performing the classification, we extracted the near-infrared, red, and blue bands from the NAIP image. We chose these bands because of their ability to distinguish between vegetation, manufactured objects, and other urban features (Wood et al., 2013).

To further differentiate between vegetation and other urban land-cover features, we created a Normalized Difference Vegetation Index (NDVI) layer (Pettorelli et al., 2011). NDVI measures vegetation 'greenness', ranging from a scale of  $-1$  (least green) to  $1$  (most green). NDVI is derived with the following equation:

$$\text{NDVI} = \frac{\text{Near Infrared Band} - \text{Red Band}}{\text{Near Infrared Band} + \text{Red Band}}$$

The green vegetation in our study area was typically characterized by high, positive NDVI values; dead vegetation, e.g., grass in the winter months, had low, positive values ( $\sim 0.05$ ). Bare ground had values closest to zero, and impervious surfaces all had negative values.

To increase the classification accuracy, we acquired raw LiDAR point cloud data for the study area from a National Oceanic and Atmospheric Administration (NOAA) data repository (OCM Partners, 2022). We used the first-return LiDAR data points to create a Digital Surface Model (DSM) and the ground return data points to create a Digital Terrain Model (DTM). Using these two layers, we created a Normalized Digital Surface Model (nDSM) that depicts features elevated from the

ground, such as trees and buildings. nDSM is derived by subtracting the DSM with the DTM layer. We created the nDSM layer to differentiate trees from grass by their height differences. Finally, we combined the nDSM, NDVI, and the 3-band NAIP image to create a new 5-band image layer used as the raster input for the final classification. We used a Support Vector Machine (SVM) pixel-based classification to classify tree cover, grass cover, bare ground, and impervious surfaces within and around each park. Water and shadows were also classified since these features were common throughout the landscape. However, we did not include water or shadows in our final analysis assuming they had a small effect on the landbirds of our study.

To assess the accuracy of the remote sensing classification, we used 200 assessment points and computed a confusion matrix that revealed a classification accuracy of approximately 86% (Supplementary Table S2). We then used the *Tabulate Area* tool in ArcMap to calculate the proportion of each feature type within and around each park. Additionally, we used the *Near* tool in ArcGIS Pro to calculate the Euclidian distance between each park and the nearest natural area. We designated areas as 'natural' if classified as 'protected areas' or 'open spaces' within LA County park's polygon shapefile (Los Angeles County, 2009). Last, we determined the median income of census tracts where parks were situated using spatial data organized by Southern California Association of Governments (2016). We used the median income data as our indication of potential luxury-effect patterns based on our sampling design (Leong et al., 2018; Schell et al., 2020). These income data were from 2016 and based on projections from the United States 2010 census (U.S. Census Bureau/American FactFinder, 2010). We used these 2016 data as we assumed they approximated income levels in the sections of the city that were comparable to the time we collected data in 2017/2018 and 2018/2019. All remote sensing and spatial analyses were completed using ArcMap and ArcGIS Pro (ESRI, 2020).

## Statistical analysis

### Objective 1: Patterns of habitat features and avian communities

We completed three analyses to characterize patterns of habitat features and bird abundance in parks across the income gradient. First, to quantify differences in bird guild abundances, park features, and landscape characteristics across socioeconomic statuses, we performed a series of one-way analysis of variance tests (ANOVA). The categorical fixed factor for each model was the park-needs category (low, moderate, high). If tests were significant, we employed a Tukey's HSD test. As we were making three comparisons among income levels for a particular variable, we used a Bonferroni correction of the alpha value,  $\alpha = 0.05/3 = 0.02$  to assess significance.

Second, to identify the degree of dissimilarity in the bird community concerning the park-needs categories, we conducted a one-way analysis of similarities test (ANOSIM; Oksanen et al.,

2019), using the Bray–Curtis dissimilarity of the square-root transform of the 33 species from the *N*-mixture analysis grouped among high, moderate, and low park needs groups. If an ANOSIM test was significant at the alpha value of 0.05, we calculated pairwise comparisons by performing an ANOSIM analysis of either low-high, low-moderate, or moderate-high. Like the ANOVA analysis, as we were making three comparisons among income levels for a particular variable, we used a Bonferroni correction of the alpha value,  $\alpha = 0.05/3 = 0.02$  to assess significance.

Third, to further assess differences in the avian community across the park-needs gradient, we computed a non-metric multidimensional scaling (NMDS) analysis. We again used the Bray–Curtis dissimilarity of the square-root transform of the 33 avian species from the *N*-mixture analysis. We created an ordination graph of the 2-D representation of the avian community using the *vegan* package in R, and we overlaid habitat vectors on the ordination using the 'envfit' function in *vegan* (Oksanen et al., 2019). The envfit function assesses the correlation of both habitat and avian species vectors with the first two axes of the ordination (Oksanen et al., 2019) and thus provides a measure of continuous change of the avian community concerning habitat variables across the park needs gradient.

## Objective 2: Relationships among habitat features and bird abundance

To understand the relative effects of local and landscape habitat features on avian park communities, we used a model selection approach, where we fitted a series of generalized linear models (GLMs), regressing the independent habitat variables against the seven bird guilds, which were the dependent variables in the analysis. Because our data were based on counts (abundance), we used Negative Binomial GLMs with a log-link function. We used Negative Binomial models to account for overdispersion in the Poisson distributed count data, which was evident based on calculating the ratio of the residual deviance to the residual degrees of freedom for each model (Zuur et al., 2011). We developed seven distinct model sets, with one for total bird abundance, and then six others for the bird guilds (forest and woodland, shrubland, grassland, urban, migratory, and resident) regressed against 11 independent variables and the intercept-only model. We fitted all models as univariate combinations of an independent and dependent variable. We did not explore multi-variable models or interactions primarily because we were interested in the general correlation of a given independent variable with a dependent variable. Further, numerous independent variables were moderately to highly correlated, thus making fitting multiple variable models challenging (Supplementary Figure S2). For organization purposes, we grouped our independent variables based on whether they were related to the luxury effect (median income), island biogeography (park size and distance to the nearest natural area); park composition (the % cover of impervious surface, trees, grass, and bare ground); and urban habitat features surrounding parks (the % cover of impervious surface, trees, grass, and bare ground).

Each independent variable in our analysis was either biologically relevant to the avifauna of our study (e.g., % tree cover), or commonly used in urban ecology studies as a means for understanding potential conservation and habitat associations (e.g., % bare ground). Therefore, each model had biological or management significance.

We used an Akaike's Information Criterion (AIC) model selection framework to determine which variable was the most important predictor of bird abundance in parks within each of the three analysis extents. We determined 'top models' as those with a  $\Delta AIC < 2$  (Anderson and Burnham, 2002). We also computed  $R^2$  values based on the Kullback–Leibler-divergence ( $R_{kl}^2$ ) generated from calculating the likelihood ratio index of a fitted model (Cameron and Windmeijer, 1997). We completed all analyses using the R programming language (R Core Team, 2017), with code and figures run and created using 'rmarkdown' (Allaire et al., 2022). We used Adobe Illustrator to finalize the figures (Adobe Inc., 2019).

## Results

The average detection probability for the 33 species included in the *N*-mixture analysis was 0.30 (Table 1), with the Eurasian Collared-Dove (*Streptopelia decaocto*) having the highest detection probability ( $p = 0.52$ ) and the Anna's Hummingbird (*Calypte anna*) having the lowest ( $p = 0.11$ ). Four bird species were detected at every park (Naïve detection), including the Anna's Hummingbird, the Black Phoebe (*Sayornis nigricans*), the Ruby-crowned Kinglet (*Regulus calendula*), and the Yellow-rumped Warbler (*Setophaga coronata*; Table 1). The average mean-estimated abundance for all species was 12.49 individuals per park (Table 1). The most abundant birds in our study were the House Finch (*Haemorhous mexicanus*; mean estimated abundance across parks = 83.3) and the Yellow-rumped Warbler (65.8), and the rarest species were the Acorn Woodpecker (*Melanerpes formicivorus*; mean estimated abundance across parks = 0.23), the Band-tailed Pigeon (*Patagioenas fasciata*; 0.78), the American Robin (*Turdus migratorius*; 0.88), and the Black-throated Gray Warbler (*Setophaga nigrescens*; 0.98), all with mean abundances of less than one per park (Table 1).

## Objective 1: Patterns of habitat features and avian communities

Overall, there were few differences in habitat characteristics in parks and the surrounding urban environment across the low-, moderate-, and high-needs gradient of our study (Table 2). Notable variables that varied included the median income of the residential areas surrounding parks (value of  $p < 0.01$ ), which was 30% higher in low than high needs areas; the distance to natural areas, where parks in high-needs areas were over twice as far from natural areas as parks in high-income areas (value of  $p < 0.01$ ),

**TABLE 2** Mean $\pm$ S.E. summaries of the total abundance of birds, and six additional groups indicating combinations of bird species associated with forest, shrub, grassland, or urban ecosystems during the breeding period, or whether species are wintering migratory birds (migratory) or resident to the L.A. study area.

	Low	Moderate	High
<b>Bird abundance (km<sup>2</sup>)</b>			
Total	25.9 $\pm$ 9.69	28.4 $\pm$ 12.8	31.5 $\pm$ 13.8
Forest and woodland	11.5 $\pm$ 4.47	12.4 $\pm$ 4.76	13 $\pm$ 6.06
Shrub	4.24 $\pm$ 2.49	4.34 $\pm$ 2.19	4.91 $\pm$ 2.45
Grassland	1.9 $\pm$ 0.89	3.17 $\pm$ 2.57	2.98 $\pm$ 1.98
Urban	14.7 $\pm$ 5.94	16.5 $\pm$ 7.96	19.1 $\pm$ 9.3
Migratory	6.94 $\pm$ 2.97	6.93 $\pm$ 3.12	7.59 $\pm$ 3
Resident	19 $\pm$ 7.18	21.4 $\pm$ 9.8	23.9 $\pm$ 11.4
<b>Luxury-effect</b>			
Median income	77,289 <sup>B</sup> $\pm$ 22,320	67,156 <sup>AB</sup> $\pm$ 25,480	53,547 <sup>A</sup> $\pm$ 19,970
<b>Island biogeography</b>			
Park size (km <sup>2</sup> )	0.07 $\pm$ 0.02	0.07 $\pm$ 0.02	0.07 $\pm$ 0.03
Dist. natural area (km)	2.94 <sup>B</sup> $\pm$ 1.64	2.38 <sup>B</sup> $\pm$ 1.36	6.47 <sup>A</sup> $\pm$ 3.26
<b>Park habitat (within)</b>			
Impervious	0.19 $\pm$ 0.09	0.21 $\pm$ 0.13	0.21 $\pm$ 0.13
Trees	0.26 $\pm$ 0.09	0.26 $\pm$ 0.08	0.22 $\pm$ 0.07
Grass	0.45 $\pm$ 0.09	0.38 $\pm$ 0.07	0.39 $\pm$ 0.1
Bare	0.07 $\pm$ 0.03	0.1 $\pm$ 0.05	0.09 $\pm$ 0.04
<b>Urban habitat (adjacent)</b>			
Impervious	0.61 $\pm$ 0.06	0.59 $\pm$ 0.1	0.62 $\pm$ 0.08
Trees	0.12 <sup>B</sup> $\pm$ 0.03	0.14 <sup>AB</sup> $\pm$ 0.06	0.09 <sup>A</sup> $\pm$ 0.03
Grass	0.15 $\pm$ 0.04	0.14 $\pm$ 0.04	0.14 $\pm$ 0.06
Bare	0.07 $\pm$ 0.03	0.07 $\pm$ 0.03	0.07 $\pm$ 0.04

See [Supplementary Table S2](#) for combinations. We also display summaries of 11 urban form or habitat variables grouped whether they were related to the luxury effect, island biogeography, within park habitat (within), or adjacent urban habitat of parks. The low, moderate, and high categories refer to park needs.

The km<sup>2</sup> following 'Bird abundance' and 'Park trees and shrubs' indicates values for each variable within the group were standardized by the area of parks that were surveyed (Park Size km<sup>2</sup>).

Variables with different superscript letters indicate significant differences based on a one-way ANOVA and Tukey–Kramer test. We used a Bonferroni adjusted value of  $p$  of 0.05/3 = 0.02 to account for the three comparisons made within groups.

which indicated their general position in the center of the metropolis; and the proportions of tree cover in the urban environment surrounding parks, which was 25% greater in low than high needs areas of the city (value of  $p=0.02$ ; [Figure 3](#); [Table 2](#)). Interestingly, the cover of trees within parks did not vary across the park-needs gradient ([Figure 3](#); [Table 2](#)), suggesting that the luxury-effect phenomenon of tree cover within residential zones does not apply to urban parks (e.g., see adjacent tree cover results). There were no differences in the avian guilds across the park-needs gradient also suggesting that the luxury effect does not apply to explaining park avifauna in L.A. ([Table 2](#)).

Similar patterns were also evident when analyzing the dissimilarities of avian communities among the park-needs categories. There was slight evidence of dissimilarities in communities for total bird abundance (ANOSIM  $R=0.07$ , value of  $p=0.07$ ) and forest and woodland bird abundance ( $R=0.08$  value of  $p=0.05$ ). However, there were no significant differences in dissimilarities when analyzing pairwise comparisons. ANOSIM values range from  $-1$  to  $1$ , with values closer to zero indicating no dissimilarities across groups. Thus, the effects were weak for total and forest and woodland abundance. The only group that did

show evidence of dissimilarity across the park-needs groupings was resident birds ( $R=0.07$ , value of  $p=0.04$ ), which displayed a trend in dissimilarity between parks in high- and low-needs areas ( $R=0.06$ , value of  $p=0.06$ ), and between parks in moderate- and low-needs areas (ANOSIM  $R=0.11$ , value of  $p=0.06$ ). However, we note the pairwise comparisons were not significant at the Bonferroni adjusted alpha value of 0.02, again suggesting weak dissimilarities. All other avian guilds were similar across the park-needs categories, with value of  $ps$  ranging from 0.13 (urban), 0.19 (grassland), 0.39 (shrub), and 0.42 (migratory).

The NMDS analysis had reached a stress solution of 0.22 suggesting modest confidence in the outputs. Nevertheless, the analysis revealed a few important distinctions in habitat characteristics and avian communities across the continuous park-needs gradient. There were five important habitat vectors identified, including urban trees surrounding parks, bare ground within parks, distance to the nearest natural area, park size, and median income ([Figure 4](#)). Median income was positively correlated with axis 1, whereas park bare ground cover was negatively correlated with axis 1 ([Figure 4](#)). The distance to the nearest natural area and park size were positively correlated with



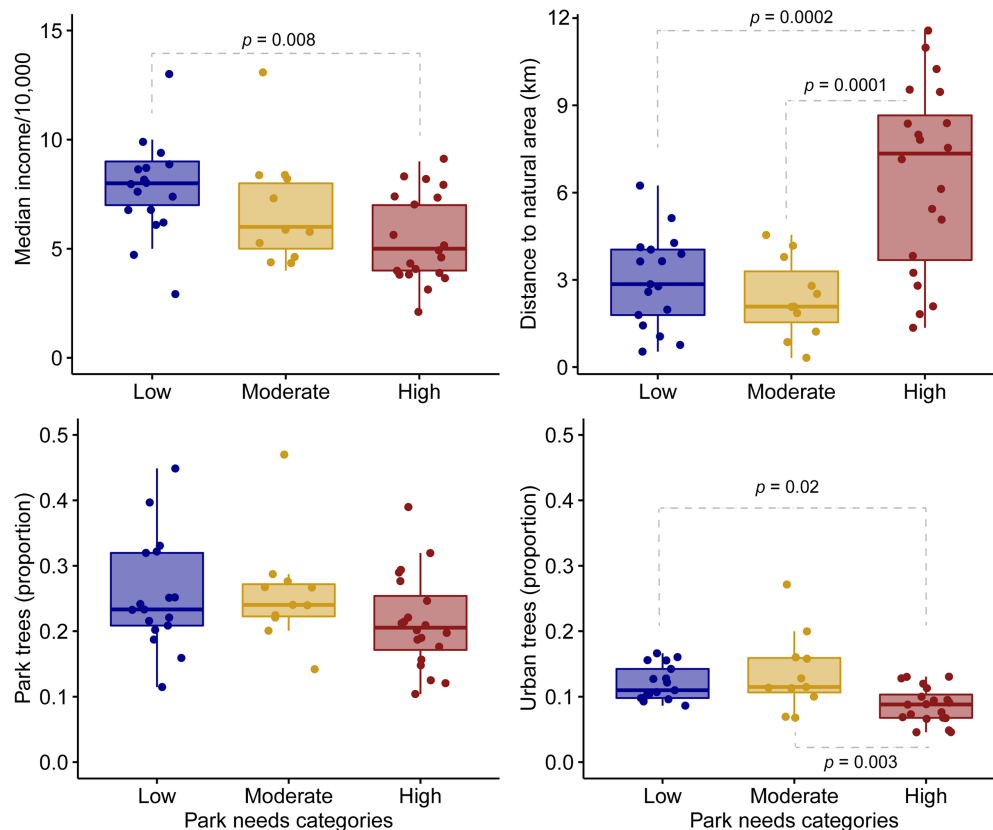


FIGURE 3

Box plots depicting patterns of median income, distance to the closest large natural area (km), the proportion of tree cover within parks [Park trees (proportion)], and the proportion of tree cover adjacent to parks [Urban trees (proportion)] across a gradient of low-, moderate-, and high-park needs. Dotted gray lines linking boxes indicate significant differences based on a one-way ANOVA and Tukey–Kramer test, or a Kruskal–Wallis rank-sum test followed by a nonparametric multiple-comparisons procedure, based on relative contrast effects. We used a Bonferroni adjusted value of  $p$  of  $0.05/3=0.02$  to account for the three comparisons made within groups.

axis 2, while urban trees surrounding parks was negatively correlated with axis 2 (Figure 4). Avian communities from parks in low-needs areas were weakly positively associated with income and negatively with bare ground cover within parks (Figure 4). Parks in high-needs areas were weakly aligned with distance to a natural area and park size, again indicating the location of the high-needs areas in the center of the city, further from the surrounding natural areas where few large parks occurred (Figure 4). Birds in parks in low-needs areas of LA were aligned with income (positively) and bare ground cover (negatively) within parks (Figure 4).

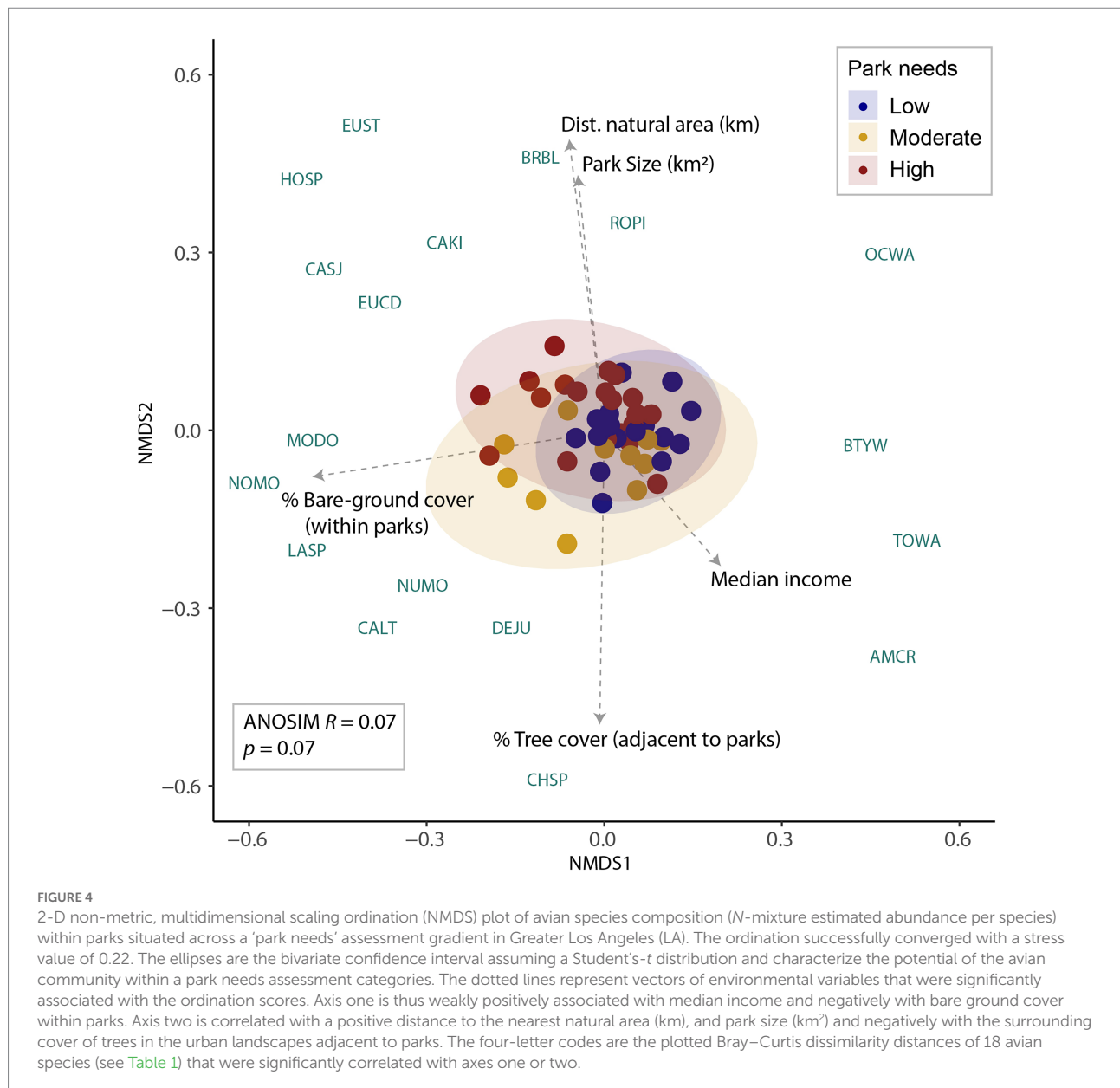
## Objective 2: Relationships among habitat features and bird abundance

The most influential variables explaining bird abundance varied for each of the bird guilds but were generally aligned with island biogeography variables, and then components of the parks and not the surrounding landscape (Table 3; Appendix S1). Forest and woodland birds and migratory birds were best explained by

distance to the nearest natural area (negative and positive association, respectively; Table 3; Figures 5A,E), whereas shrub and grassland bird abundance was best described by park size (Table 3; Figures 5B,C). The park-size finding suggests a species-area effect, which typically explains richness patterns. However, in our case, park size characterized avifaunal abundance. The cover of bare ground was positively related to total, urban, and resident bird abundance and highly competitive with grassland bird abundance ( $\Delta AICc = 0.10$ ; Table 3; Figures 5D,F). There were fewer relationships between the urban environment habitat variables and birds within parks (Table 3). These results indicate that birds will likely use parks as habitat throughout the city depending on the specific management of the parks themselves, and not necessarily due to drivers from the adjacent habitat.

## Discussion

Our results indicated that parks are a refuge for avifauna in park-poor sections of cities. We uncovered three lines of evidence to support our main conclusion. First, we initially predicted a



refugia effect, where parks with higher stressors surrounding their boundaries would have higher individuals than those with low stresses. Our analysis found support for this prediction. Bird abundance patterns for species affiliated with forest, shrub, and woodland ecosystems were generally similar across the park-needs gradient of LA, indicating that in areas of the metropolis with high stresses (low income and high-park needs) birds utilize parks in relatively high frequencies. Interestingly, this pattern is generally opposite to what is found outside of parks, where forest-affiliated birds are far denser in high-income residential areas of LA (low park needs) than in low-income areas (high park needs; Wood and Esaian, 2020). This result suggests that birds typical of natural ecosystems surrounding L.A. use parks in otherwise inhospitable areas of the city at comparable levels to locations that have abundant greenery outside of park

boundaries. In a similar line of evidence, avian communities varied slightly among parks in high and low-needs areas of the city, with few habitat variables weakly associated with avifaunal community structure, including median income, bare ground cover within parks, urban tree cover surrounding parks, park size, and distance to natural areas. Though the patterns were weak, these results, especially for median income and urban tree cover surrounding parks, provided some support that the surrounding cityscape may indeed filter the species pool found within parks (e.g., Aronson et al., 2016). However, we again stress that the patterns we uncovered in parks are far weaker than the filtering effects found outside of parks in residential areas (Wood and Esaian, 2020), again providing support for their refugia potential in dense urban conditions. Lastly, bird abundance patterns were related to numerous island biogeographic and

**TABLE 3**  $\Delta$ AICc values based on a model selection routine for bird abundance regressed against 11 independent variables and the intercept-only model.

	Total	Forest	Shrub	Grassland	Urban	Migratory	Resident
Intercept	9.74	7.09	2.69	12.33	10.19	2.76	10.23
Luxury-effect							
Median income	11.21	8.44	4.82	10.79	10.45	4.76	10.86
Island biogeography							
Park size (km <sup>2</sup> )	<b>3.01<sup>+</sup></b>	<b>5.65<sup>+,†</sup></b>	<b>0<sup>+</sup></b>	<b>0<sup>+</sup></b>	<b>6.52<sup>+</sup></b>	<b>0.18<sup>+</sup></b>	<b>4.85<sup>+</sup></b>
Distance to natural (km)	11.91	<b>0<sup>+</sup></b>	2.47	<b>9.61<sup>+,†</sup></b>	<b>8.94<sup>+,†</sup></b>	<b>0<sup>-</sup></b>	11.07
Park habitat (within)							
Impervious (%)	<b>3.09<sup>-</sup></b>	<b>4.72<sup>-</sup></b>	<b>1.37<sup>-</sup></b>	13.50	<b>5.59<sup>-</sup></b>	<b>0.37<sup>-</sup></b>	<b>4.93<sup>-</sup></b>
Trees (%)	10.50	8.63	3.26	9.80	10.33	5.00	10.48
Grass (%)	<b>3.57<sup>+</sup></b>	8.74	<b>2.13<sup>+,†</sup></b>	12.39	<b>4.56<sup>+</sup></b>	<b>0.71<sup>+</sup></b>	<b>5.23<sup>+</sup></b>
Bare (%)	<b>0<sup>+</sup></b>	6.96	3.53	<b>0.10<sup>+</sup></b>	<b>0<sup>+</sup></b>	2.31	<b>0<sup>+</sup></b>
Urban habitat (adjacent)							
Impervious (%)	10.79	<b>5.48<sup>-</sup></b>	4.93	14.45	12.07	4.37	11.45
Trees (%)	11.98	7.61	3.59	12.29	12.21	5.03	12.47
Grass (%)	11.92	9.29	3.97	14.61	12.23	4.93	12.26
Bare (%)	<b>7.03<sup>+</sup></b>	6.99	4.18	13.32	<b>7.77<sup>+</sup></b>	3.65	<b>7.49<sup>+</sup></b>

The seven dependent variables refer to birds affiliated with forest and woodland (forest), shrub, grassland, or urban ecosystems during the breeding period. Migratory and resident indicate whether birds migrate from the Greater Los Angeles wintering grounds of this study to more northerly breeding grounds (Migratory), or whether birds breed locally (Resident). Total refers to the total estimated bird abundance. Independent variables were grouped based on whether they were related to island biogeography, within park habitat (within), or adjacent urban habitat of parks.

Values in bold indicate significant relationships (value of  $p < 0.05$ ), and + and – signs following bolded models indicate the direction of the relationship. <sup>†</sup>Indicates significant relationships at value of  $p < 0.10$ .

habitat variables within parks, which were stronger than habitat variables surrounding parks. These findings suggested the important role parks have in providing habitat for birds, regardless of whether they are in high- or low-park needs areas of the city. Overall, in addition to the benefits to people, our work suggests park development in park-poor areas of L.A. would also have positive effects on birds.

## The luxury effect, parks, birds, and their habitat

Among the many known drivers of biodiversity in cities, the luxury effect posits that vegetation cover and wildlife biodiversity follows patterns of wealth (Leong et al., 2018). The Park Score Index clearly describes the luxury effect highlighting the disparity in the distribution of parks in high- and low-income areas of LA. Because the luxury effect is a prevalent and defining feature of biodiversity in residential areas and other greenspaces of LA (Clarke et al., 2013; Wood and Esaian, 2020), we were interested in testing the luxury effect based on avifaunal patterns found within parks across the park-needs gradient. We found evidence for the luxury effect of the tree cover surrounding parks, which has been repeatedly documented in LA and many other cities (e.g., Schwarz et al., 2015). But we did not find support for the luxury-effect hypothesis for other habitat conditions, especially within parks. While our findings did not match our expectations, our results were in line with a handful of other studies from around the world. For example, in

Sydney, Australia, topography was the strongest predictor of plant abundance within parks rather than income (Zivanovic and Luck, 2016). Further, in Phoenix, Arizona, the income of the surrounding residential areas was not a strong predictor of park vegetation. Instead, the median year of development and whether residents had a graduate degree best-explained park vegetation abundance and richness (Martin et al., 2004). The patterns from Sydney and Phoenix along with our own suggest that urban parks are likely built and managed similarly across a cityscape regardless of the surrounding socioeconomic patterns.

The similarity in habitat conditions within parks across the park-needs gradient carried over to influence birds, which, also indicated a lack of support for the luxury-effect hypothesis in parks in LA. In other areas of the world, habitat features that were similar between high- and low-income areas of cities supported similar biodiversity patterns. For example, there was no evidence of the luxury effect when considering bird diversity patterns in greenspaces throughout Johannesburg, South Africa (Howes and Reynolds, 2021). Rather, the use of water bodies, which were historically implemented to segregate white and black populations of the city now buffer bird diversity patterns in low-income areas of Johannesburg (Howes and Reynolds, 2021). As Leong et al. (2018) suggested, correlations between bird populations and socioeconomics may be directly attributed to differences in vegetation cover across income. Given the similarity of habitats in LA parks, it appears parks can buffer avian communities across income gradients, which again suggests a refugia benefit of parks in LA for urban avifauna.

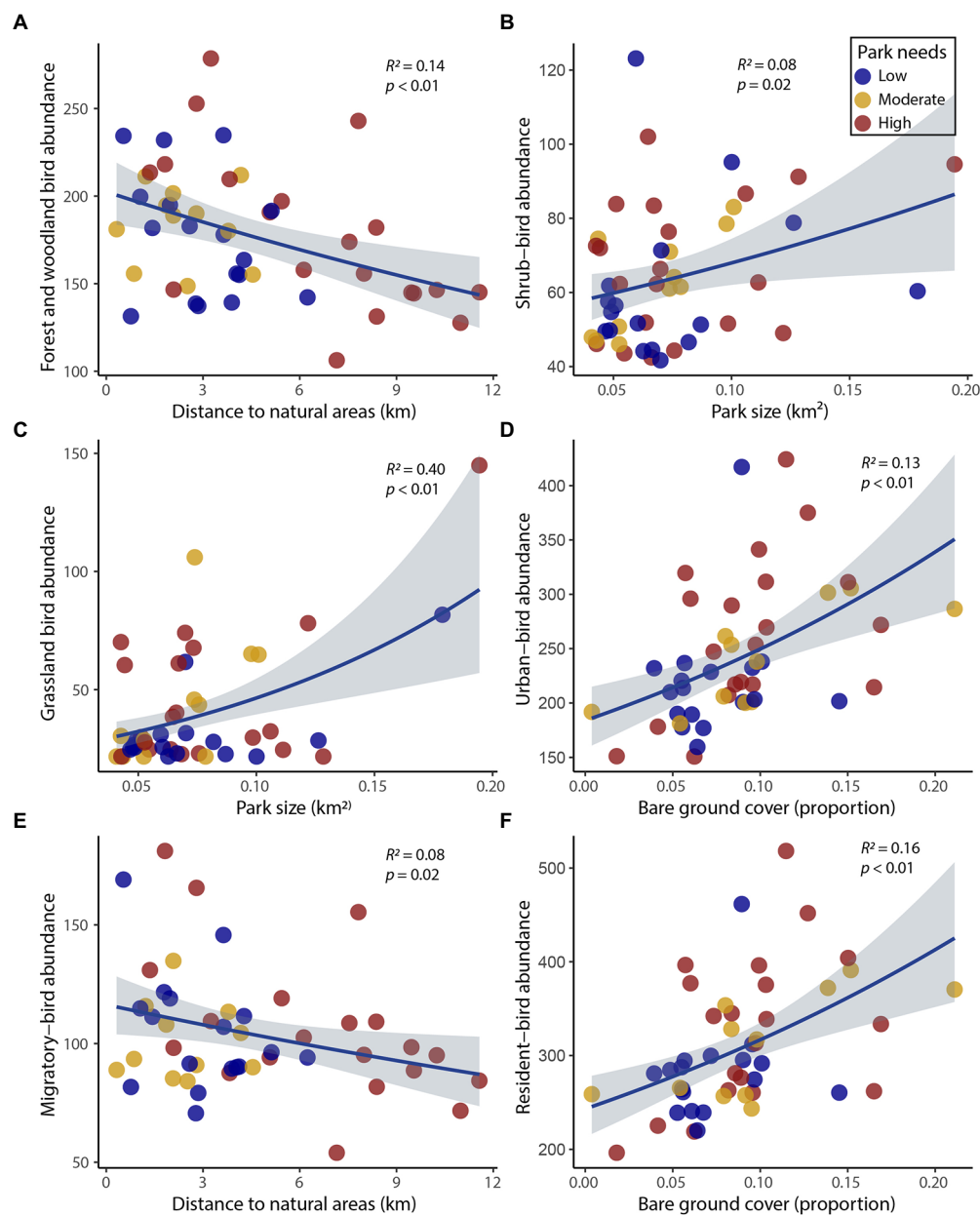


FIGURE 5

Scatterplots depicting the relationships of the top models for (A) forest and woodland, (B) shrub, (C) grassland, (D) urban, (E) migratory, and (F) resident bird abundance with independent variables. We generated the fitted line and confidence interval (gray shading) based on a negative binomial generalized linear model analysis. The  $R^2$  values are Kullback–Leibler-divergence-based  $R^2_{kl}$  values generated from calculating the likelihood ratio index of a fitted model (Cameron and Windmeijer, 1997).

## Parks as habitats for birds

Many studies from around the world have indicated the importance of urban greenspaces, including parks, to birds, which our study strongly supports (e.g., Blair, 1996; Jokimäki and Suhonen 1998; Fernández-Juricic and Jokimäki, 2001; Cornelis and Hermý, 2004; Colding and Folke, 2009; Carbó-Ramírez and Zuria, 2011; Ikin et al., 2013; Zivanovic and Luck, 2016; Amaya-Espinel et al., 2019; Villaseñor and Escobar, 2019; Zhang and

Huang, 2020). Nevertheless, our study uncovered some potentially interesting patterns that merit discussion regarding the potential of parks as habitats for birds. For example, bird species such as the Townsend's (*Setophaga townsendi*), Orange-crowned (*Vermivora celata*), and Black-throated Gray Warblers (*Setophaga nigrescens*) were aligned with parks in low-needs areas of the city. Parks in these areas were embedded within affluent zones of LA with high tree cover surrounding the boundaries of parks (Wood and Esaian, 2020). While park features in low- and high-needs areas



of the city were similar, the surrounding tree cover was dissimilar. The three wood-warblers (*Parulidae*) are forest and woodland breeding species and are common during the nonbreeding period in parts of LA with high tree cover (Wood and Esaian, 2020). Thus, these results suggest there are potentially important neighborhood-level filters in high-income areas attracting birds to affluent sections of the city that carry over to use the parks (Aronson et al., 2016). This pattern is similar to what is found in residential areas in LA (Wood and Esaian, 2020), but, as we previously discussed, the effects were far weaker within parks.

We also uncovered similar filtering effects when examining distribution patterns of birds that require open areas within parks (e.g., bare ground), where species such as the Lark Sparrow (*Chondestes grammacus*), Cassin's Kingbird (*Tyrannus vociferans*), and California Towhee (*Melospiza crissalis*) were generally more abundant. The Lark Sparrow and Cassin's Kingbird are species of grassland and savanna-type conditions (Billerman et al., 2021), conditions that parks superficially and structurally resemble (Figure 1). The bare ground could be a surrogate for these open conditions where birds may capture insects by flying out from the perches of trees. Or it may be possible that these bird species are attracted to other resources associated with the bare ground, e.g., shrubs planted next to ballfields, seeds, or dust for bathing. Large swaths of the valleys of LA were formerly grassland and shrubland (Ethington et al., 2020), so there could be a historic signal for birds requiring these ecosystem types to use parks in an otherwise heavily urbanized landscape.

We also desired to understand the relationships of habitat adjacent to parks in influencing avifaunal patterns within parks. Surprisingly, we found few important relationships when examining the effect of the surrounding cityscape on avian abundance patterns. The exception was bare ground cover surrounding parks, which was positively related to total, urban, and resident bird abundance. Bare ground in the surrounding landscape was generally associated with construction sites or vacant lots. Unlike roads and buildings (i.e., impervious surfaces), which isolate and limit the movement of birds within the urban landscape (Fernández-Juricic and Jokimäki, 2001; Tremblay and St Clair, 2011). Bare ground may affect birds at a more local scale, for example, providing habitat for a species such as a Mourning Dove (*Zenaidura macroura*). Regardless, what is clear from our results is that habitat within parks generally had stronger effects than habitat adjacent to parks on birds in LA.

## Parks as islands in the cityscape

The theory of island biogeography has been well documented in many natural systems around the world and has been extensively tested in anthropogenic systems under the assumption that larger patches near the 'mainland' will harbor greater biodiversity (Molles and Sher, 2018). Our study suggested that parks function as island systems within the urban landscape, which supports previous investigations on this theme (e.g., Fernández-Juricic and Jokimäki,

2001). Larger parks generally had greater bird abundance than smaller ones, which indicates a modification of the classic species-area curve (theoretically focused on richness) suggesting larger parks will harbor more individuals (Zhang and Huang, 2020). Amaya-Espinel et al. (2019), also reported greater bird abundance with the increasing size of urban parks in Santiago, Chile, as did Kang et al., (2015) in remnant urban forest patches of Seoul, Korea. The opposing effects of distance to natural areas on bird compositional patterns implies that the definition of a 'mainland' is not uniform for all birds in the urban context. While we defined a mainland as any natural area (i.e., protected areas and open spaces), the mainland for synanthropic species is likely the city itself, as evidenced by the distinct compositional patterns of urban birds in parks further from natural areas (Appendix S1). Taken together, our findings provide strong support that island biogeographic effects explain a significant amount of the variability in bird community patterns within parks throughout LA.

## Income inequality, park avifauna, and the virtuous cycle

Like many cities across the world, LA's park-poor areas are generally embedded within low-income areas of the metropolis. These areas are characterized by high-building density, vast stretches of impervious surface, and little green infrastructure, all of which unsurprisingly provide little habitat for birds that are not synanthropic. Moreover, park-poor areas of LA also have some of the lowest densities of city parks *per capita* (Wolch et al., 2005), presenting a disproportionate public health concern for human communities (de Vries et al., 2003). Our work details the value of parks in buffering avian communities in park-poor areas and points towards a potential win-win situation when also considering the public health crisis that is prevalent in disadvantaged communities in United States cities. A conceptual approach that highlights this win-win scenario is *via* a framework for socio-ecological virtuous cycles in conservation (Morrison, 2016). The framework suggests a series of linked objectives that follow a particular intervention to improve conditions for biodiversity, which are interrelated with benefits to individual people and their communities. A potential intervention based on our results is simply park development, which is aggressively being pursued in LA, especially in underserved communities (City of Los Angeles Department of Recreation and Parks, 2019). Given the many known benefits of nature on public health, park development could carry over to improve the well-being of people in cities (e.g., Brown and Grant, 2005). The benefit of parks could then inspire a continued desire for change and improvements within a community. This may, theoretically, lead to sustained benefits to the individuals of a community and the community as a whole. While the application of the conceptual nature of a virtuous cycle is infinitely more complicated in practice, L.A. is providing a model case study

for the benefits of park development on its biodiversity and people. Our work strongly supports the benefits of parks to birds in park-poor communities. Follow-up work should blend biodiversity research with the people who utilize parks, including their feelings or beliefs, cultural preferences, and desires for future greening initiatives in their communities to quantify the win-win potential of parks.

## Data availability statement

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

## Ethics statement

Ethical review and approval was not required for the animal study because this work is fully observational, and no birds were handled during any duration of the study.

## Author contributions

AV and EW conceived the research, designed the study, analyzed data and produced figures, and wrote and edited the manuscript. AV conducted fieldwork and data development. All authors contributed to the article and approved the submitted version.

## Funding

This work was supported by the Society for Conservation Biology, the Santa Monica Bay Audubon Society, and the

California State University Minority Opportunities in Research (MORE), grant #R25 GM061331.

## Acknowledgments

The authors thank A. Trigueros, G. Sercel, N. Garcia, E. Pedroza, C. Benitez, P. Larramendy, and M. Puche for their help with data collection and project establishment. S. Wright and K. Fisher provided valuable feedback on the manuscript. Two reviewers provided helpful comments.

## Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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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.2022.958572/full#supplementary-material>

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## OPEN ACCESS

## EDITED BY

Federico Morelli,  
Czech University of Life Sciences  
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## REVIEWED BY

Piotr Tryjanowski,  
Poznan University of Life Sciences,  
Poland  
John Morgan,  
La Trobe University, Australia

## \*CORRESPONDENCE

John English  
j.english@mail.utoronto.ca

## SPECIALTY SECTION

This article was submitted to  
Urban Ecology,  
a section of the journal  
Frontiers in Ecology and Evolution

RECEIVED 15 April 2022

ACCEPTED 10 October 2022

PUBLISHED 31 October 2022

## CITATION

English J, Barry KE, Wood EM and  
Wright AJ (2022) The effect of urban  
environments on the diversity  
of plants in unmanaged grasslands  
in Los Angeles, United States.  
*Front. Ecol. Evol.* 10:921472.  
doi: 10.3389/fevo.2022.921472

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# The effect of urban environments on the diversity of plants in unmanaged grasslands in Los Angeles, United States

John English<sup>1\*</sup>, Kathryn E. Barry<sup>2</sup>, Eric M. Wood<sup>3</sup> and  
Alexandra J. Wright<sup>3</sup>

<sup>1</sup>Department of Biological Sciences, University of Toronto, Toronto, ON, Canada, <sup>2</sup>Ecology and Biodiversity, Faculty of Science, Institute of Environmental Biology, Utrecht University, Utrecht, Netherlands, <sup>3</sup>Department of Biological Sciences, California State University, Los Angeles, Los Angeles, CA, United States

Urbanization is a strong driver of plant diversity and may have complex effects on developed ecosystems. Nevertheless, it remains unclear whether urban environments increase or decrease plant biodiversity compared with rural environments. Further, it is also unclear how non-native plant species contribute to spatial diversity patterns and ecosystem services. Better understanding these diversity drivers across gradients of urbanization has the potential to enhance native species conservation (e.g., targeted restoration activities), leading to positive feedbacks for broader promotion of biodiversity and societal benefits (e.g., links with native biodiversity and human health). In this study, we hypothesized that for plant species in unmanaged grasslands, urbanization would lead to declines in diversity at both small and medium scales. We established a network of remnant grassland sites across an urban to rural gradient in Los Angeles, CA, USA. Across this gradient we assessed patterns of alpha and beta diversity during the 2019 growing season. We found that local plant alpha diversity in remnant grasslands declined in urban landscapes (measured by surrounding percent development) due mostly to loss of native species. However, at intermediate scales across unmanaged parks and greenspaces, we saw increases in beta diversity at more urban locations. This was possibly due to the patchy dominance of different exotic species at urban locations; whereas, in rural locations non-native and native species were common across plots. Conservation is often informed by examinations of large scale, city-wide assessment of diversity, however, our results show that urban plant diversity, particularly native species, is affected at all spatial scales and beta-diversity can add important insights into how to manage urban ecosystems. Conservation that accounts for alpha and beta

diversity may promote “virtuous cycle” frameworks where the promotion and protection of biodiversity simultaneously reduces the negative effects of invasion.

#### KEYWORDS

urban ecology, alpha diversity ( $\alpha$ ), beta diversity ( $\beta$ ), conservation, California grasslands

## Introduction

Urbanization is the most comprehensive form of land-use alteration, resulting in environments that are radically different from less developed areas (Shochat et al., 2006). According to some models, global urban land cover is set to increase by 78 to 171% by 2050 (Huang et al., 2019). Furthermore, cities are often created in biodiversity hotspots, resulting in profound losses of global vegetation land cover (Myers et al., 2000). A shift from natural to urban land cover can result in myriad changes including an increase in impervious surfaces (Stewart and Oke, 2012), reduced soil water absorption and increased flooding (Scalenghe and Marsan, 2018), increased irradiative heating (urban heat islands; Taha, 2017), increased habitat disturbance (Knapp et al., 2008), and changes in soil characteristics (Pickett et al., 2001; Kowarik, 2011).

Beyond the physical implications of urbanization, past work has indicated that urbanization can have strong biotic effects as well. In particular, urbanization may favor invasive exotic species over native species (McKinney, 2006, 2008; Wania et al., 2006; Avolio et al., 2019), but this effect is not consistent and depends on city-specific conditions (Kowarik, 2011). The mechanisms underlying the relationship between urbanization and exotic invasion are still unclear, in part due to many urban biodiversity studies being conducted in highly managed city parks and greenspaces with relatively few studies occurring in unmanaged, remnant urban locations (Avolio et al., 2019; Knight et al., 2021). Developing a better understanding of these mechanisms should be a conservation priority, as in remnant natural areas there can be a high proportion of plant cover by invasive, exotic species (Avolio et al., 2019). There remains a need for robust biodiversity assessments focused on remnant locations.

Here, we focus on three potential mechanisms for increases in exotic species in urban environments. First, exotic species introductions may be higher in urban areas (McKinney, 2006, 2008). Tait et al. (2005) found that in Adelaide, Australia plant species richness increased by 46% from 1836 to 2002 due to the introduction of exotic plant species outpacing extinctions. Second, urban development modifies natural habitats. This modification often results in the loss of native species with high habitat specialization (Knapp and

Ingolf, 2009). Third, urban environments may be especially stressful with higher temperatures, increased drought, and widespread pollutants (Calfapietra and Pen, 2015; English and Wright, 2021). This may benefit certain invasive exotics that favor these conditions. For example, species that can exploit anthropogenic nitrogen deposition and tolerate higher thresholds of water stress (Valliere et al., 2017). Past work has shown that exotic Mediterranean grasses (primarily *Avena barbata* and several species of *Bromus*) in California may have become particularly invasive due to their ability to tolerate stress and disturbance (D’Antonio et al., 2007). The primary forces that led to these exotic species becoming dominant were better adaptations to drought, intensification of crop agriculture, and the intense year-round grazing pressure that occurred during the 1860s–1880s. Conversely, native species may be more closely adapted to historical conditions and/or less stressful conditions that more closely mirror those found in rural locations.

Exotic species invasions in urban environments are also correlated with both increased and decreased biodiversity. The effect of exotic species appears to depend on the level of urbanization, taxa, biodiversity metric considered (e.g., alpha vs. beta), and other local variables (McKinney, 2006; Schwarz et al., 2017). For example, a moderate level of urbanization (e.g., suburban neighborhoods) may increase the overall number of species (alpha diversity) because exotic species gains outpace native species losses. In fact, past work has shown that plant communities in the transition zone between the urban core and the city outskirts foster the highest levels of diversity (Zerbe et al., 2003). The level of disturbance in the urban core is too high for many plant species to grow (Hahs and McDonnell, 2006; McKinney, 2006) and the rural outskirts experience competitive exclusion from dominant, well-established native species (D’Antonio et al., 2007). Moderately urban areas act as a sort of Goldilocks zone where there is high enough disturbance to disrupt competitive exclusion by dominant species, but not enough disturbance to inhibit the growth of species. Moderately urbanized areas are also often suburban neighborhoods where introductions of exotic species are the highest (due to gardening and horticulture, Kowarik, 1995; McKinney, 2008). Additionally, these urban areas can have high heterogeneity between locations, fostering

different plant assemblages at small scales (McKinney, 2008; Clarke et al., 2013). Consequently, we may see the highest levels of alpha diversity (the total number of species at a site) and beta diversity (species turnover between plots) in these suburban areas.

Alternatively, should dominant exotic species respond positively to urbanization, alpha and beta diversity decline with increasing urbanization. Specifically, if all levels of urbanization favor exotic species and these species tend to dominate and outcompete all others (exotic and native), this could drive both alpha and beta diversity down. For example, Southern California experiences unusually high levels of nitrogen deposition such that annual fluxes of nitric oxide (NO) from high-deposition chaparral and forested areas in the Los Angeles air basin are similar to those of fertilized croplands (Fenn et al., 2003). Valliere et al. (2017) found that in Southern California, higher nitrogen deposition reduced native plant cover and concomitantly increased cover and biomass of non-native annuals. For dominant exotic species that respond positively to these environmental changes, urbanization may increase the rate of invasion, and the invasive species may then outcompete remaining natives and/or less stress-adapted exotics.

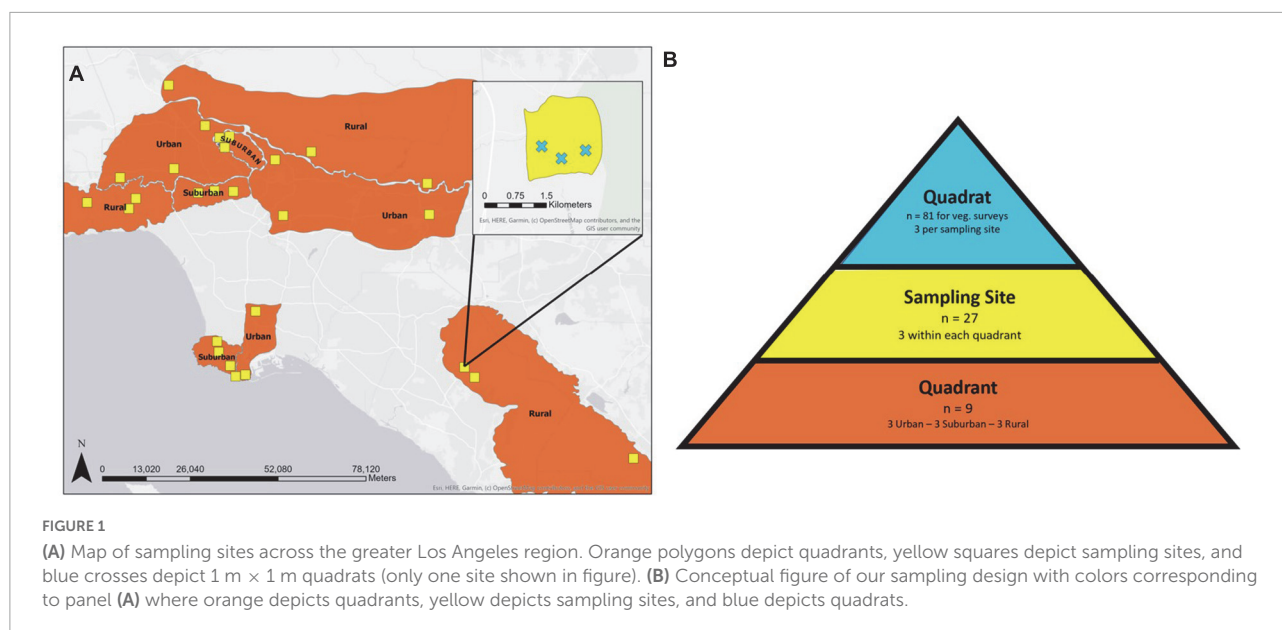
While alpha diversity may increase or decrease with urbanization intensity, beta diversity is likely highest in suburban environments. Suburban environments support the highest habitat heterogeneity because novel urban habitats are interspersed with historical habitat remnants. The combination of habitat types may maintain or increase species richness of both exotic and native species (Pysek, 1993; Pickett et al., 2001). In addition to habitat heterogeneity, patchy local extinctions may drive variation in species composition in urban habitats (Niemelä, 1999; McKinney, 2002). Extinctions occur due to habitat degradation and destruction, causing species with poor dispersal ability to become isolated or have their habitat patch further decreased in size. Consequently, due to low integration between urban patches, there can be high variation in colonization and extinction at sites resulting in high urban beta diversity.

Typically, urban biodiversity studies are conducted on managed properties (Godefroid and Koedam, 2007; Walker et al., 2009; Avolio et al., 2019). This is standard experimental practice considering the constraints of most western cities: urban developers deconstruct landscapes and replace natural areas with tightly managed non-native vegetation to create lawns and other urban landscapes (Walker et al., 2009; Faeth et al., 2011). The result of this is that urban ecosystems often have very little unmanaged land where species recruitment can naturally occur. However, due to its unique geography, Los Angeles may be an exception. There are 886,443 acres of protected public lands in Los Angeles County, 34% of total County land (Gold et al., 2015). While the abundance

of protected area varies with topography, there remains a significant amount of the Los Angeles region that does not have direct human management regimes applied to it and has never been developed. The management of these spaces should be of high concern, as invasion by Mediterranean grasses pose a threat to native species and ecosystems. For example, as of 2018, herbaceous cover represented roughly 31% of the Angeles National Forest (directly adjacent to the city of Los Angeles, CA, USA). This is a high-traffic forest that was historically dominated by native chaparral (Park et al., 2018). A large portion of this invasion is by species in three genera: *Avena*, *Bromus*, and *Brassica*. Species in each of these genera have been shown to reduce the diversity and abundance of native species. Some of these exotic species have a seed bank density an order of magnitude greater than native forbs and shrubs (Cox and Allen, 2008; Abella et al., 2011; Vallejo et al., 2012). Consequently, Los Angeles may be a novel system to examine and develop assessments for natural dynamics of biodiversity along an urban to rural gradient without confounding urbanization with management.

Better understanding the diversity dynamics between native and invasive species across urban gradients has the potential to highlight interventions that could enhance native species (e.g., targeted restoration activities), eventually leading to positive feedbacks for broader promotion of biodiversity and societal benefits (e.g., links with native biodiversity and human health; Dean et al., 2011; Morrison, 2016). While limited, there have been studies linking access to greenspace and access to biodiverse urban locations with the physical, mental, and social health of surrounding communities (Marselle et al., 2021). Given that large, unmanaged grasslands are uncommon in urban areas, the management of native species and biodiversity in these systems could provide novel benefits to the local non-human and human communities.

Here, we assess how exotic species, native species, and overall grassland plant diversity (both alpha and beta) change across an urban-to-rural gradient in Los Angeles, CA, USA. Specifically, we had two objectives: to investigate (a) how urbanization affects different plant diversity metrics and (b) differences in how native and exotic species respond to urbanization. To address these, we established a network of unmanaged grasslands distributed across the greater Los Angeles area. We used this network of grassland patches situated within open spaces across an urban to rural gradient to test the following hypotheses: (H1) native plant species decline and exotic species increase with increasing urbanization, (H2) alpha diversity (Shannon's index) of plants peaks in moderately developed areas (due to introduction of exotic species outpacing the exclusion of natives) but declines in our most developed areas, and (H3) beta diversity of plants peaks in moderately developed areas due to novel urban conditions increasing the recruitment of novel exotic species in combination with the local extinction of native species.



## Materials and methods

### Study area

The study area encompassed the greater Los Angeles area, covering over 10,000 km<sup>2</sup> ranging from the San Gabriel mountain range to the Santa Ana mountain range to the south, and the Los Angeles county barrier to the east (Figure 1A). All locations exist within a Mediterranean climate associated with moderately wet winters with cool temperatures and dry summers with high temperatures (Gómez et al., 2004). Precipitation during the growing season (November–April) from 1969 to 2018 averaged 614.68 mm and ranged between 167.64 and 1,513.84 mm (PRISM Climate Group, Oregon State University). Mean surface temperature over the same period was 6.9°C and ranged from 4.6 to 8.4°C.

### Site selection

Potential field locations were identified using ArcMap (Version 10.5) where we selected within a range of a 5–30% slope,  $\leq 1,200$  m elevation,  $\geq 1,400$  m<sup>2</sup> patch size, and south-facing aspect within Los Angeles County, Orange County, and the western 1/3 of Riverside County. In order to sample across urbanization levels, we identified 16 quadrants throughout the greater Los Angeles region based on neighborhood boundaries. Each of these quadrants was classified based on percent impervious surfaces within the neighborhood polygon. This resulted in a range from rural or undeveloped areas with less than 20% impervious surface area, moderate levels of urbanization ranging from 20 to 50% impervious surface area,

to “hardscape” areas with over 50% impervious surface cover. These “hardscaped” areas are highly fragmented and thus we do not have field sites with higher than 66% surrounding impervious surface.

We then randomly chose nine quadrants from the 16 identified: three quadrants were in rural areas, three were in suburban areas, and three were in urban areas. Within each of these quadrants, all green spaces larger than 100 square meters were identified, delineated using GIS, and three local sampling sites were randomly selected from these. Sampling sites were then ground-truthed to confirm that they were unmanaged and unmaintained grasslands. If a site was actively managed by the community or a designated land manager (e.g., mowing or native species planting), it was removed from the dataset and replaced with either a nearby site (first choice) or a newly randomly selected site (if a nearby site was unavailable). This resulted in a total of 27 locations. For each site we also measured elevation and distance to nearest coastline as two other drivers of community composition and diversity along our urban gradient.

### Plant diversity survey

In April 2019, we determined species identity and quantified the abundance of all plant species at each of our sites. At each of the 27 locations, three 1 m x 1 m quadrats were selected (for a total of 81 plots across the gradient) using the generate random points tool for our site layer in ArcMaps. In each of these quadrats all plants in all taxa were identified to the species level (Supplementary Table 1). In order to estimate abundance, we visually determined percent cover of each individual species in the plot using a plot grid to increase the accuracy of our estimates. At each of the three plots within a location, we



TABLE 1 Model selection for what abiotic factors best fit alpha diversity.

Model	K	AICc	Delta_AICc	AICcWt	Cum_Wt	Res_LL
Urban development%, NO3	6	84.44	0.00	0.93	0.93	−35.54
Urban development%, NH4	6	91.05	6.61	0.03	0.96	−38.85
Urban development%	5	91.41	6.98	0.03	0.99	−40.31
Urban development%, NH4, NO3	7	93.17	8.73	0.01	0.99	−38.67
Urban development%, distance to coast, NO3	7	94.17	9.73	0.01	1.00	−39.17
Urban development%, distance to coast, NH4	7	101.31	16.87	0.00	1.00	−42.74
PercDev, distance to coast	5	101.60	17.16	0.00	1.00	−44.23
PercDev, distance to coast, NH4, NO3	8	103.22	18.78	0.00	1.00	−42.41

Factors included were surrounding percent development (Urban development%), distance to nearest coast (distance to coast), nitrate (NO3), and ammonium (NH4).

used our species abundance and evenness data to determine plot diversity using the Shannon diversity index (Shannon and Weaver, 1964). We also collected soil cores from a depth of 1 ft with a 1-inch diameter. Soil cores were stored in a freezer after collection and were sent to the UC Davis Analytical Lab in August 2021. Samples were analyzed for nitrogen content in the form of nitrate (NO3) and ammonium (NH4).

## Data analysis

We used a model selection approach to assess the best-fitting model for what was driving alpha diversity (Supplementary Equation 1a) and ranked the candidate models with AIC to determine the best-fitting model. This model selection was conducted to account for other environmental factors that may affect the diversity in our plots. We fit linear mixed effects models that included continuous fixed effects of percent development (impervious surface area in a 2 km buffer around each site), elevation (Molina-Venegas et al., 2016), distance from coast (Stromberg et al., 2001), NO3, and NH4 (Valliere et al., 2017) for all of our 27 sites. We included these additional effects to account for other drivers of diversity in Mediterranean grasslands. Additionally, due to our nested design, we included site location nested within quadrant as a random effect given the spatial blocking of our study (plots at each site will have similar conditions to one another and sites located within the same quadrants of the city will have similar conditions; Table 1). Our sites are co-located along the rural to urban gradient because unmanaged grasslands are spatially clumped within this gradient. Because of this design, there is spatial autocorrelation in our sites that is inherently related to the gradient that we are examining. To control for this to the best of our ability, we added blocking variables for the site to the random effect structure of our model. This cannot completely control for spatial autocorrelation but does control for severe autocorrelation within sites. Since percent development and elevation were correlated variables (Supplementary Figure 1), we removed elevation from subsequent analyses while keeping percent development due to the nature of our hypotheses.

TABLE 2 Three most common exotic species.

Species	Presence across all quadrats	Average percent cover
<i>A. barbata</i>	69.1%	16.6%
<i>B. diandrus</i>	76.5%	24.2%
<i>B. reubens</i>	34.6%	16.3%

For each species, the percentage of quadrats it was found in and the average percent cover of the quadrats it was present in is listed.

To address our first hypothesis, we calculated Spearman's correlation coefficient on total, native, and the three most common species (Table 2, and Supplementary Equation 1c) against the continuous measure of percent development around each site. These three species (*A. barbata*, *B. diandrus*, *B. reubens*) are invasive exotics known to be detrimental to local ecosystems and were the most abundant across our plots. We wanted to determine if development might be enabling their dominance. Additionally, we ran mixed-effects ANCOVAs using the same random effects structure as above to account for spatial blocking of our study design. We assessed the correlation between our three most dominant invasive species (above) and alpha and beta diversity in our plots.

To address our second and third hypotheses, we analyzed the effects of urbanization on species diversity and abundance, as well as the presence of native and exotic species. We assessed the effects of all fixed and random effects on alpha diversity (Shannon diversity index), beta diversity among the three quadrats at each site (betapart Package in R, version 1.5.1; Orme, 2012), number of native species, and number of exotic species.

## Results

### Effects of urban gradient

We found a total of 52 species across all grasslands. At each site there were  $7.63 \pm 2.66$  total species,  $6.67 \pm 1.96$

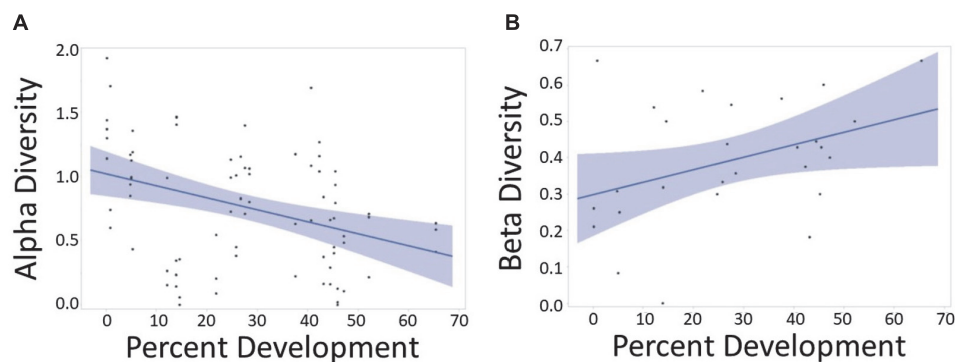


FIGURE 2

Percent development around a 2 km radius of field sites against alpha and beta diversity. **(A)** Plot showing alpha diversity where each data point represents Shannon's diversity at one of the three quadrats at each site. As surrounding percent development increased, alpha diversity decreased [ $F_{(1,12)} = 7.24$ ,  $p = 0.020$ ]. **(B)** Plot showing beta diversity where data points reflect beta diversity across each site. As percent development increased, beta diversity increased [ $F_{(1,12)} = 4.50$ ,  $p = 0.055$ ].

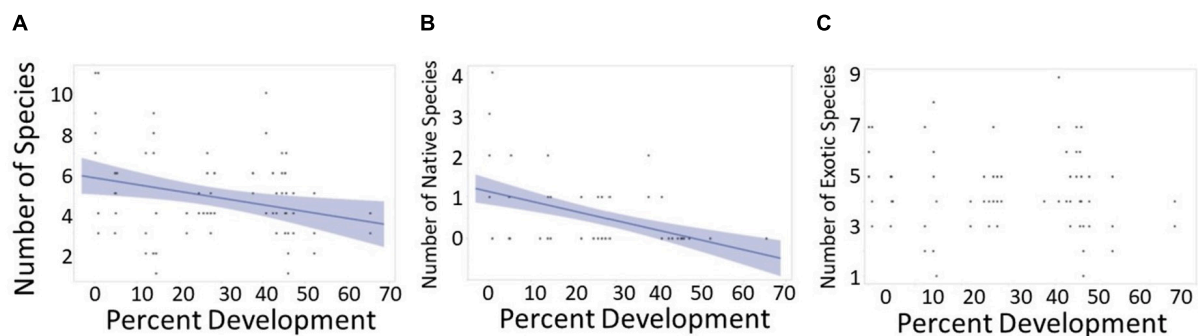


FIGURE 3

The effect of number of overall species, native species, and most common exotics vs. percent development. **(A)** Percent development had an effect on the overall number of species at each quadrat [ $F_{(1,12)} = 6.05$ ,  $p = 0.030$ ]. Spearman's correlation coefficient showed this to be a weak negative effect ( $r_s = -0.28$ ,  $n = 81$ ,  $p = 0.01$ ). **(B)** Percent development had an effect on native species [ $F_{(1,12)} = 8.70$ ,  $p = 0.012$ ] which Spearman's correlation coefficient found to be a moderately strong negative monotonic correlation ( $r_s = -0.47$ ,  $n = 81$ ,  $p < 0.001$ ). **(C)** Percent development did not have an effect on exotic species [ $F_{(1,12)} = 3.15$ ,  $p = 0.10$ ].

exotic species, and  $1 \pm 1.32$  native species. Our best-fit model showed that the combination of percent development and  $\text{NO}_3$  best explained alpha diversity (Table 1 and Supplementary Equation 1b). This model was used in all subsequent analyses. We found that percent development had a negative effect on alpha diversity [ $F_{(1,12)} = 7.24$ ,  $p = 0.02$ ] and a positive effect on beta diversity [ $F_{(1,12)} = 4.50$ ,  $p = 0.055$ ] across our urban gradient (Figure 2).

The overall number of species in each quadrat was negatively affected by percent development [Figure 3A,  $F_{(1,12)} = 6.05$ ,  $p = 0.03$ ]. Native species were negatively affected by percent development [Figure 3B,  $F_{(1,12)} = 8.70$ ,  $p = 0.012$ ] whereas exotic species were not affected [Figure 3C,  $F_{(1,12)} = 3.15$ ,  $p = 0.10$ ]. There were no other significant relationships between abiotic factors and community composition.

## Species-specific responses to urban gradient

The three most abundant species were all exotic annual grasses (Table 2). The next most abundant species was the exotic annual herb *Brassica nigra* which covered an average of 1.4% of all quadrats. Two exotic grasses (*B. diandrus* and *B. reubens*) were correlated with decreased alpha diversity of the plots (Supplementary Figure 2). None of the invasive exotic grasses were correlated with beta diversity.

## Discussion

We found that plant diversity in unmanaged grasslands in Southern California was affected by urbanization. In addressing

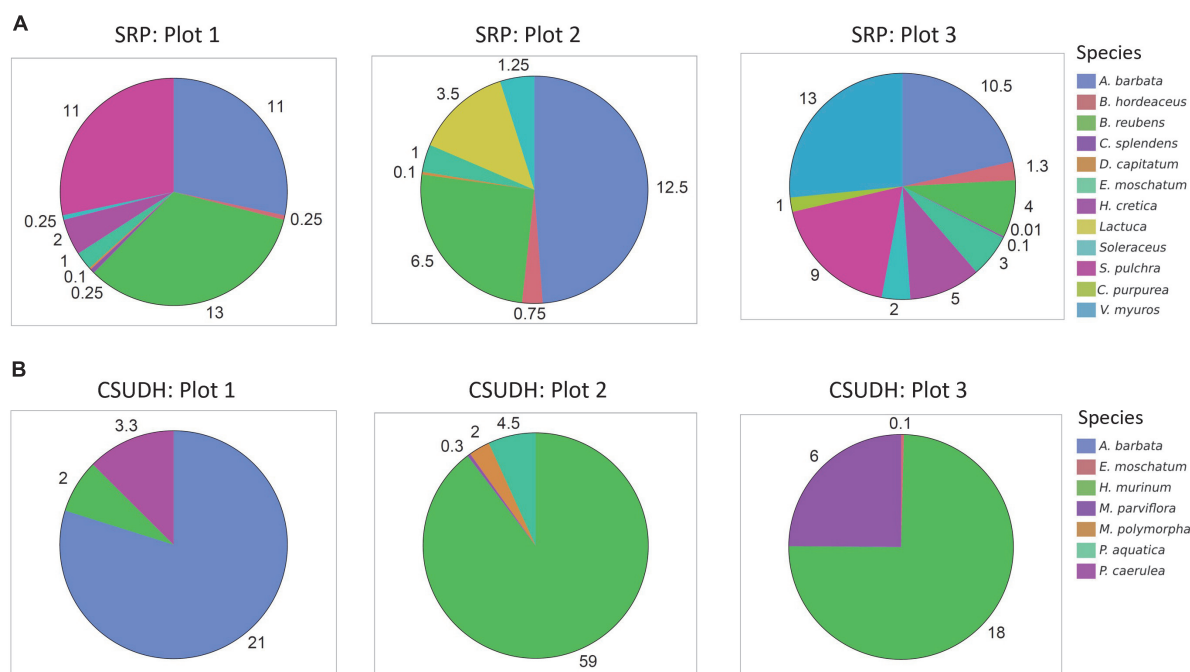


FIGURE 4

Pie charts showing the species distributions at each quadrat at two of our field sites. (A) Percent abundance of present species at our least developed site, the Santa Rosa Plateau Ecological Reserve. At these plots there was relatively high diversity, but nearly identical species in each of the three quadrats. (B) Percent abundance of present species at our most developed site, California State University– Dominguez Hills. At these plots there were no native species present, with different exotic species dominating quadrats.

our first hypothesis concerning how native and exotic plant species respond to urbanization, consistent with previous literature (Avolio et al., 2019), we saw a reduction in native species with increasing levels of development. However, we did not see an increase in exotic species with increasing development in our unmanaged site network. Instead, we saw that exotic species were equally well represented in Southern Californian grasslands at both low and high levels of development. This resulted in an overall reduction in alpha diversity across our gradient as only native species were lost in the more urban sites with no new exotic species appearing in more urban sites, contrary to our expectations (H2). Lastly, our prediction that beta diversity would be highest in suburban areas was moderately supported; we found the highest levels of beta diversity in our most urban sites, though these sites were still more suburban than what past studies likely deemed the urban core (McKinney, 2008).

Contrary to previous studies of plant diversity across managed urban gradients (McKinney, 2008), we did not find that species diversity peaked in moderately developed areas. Comparing our most rural sites to our most urban, there was an average loss of 2.5 species at the site level constituting a 41.6% loss of diversity in our most urban plots. In contrast to other studies of managed urban greenspaces (Walker et al., 2009; La Sorte et al., 2014), remnant grasslands in our study

did not appear to have any locally available horticultural species colonizing them (Supplementary Table 1). Additionally, our results differ from studies of urban plant biodiversity in coastal sage scrub communities that show remnant natural areas hosting a majority of native species compared to exotic invasive species (Avolio et al., 2019).

Unmanaged grassland communities across the Los Angeles area consequently appear to be an ecosystem particularly affected by exotic invasion and biotic homogenization. Consequently, Los Angeles offers a unique opportunity to assess the effect of urbanization on ecological assembly in an unmanaged context. Our sites were not specifically managed in any way. Instead, these grasslands are likely undergoing succession just like their rural counterparts, the only difference being the environmental gradients that result from urbanization (e.g., nitrogen, temperature, and habitat fragmentation). This may contribute to the homogenization of urban locations, as exotic species in California grasslands can be very successful at expanding into ranges where resource limitation is alleviated (e.g., urban areas that experience fertilizer runoff and increased nitrogen deposition; Bettez and Groffman, 2013; Eskelinen and Harrison, 2015). The success of exotic species under these urban conditions consequently leads to native species being limited to marginal habitats. Future restoration in these systems should prioritize soil recovery and revegetation to

facilitate establishment of native plant species (Beltran et al., 2014). Additionally, this may be evidence of “extinction debt” common to urban areas where native species go locally extinct over long time periods due to disturbance (Hahs et al., 2009; du Toit et al., 2016). The effects of declining habitat connectivity in similar semi-natural grassland diversity have been realized after 50–100 years (Lindborg and Eriksson, 2004), similar to the surrounding landscape history of our field sites.

One other likely reason for the discrepancy between our results and previous studies is that our network of sites included locations with a maximum of 66% development. Thus, our study does not examine the true “urban core” the way past urban biodiversity surveys have (McKinney, 2008; **Supplementary Figure 3**). We are consequently not capturing truly hardscaped areas given that past studies have included sites up to 95% development (Yan et al., 2019). Furthermore, past surveys of urban diversity have often used qualitative metrics for developments and lack a quantitative measure of development around sites (Hope et al., 2008; Walker et al., 2009; Avolio et al., 2019). The differences in our results may result from differences in how urbanization levels are defined. Additionally, because our unmanaged grasslands are clumped along the rural to urban gradient, there is likely some spatial autocorrelation in our data that we are unable to address. We do not, however, believe that this spatial autocorrelation is likely to alter our results.

We found that beta diversity steadily increased with increasing urbanization across our gradient. This might be driven by patchy extinctions of subordinate species coupled with dominant exotics that dominate across different landscape settings. For example, our most urbanized site was California State University- Dominguez Hills located in the city of Carson. This site is on the campus of a highly developed public university and is surrounded by 65.4% development. At this location, we identified seven species, none of which were native species. However, each of the three plots at this site were dominated by a different exotic species and the subordinate species in this community differed from one quadrat to another, leading to greater beta diversity (**Figure 4**). Conversely our least developed site was the Santa Rosa Plateau Ecological Preserve in the city of Murrieta. The Santa Rosa Plateau is a preserved area surrounded by 0.2% development. We identified 12 species where 4 were native to the region. There was relatively high diversity in each plot, but also nearly identical species in each of the three quadrats.

While we did not survey the most highly developed areas (e.g., >66% impervious surfaces), our findings are consistent with research showing that moderately developed areas often foster higher levels of beta diversity (Rebele, 1994; La Sorte et al., 2014). Urban landscapes have a large variety of habitat types and ecological communities associated to each (Norton et al., 2016). Further, previous work has shown that rare native species can go locally extinct with urbanization (Kühn and Klotz, 2006) compared to exotic species that show lower levels

of turnover (La Sorte et al., 2014). Across our urban gradient we may see an increase in beta diversity as we move from higher levels of species, namely native species, to more urban areas where these species become rare and are thus more likely to go locally extinct. Importantly, our results suggest that fine-scale surveys of beta diversity patterns are essential to our understanding of larger scale patterns of diversity and how to conserve regional diversity. Examinations of urban beta diversity have the potential to spatially inform conservation practices such as protected area selection and where corridor and dispersal facilitation could be beneficial (Socolar et al., 2016).

Future examinations of plant diversity could benefit from including more comprehensive biodiversity metrics not just limited to the inclusion of beta diversity. The inclusion of multiple metrics may allow for more comprehensive and interesting investigations into how different diversity metrics interact with one another. For example, assessments of functional diversity can provide unique insights into ecosystem stability as a supplement to phylogenetic diversity. This can be beneficial as phylogenetic diversity alone may be an imprecise proxy for assessing the functional diversity of urban plant communities (Lososová et al., 2016). In turn, the use of these approaches can aid land managers in supporting conservation efforts following “virtuous cycle” frameworks where the promotion and protection of biodiversity could simultaneously reduce the negative effects of invasion.

We believe the greater Los Angeles area would be an ideal location for these future studies given the unique and interwoven availability of this system. While managed grasslands in the form of yards and parks are more ubiquitous across many urban areas, large unmanaged urban grasslands remain novel. Promoting biodiversity in these areas should be a potential conservation priority given the unique potential they have to provide local communities with a generally inaccessible ecosystem type. Having native species more widely accessible to communities may in turn increase attention to preserving these species, creating a cycle where attention increases availability.

## Conclusion

Across remnant grasslands in Los Angeles, our data show that alpha diversity is decreasing across a rural to urban gradient, driven by losses in native species. However, possibly due to stochastic local extinctions, beta diversity in our most urban sites was higher than nearby rural areas. While we can only speculate on the mechanism, native species were negatively affected along our development gradient while the number of exotic species remained constant. Future conservation at these urban locations should prioritize proven restoration efforts such as soil recovery and revegetation to promote native species. Transitioning into more developed areas across our



gradient, exotic species were not introduced into these systems in higher proportions but rather those already present appeared to displace native species. We suggest that conservation efforts should utilize multiple biodiversity metrics, including beta diversity, to aid our understanding of how biodiversity patterns operate at different scales and supporting efforts that utilize “virtuous cycle” conservation frameworks.

## Data availability statement

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

## Author contributions

AW conceived the experiment and provided partial funding. JE and AW established the experimental design. JE collected the data, analyzed the data, and wrote the manuscript. KB and EW advised on data analysis. JE, AW, KB, and EW contributed to manuscript writing. All authors contributed to the article and approved the submitted version.

## Funding

This work was supported by a NOAA CIMEC Education Outreach Grant (ID #231688) as well as startup funds provided by California State University, Los Angeles awarded to Alexandra Wright.

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## Acknowledgments

We thank Brian Hsieh, Daniel Guzman, Bryan Miranda and Miles Kilpatrick for field assistance and data collection. Thanks to Beatriz Aguirre, Regina Mae Francia, and Sam Watson for the support. We thank Steve LaDochy for their comments on the manuscript. We also thank Justin Valliere for help with species identification.

## Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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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.2022.921472/full#supplementary-material>

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EDITED BY  
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United States

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Hao Zhang,  
Fudan University, China  
Alicia Coleman,  
University of Connecticut,  
United States

\*CORRESPONDENCE  
Michael L. Treglia  
michael.treglia@tnc.org

SPECIALTY SECTION  
This article was submitted to  
Urban Greening,  
a section of the journal  
Frontiers in Sustainable Cities

RECEIVED 15 May 2022  
ACCEPTED 19 October 2022  
PUBLISHED 23 November 2022

CITATION  
Treglia ML, Piland NC, Leu K, Van  
Slooten A and Maxwell EN (2022)  
Understanding opportunities for urban  
forest expansion to inform goals:  
Working toward a virtuous cycle in  
New York City.  
*Front. Sustain. Cities* 4:944823.  
doi: 10.3389/frsc.2022.944823

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# Understanding opportunities for urban forest expansion to inform goals: Working toward a virtuous cycle in New York City

Michael L. Treglia<sup>1\*</sup>, Natalia C. Piland<sup>1</sup>, Karen Leu<sup>2</sup>,  
Alaina Van Slooten<sup>1</sup> and Emily Nobel Maxwell<sup>1</sup>

<sup>1</sup>New York State Cities Program, The Nature Conservancy, New York, NY, United States, <sup>2</sup>The Nature Conservancy, Cold Spring Harbor, NY, United States

Urban forests are critical infrastructure for mitigating environmental and social challenges cities face. Municipalities and non-governmental entities, among others, often set goals (e.g., tree planting or canopy targets) to support urban forests and their benefits. We develop the conceptual underpinnings for an analysis of where additional canopy can fit within the landscape, while considering factors that influence where trees can be planted, and canopy can grow ("practical canopy"). We apply this in New York City (NYC) to inform the setting of a canopy goal by the NYC Urban Forest Task Force (UFTF) for the *NYC Urban Forest Agenda*, which may trigger a *virtuous cycle*, or a positive feedback loop where people are mobilized to protect the urban forest, and its benefits that ultimately motivate people to commit to its conservation. We further develop framing for a "priority canopy" analysis to understand where urban forest expansion should be prioritized given more context (e.g., environmental hazards and local preferences), which can inform how expansion of the urban forest is achieved. We estimate an opportunity for 15,899 ha of new canopy in NYC given existing opportunities and constraints (practical canopy), which, if leveraged, could result in nearly doubling the canopy as of 2017 (17,253 ha). However, like existing canopy, practical canopy is not evenly distributed, in general, or across jurisdictions and land uses. Relying solely on areas identified as practical canopy to expand the urban forest would exacerbate these inequities. We discuss how the NYC UFTF established a visionary and achievable goal of at least 30% canopy cover by 2035, informed by this analysis and guided by priorities of equity, health, and resilience. Achievement of this goal will ultimately require a combination of protecting and stewarding the existing resource, and leveraging opportunities for tree planting. Achieving a more equitable urban forest will also require identification of priority canopy, and, in cases, creation of new opportunities for tree planting and canopy expansion. Overall, the collaborative establishment of such goals based on local context can be instrumental in creating a *virtuous cycle*, moving conservation actors toward exercising influence and agency within the social–ecological system.

## KEYWORDS

tree canopy goal, urban conservation, urban forest equity, urban forest goals, social ecological system, urban tree canopy, tree equity, sustainability planning

## Introduction

Urban forests are complex systems that include all trees in a city and the physical and social infrastructure on which they depend (adapted from Robertson and Mason, 2016). They also serve as critical infrastructure for mitigating various social and environmental challenges cities face. For example, urban forests help reduce the urban heat island effect (Alonzo et al., 2021), they support management of stormwater runoff (Selbig et al., 2022), and they are both comprised of and are habitat for various animal and plant species (Derby Lewis et al., 2019). Furthermore, benefits of urban forests including air quality improvement (Lai and Kontokosta, 2019), carbon sequestration (Nowak et al., 2013; Pregitzer et al., 2022), community cohesion (Campbell et al., 2016; Svendsen et al., 2016), and mental wellbeing (Berman et al., 2021), among others, are increasingly demonstrated and understood. Despite the increasing recognition of the roles that urban forests play, recent work indicates they are declining throughout the United States (Nowak and Greenfield, 2018). However, intentional planning for and maintenance of urban forests can help sustain and expand them through the long term (Dwyer et al., 2003).

As Morrison (2015, 2016) has described, targeted planning for conservation of a resource, with engagement of stakeholders and explicit consideration of people as part of a social–ecological system, can spur a positive feedback loop in which benefits of conservation outcomes beget more sustained conservation. This is described as the *virtuous cycle* framework, with the positive feedback loop itself being the eponymous “virtuous cycle” (Morrison, 2015, 2016). Assumptions of the framework are as follows: there is an objective (e.g., of a conservation organization) to protect an aspect of nature; people are integral to any conservation outcome; conservation needs to be incorporated into the landscape, rather than relying on relegating specific areas for conservation (e.g., of “wild nature,” *sensu* Morrison, 2015); conservation solutions are more durable when they tend to be made more mainstream and solutions can be made self-sustaining; and, while work focuses in certain places, it is important to strive to effect change more broadly. Ultimately, the virtuous cycle framework is intended to leverage theories of change, or hypotheses of how planning with people will benefit all nature (including people) in ways that will garner broader support for the focal resources. The framework can apply to urban forests, supporting the incorporation of human dimensions into their resource planning—a key need, previously identified by Dwyer et al. (2003).

Municipalities, non-governmental entities, stewardship or conservation organizations, and collaborative groups or coalitions sometimes support planning and maintenance of urban forests by setting goals to maintain or expand them and their benefits. These goals are often set within one of two frames—as tree planting targets, through which a number of

new individual trees is set for planting, or tree canopy cover targets, which aim to increase the cumulative land area covered by leaves and branches of trees (McPherson and Young, 2010). While tree planting goals can be galvanizing, particularly shortly after they are established (Eisenman et al., 2021), they alone do not account for factors such as ongoing loss or removal of trees, or for the ongoing management needs of existing trees that support canopy expansion through time. They functionally only consider one element of a dynamic system and may not, in and of themselves, capture net effects of overall management of the urban forest (McPherson and Young, 2010). Achieving and maintaining a specific canopy cover ultimately requires holistic management of the urban forest that considers the life cycle of trees, including tree protection and care, in addition to planting (e.g., see the Chicago Region Tree Initiative 2050 Master Plan; Morton Arboretum, 2018). Furthermore, benefits of individual trees may be difficult to holistically track (depending on species, size, local context, and other factors), particularly while accounting for trees removed, while benefits can be calculated based on canopy cover, as with urban heat amelioration (Ziter et al., 2019) and stormwater management associated with interception of precipitation (Hirabayashi, 2015). Given these considerations, we focus on urban forestry goals for canopy rather than tree planting targets.

It is important that canopy goals respond to local constraints and opportunities to realize desired benefits. For example, factors such as residents’ demand for or interest in trees and their benefits, soil conditions, and availability of resources for maintenance can play important roles. This insight was gleaned from experience of urban foresters, researchers, and community members and informed a transition by American Forests (a leading urban forestry organization) away from a universal recommendation of 40% canopy cover in cities (Leahy, 2017). The updated guidance came after more nuanced methodologies and processes to set canopy goals had been developed, including the “Three P’s” (Raciti et al., 2006): (1) the “possible canopy,” which answers the question, “Where is it biophysically feasible to plant trees?”; (2) the “potential canopy,” which answers, “Where is it economically likely to plant trees?”; and, (3) the “preferable canopy” which answers, “Where is it socially desirable to plant trees?” Answering the questions embedded within the three P’s, as well as identifying where trees already are, can support the community of people and organizations that plan for and manage the urban forest (Raciti et al., 2006). The concept of “possible canopy” has been applied in myriad municipalities (often cities and broader counties) including in New York City (NYC), New York (Grove et al., 2006; O’Neil-Dunne, 2012); Philadelphia, Pennsylvania (O’Neil-Dunne, 2011, 2019); and Charlotte and Mecklenburg County, North Carolina (O’Neil-Dunne, 2014). There are important examples of advancing beyond that, toward “preferable canopy” and prioritization schemes for new canopy (Locke et al., 2010, 2013), though efforts



to refine mapping of where new canopy can go, and grounding prioritization in more localized needs, have been limited.

A combination of the natural history and landscape context of cities, and the historic priorities and decisions of institutions and communities of people affecting land use, have contributed to the current urban forest in a given city (Roman et al., 2018). In particular, the natural history of a city has implications for the characteristics of the urban forest that the city might strive for. For example, in Phoenix, the vision for its urban forest is one that “reflects and preserves the beauty of the Sonoran Desert,” focusing on local species, such as palo verde (*Parkinsonia florida*), ironwood (*Olneya tesota*), and mesquite (*Prosopis* spp.), with a 25% tree canopy cover goal by 2030 (City of Phoenix, 2009). In contrast, in subtropical, humid Louisville, Kentucky, a goal of 45% canopy cover was set to aggressively combat trends of tree loss and ongoing risks, particularly for ash trees (*Fraxinus* spp.), identified in local research efforts (Louisville-Jefferson County Metro Government, 2015). In some cases, local stakeholders may also decide areas are not appropriate for urban forestry because of their natural history. For example, in NYC, the master plan for the reclamation of the Fresh Kills Landfill ultimately prioritized restoring tidal marshes to the area (Field Operations, 2006).

While natural history provides a lens for ecological opportunities and constraints, decisions about a city landscape are ultimately influenced and made by people and institutions with varying priorities and levels of both direct and indirect influence. The distribution of tree canopy thus often reflects legacies of historic policy, land use, and sometimes socially exclusionary efforts, which had influence on the urban forest. For example, in United States cities, tree canopy is often less prevalent in areas that were historically the subject of discriminatory lending practices, such as “redlining,” which codified neighborhood demographic make-up as a determinant for default risk on property loans (Locke et al., 2021). The result of redlining was systemic disinvestment in immigrant (particularly Mexican, Jewish, and Asian), poor, and, especially, Black (including Black Latinx) neighborhoods, as residents were less able to attain loans and mortgages from banks (Woods, 2012). Furthermore, in many areas, it was common to add racially restrictive covenants in property deeds that prohibited the sale of homes to people of color (Nardone et al., 2021). Thus, people of color have had limits, beyond economic, in where they can purchase property, sometimes keeping them in the redlined areas that not only tend to have less tree canopy (Locke et al., 2021), but also have less vegetation overall (Namin et al., 2020), and are significantly hotter (Hoffman et al., 2020). Variation in conditions within a city can also be associated with zoning and land use (e.g., see Maantay, 2002, 2007) and highlights the need for place-specific investigation of social and development histories that have shaped the current landscape. For example, in NYC, while there is lower tree canopy cover in redlined areas in four out of the five boroughs, there is no discernable trend

in Manhattan, where lower tree canopy tends to be associated with higher incomes (Treglia et al., 2021a). Such variation may be the result of varying development histories across the five boroughs, as Manhattan is historically more densely developed as a whole and there is not much variation in tree canopy across most parts of the borough. Nonetheless, benefits from an expanded urban forest can have the greatest positive impact in neighborhoods with socially vulnerable residents (Zhou et al., 2021). Such expansion of the urban forest can be driven by current priorities, but aspects of it may be influenced by historic factors that set forth constraints in the contemporary landscape, such as where there is pavement, underground utilities, and land uses or built features that may conflict with trees, their roots, or their canopy.

Understanding natural and social context can help guide setting and implementation of urban forestry goals, and engagement with stakeholders in the process can set off a virtuous cycle. In support of that, we developed the concept of “practical canopy,” a data-based analysis that identifies where new canopy can likely fit within a given landscape, to inform setting of tree canopy goals while accounting for local context—particularly factors that affect where trees may be planted and where canopy can grow given real world constraints. We also propose a subsequent step, mapping of “priority canopy.” This step goes beyond the question of what opportunities *currently* exist to develop a better understanding of where expansion of the urban forest is locally desired or needed, which can indicate, in some cases, that landscape change is required to achieve these priorities. We build on existing approaches, incorporating elements from all “Three P’s” (Grove et al., 2006). We then describe our effort to map practical canopy in NYC to support development of a canopy cover goal by the collaborative stakeholder group, the NYC Urban Forest Task Force (UFTF), for inclusion in the NYC Urban Forest Agenda (NYC Urban Forest Task Force, 2021). In the past, while at least one canopy goal had been proposed, 30% by 2030 (from 2006) based on analysis of “possible canopy” (Grove et al., 2006), a tree planting goal (of one million trees within 10 years) was ultimately adopted as part of a mayoral initiative, PlaNYC (Campbell, 2017). The mapped practical canopy is not intended to be prescriptive of where trees should be planted or canopy should be added, or how a canopy goal should be achieved. Instead, it is one step in creating a *virtuous cycle* (Morrison, 2016), wherein ongoing work toward implementation and achievement of the goal can spur further interest and ultimately conservation of the urban forest. The development and results of the practical canopy analysis engaged stakeholders directly by providing information asked for in the process of setting a tree canopy goal, and moving the NYC UFTF toward exercising agency in the social–ecological system by requiring explicit articulation of values and objectives (particularly priorities of equity, health, and resilience). We suggest this virtuous cycle can begin with the engagement of stakeholders in setting an urban forest goal, with

buy-in developed through conversations built, in part, on data and analysis. It can then be reinforced as the goal and supporting information become socialized, with broader support developed as the benefits of the urban forest are more fully realized.

In mapping practical canopy, we sought to answer the following: (1) *How much opportunity for additional tree canopy do we estimate exists in the current NYC landscape?* (2) *How does this vary by geographic scale, jurisdiction, and land use?* and (3) *How does the practical canopy compare to existing and “possible” canopy (sensu Grove et al., 2006)?* Furthermore, we describe how this information supported discussions about potential to expand the urban forest in ways that address existing inequities, a priority identified by the NYC UFTF, which led to their setting a goal of at least 30% tree canopy cover by 2035 for NYC as part of the *NYC Urban Forest Agenda*. The hope is this process has set forth a virtuous cycle that continuously brings in more actors—including policymakers and those immediately affected by the resource—who strive to maintain and expand the urban forest across temporal and spatial scales for its intrinsic value and its benefits, and ultimately the sustenance of a self-supporting social–ecological system.

## Methods

### General definitions and process of mapping practical canopy

We define practical canopy as the spaces or areas within a landscape where it is estimated that new tree canopy can be grown from newly planted trees (or potentially existing ones), while accounting for constraints associated with land use, land cover, and built infrastructure. Mapping practical canopy assumes such constraints are static (i.e., unchanging in the foreseeable future), with analysis based on spatial data (raster or vector) that represent the landscape at a point in time or under different scenarios (e.g., with future development scenarios modeled). Furthermore, it requires those involved in the work (e.g., researchers, managers, and advocates) to make assumptions or decisions about how features on the landscape can functionally constrain planting of new trees and expansion of canopy (e.g., athletic fields would generally be considered as having a conflicting land use, and tall buildings could physically limit where tree canopy can grow). It is ultimately intended to offer insight into how much new canopy a landscape may accommodate in its current form. Mapping of practical canopy is not intended to be prescriptive in terms of where new canopy should be added, as it is a spatial model that does not necessarily resolve conflicting values, or incorporate local perspectives, all constraints at play, and the potential to change the landscape in ways that can create new opportunities for canopy or tree planting (by, e.g., de-paving land). However, it can support conversations about these factors.

Mapping practical canopy entails three general steps that rely on spatial data for the focal area and assumptions for where new trees can be planted and where canopy could exist in the spatial model (termed “allowability” for planting and canopy; Figure 1).

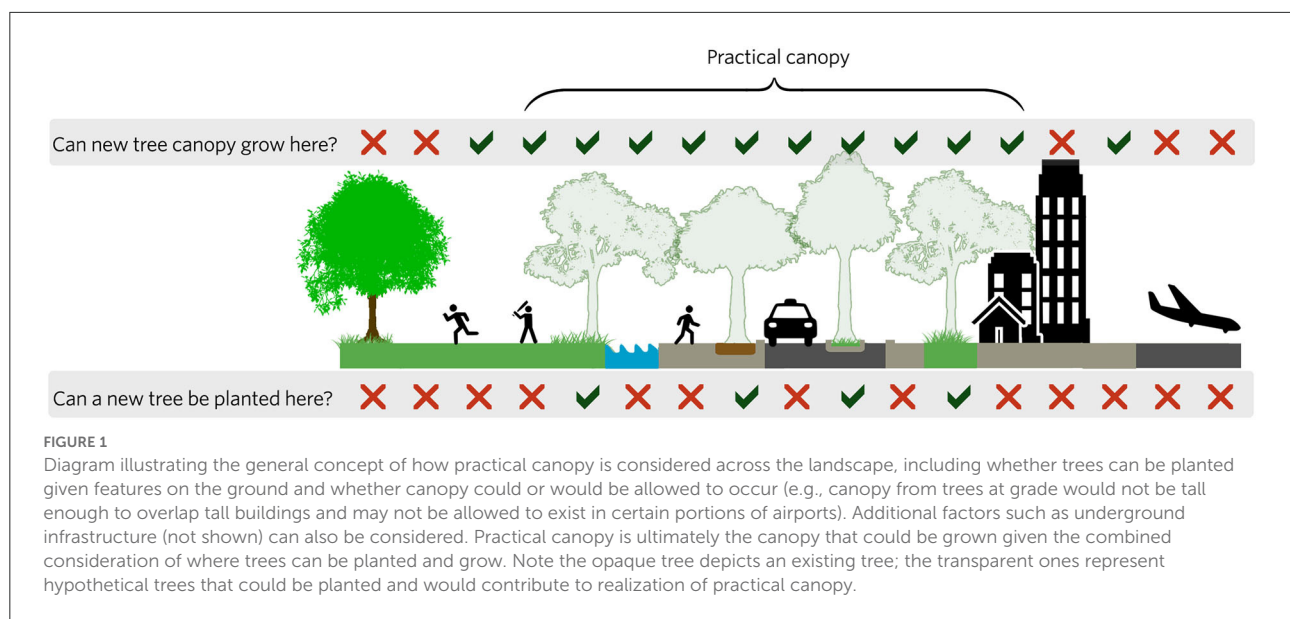
1. **Delineate *planting allowability*, or where within the landscape trees can likely be planted.** This involves developing assumptions of what types of land use and land cover are suitable for tree planting and applying them to relevant spatial data (it is then assumed that canopy could cover these spaces).
2. **Delineate *canopy allowability*, or where within the landscape tree canopy could likely exist.** This involves developing assumptions of where tree canopy would not conflict with other land use, land cover, or built environmental features in the landscape and applying them to the spatial data. This does not account for whether trees could be planted near those spaces but is framed as “if trees exist nearby, could canopy grow to fill the space?”
3. **“Grow” tree canopy from spaces considered allowable for planting (and potentially from existing canopy), constrained to areas delineated as allowable for canopy.** The maximum amount that canopy is grown can be specified based on additional assumptions regarding how large trees may be anticipated to grow.

While practical canopy mapping can be conducted for an entire city based on a holistic set of data and assumptions, it can also be stratified to incorporate unique assumptions for different geographic units or land use, zoning, and jurisdiction, among other characterizations.

### Mapping practical canopy in New York City

#### Creating a base layer: Processing land cover and land use data layers

We combined a suite of relevant data layers related to where trees can likely be planted (planting allowability) and where canopy could theoretically exist (canopy allowability) in the current landscape into a single data layer, hereafter referred to as the “base layer” (the full list of data layers used is available in [Supplementary material](#)). The base layer was developed primarily from a suite of planimetric layers reflecting features across the landscape including building footprints, roadbeds, medians, sidewalks, parking lots, and recreation fields, among others, as two-dimensional polygons. We retained information associated with these data layers as needed—for example, we included estimated building height from the building footprint layer, useful in setting



canopy allowability. While individual properties were not wholesale included in the base layer, we included boundaries of particular types for which we specifically delineated planting and canopy allowability (e.g., airports and community gardens). Furthermore, we masked out areas considered natural, as areas for which canopy is not necessarily appropriate given ecological context and management goals. We did this based on a data layer from the NYC Department of Parks and Recreation (NYC Parks) for properties managed by that agency (the Dominant Type dataset), and an ecological cover type map from the Natural Areas Conservancy (O'Neil-Dunne et al., 2014) for the rest of the landscape. For informing the discussion of practical canopy with the NYC UFTE, staff from NYC Parks and the Natural Areas Conservancy provided estimates of potential for new canopy in the near term for these spaces within city-owned land as an aggregate (i.e., not spatially explicit), suggesting a relatively small area of canopy (81 ha) may be added to these spaces as a result of natural processes (e.g., succession) or planting in the next 10–15 years.

All datasets included in the base layer were the most recent available (spanning 2010–2021) and represented an approximation of the landscape at the time of analysis. Many of the datasets originated from a set of planimetric data based on digitization of aerial imagery from 2014, though we supplemented more recent data as available, such as of building footprints and landscape elements within NYC Parks' jurisdiction. We augmented data on roads based on spatial joins between roadbeds and a

regularly updated line dataset of roadways maintained by the City government.

We generally used the spatial data as obtained from the various sources, with two main exceptions (detailed data processing steps and list of data used are available in [Supplementary material](#)). First, airports were treated as a special case, as there are often height restrictions that extend beyond their boundaries (e.g., per [Zoning Resolution of the City of New York, 1993](#)). Thus, we manually extended the boundaries of the two active airports in NYC, based on input from partners who have experience in this realm and visible patterns of limited trees along flight lines in aerial imagery. Second, boundaries of recreation fields often only encompassed actual playing surfaces (or even a subset, such as the infield diamond of a baseball field) and did not include other, adjacent, actively used spaces such as where players sit. We examined myriad examples of these data with aerial imagery, and after consultation with local experts, we buffered recreation fields by 30.48 m (100 ft) before incorporating them into the base layer to account for such limits of these data. All data used were downloaded in or reprojected to a common coordinate reference system, EPSG 2263 [New York State Plane, Long Island Zone (ft), NAD 83], which supports accurate area calculations for the focal area. Spatial data were processed using a combination of ArcGIS Pro version 2.8 (Esri Inc., 2021), PostgreSQL version 13.0/PostGIS 3.1 (PostGIS Project Steering Committee, 2021; [The PostgreSQL Global Development Group, 2021](#)), and QGIS version 3.12 (QGIS.org, 2020).

## Defining planting and canopy allowability

For each layer we incorporated into the base layer, we considered whether the areas represented could likely support new trees being planted (with canopy growing directly above those spaces; “planting allowable”), new tree canopy overhanging (“canopy allowable”), or neither (see [Figures 2A,B](#)). This enabled us to approximate where new trees and their respective canopy could be added to the landscape while avoiding fundamental conflicts with current land use (e.g., active recreation fields), land cover (e.g., avoiding existing canopy), and infrastructure (e.g., canopy generally cannot extend atop taller buildings). A list of the types of polygons present in the base layer and the designation assigned for planting and canopy allowability can be found in [Supplementary material](#).

We considered spaces as not allowable for tree planting when:

- Tree planting would, in general, be implicitly incompatible with the use of, or the infrastructure in the space, as discernable in the available data. For example, spaces encompassed within building footprints, active recreation fields, roadbeds, and water bodies were not considered “allowable” for tree planting in our analysis.
- Logistics or regulations are generally understood to substantially constrain tree planting in certain parts of the landscape with specific land uses, histories, or infrastructure, such as airports and landfills. Cemeteries were also included in this category; while some cemeteries have canopy cover and are managed in part to maintain trees, management practices and logistical constraints can vary widely and thus we erred on the conservative side in this case.
- Ground level surfaces were estimated to be paved in any way, given that there is often substantial work required to make the space suitable for planting a tree (albeit see section on street trees below). Recognizing trees require some space to even be planted, non-paved areas were required to be a minimum area of 2.32 m<sup>2</sup> (representing a small tree bed).

We considered spaces as not allowable for additional canopy on the landscape when:

- Infrastructure that trees would generally not be tall enough to overhang was present (such as buildings taller than 10.67 m and roadway overpasses; see [Supplementary material](#) for further detail).
- Clear lines of sight and unplanted areas are required as standard procedure to manage things like risk associated with downed branches (e.g., over travel and shoulder lanes of highways).

- Overhanging canopy may conflict with the primary use of a space (e.g., community gardens that rely on sun exposure for fruit and vegetable production).
- There is existing canopy.

This delineation of allowability for planting and canopy was conducted for the entirety of NYC, excluding natural areas (beyond the scope of the effort described herein) and sidewalks in rights of way, where street trees could be planted (treated uniquely, per the section Estimating planting allowability for street trees).

## Estimating planting allowability for street trees

Street trees in NYC are trees associated with public surface streets, typically planted along sidewalks, under the jurisdiction of NYC Parks. They were considered separately from other trees because they are subject to specific rules regarding where they can be planted due to their potential impacts on intersections, sidewalks, and existing street trees documented in the *Street Tree Planting Standards for New York City* ([City of New York, 2016](#)). Per these rules, a street tree should generally be planted: (1) a minimum of 6.10 m away from another street tree and (2) a minimum of 12.19 m from the corner of a road intersection ([City of New York, 2016](#)). To simulate new street trees, we used the base layer in conjunction with data from the most recent (2015–2016) street tree census, to assign areas that comply with these rules as “planting allowable” on each blockface (the continuous frontage along a block, along a single street, between corners at either end; [The City of New York, 2017](#)). We then used a data layer representing estimated capacity for street trees along each blockface (provided by NYC Parks) to determine how many additional trees may be planted given the existing ones. We then randomly placed up to that number of points along the respective blockfaces, in accordance with the aforementioned standards.

## “Growing” the canopy

With the areas considered allowable for new tree planting and canopy designated, as well as the points representing potential locations of new street trees, we modeled or “grew” the canopy (illustrated in [Figure 2](#)). This entailed buffering the plantable areas and simulated street tree points to represent canopy grown, restricted to the areas considered allowable for canopy. To set a buffer, we calculated the average estimated canopy diameter of street trees and those in landscaped portions of city-owned parkland for the 10 most common species in each, leveraging diameter at breast height from respective datasets (see [Treglia et al., 2021a](#) for a more in-depth discussion of these data) and species-specific growth equations ([McPherson et al.,](#)



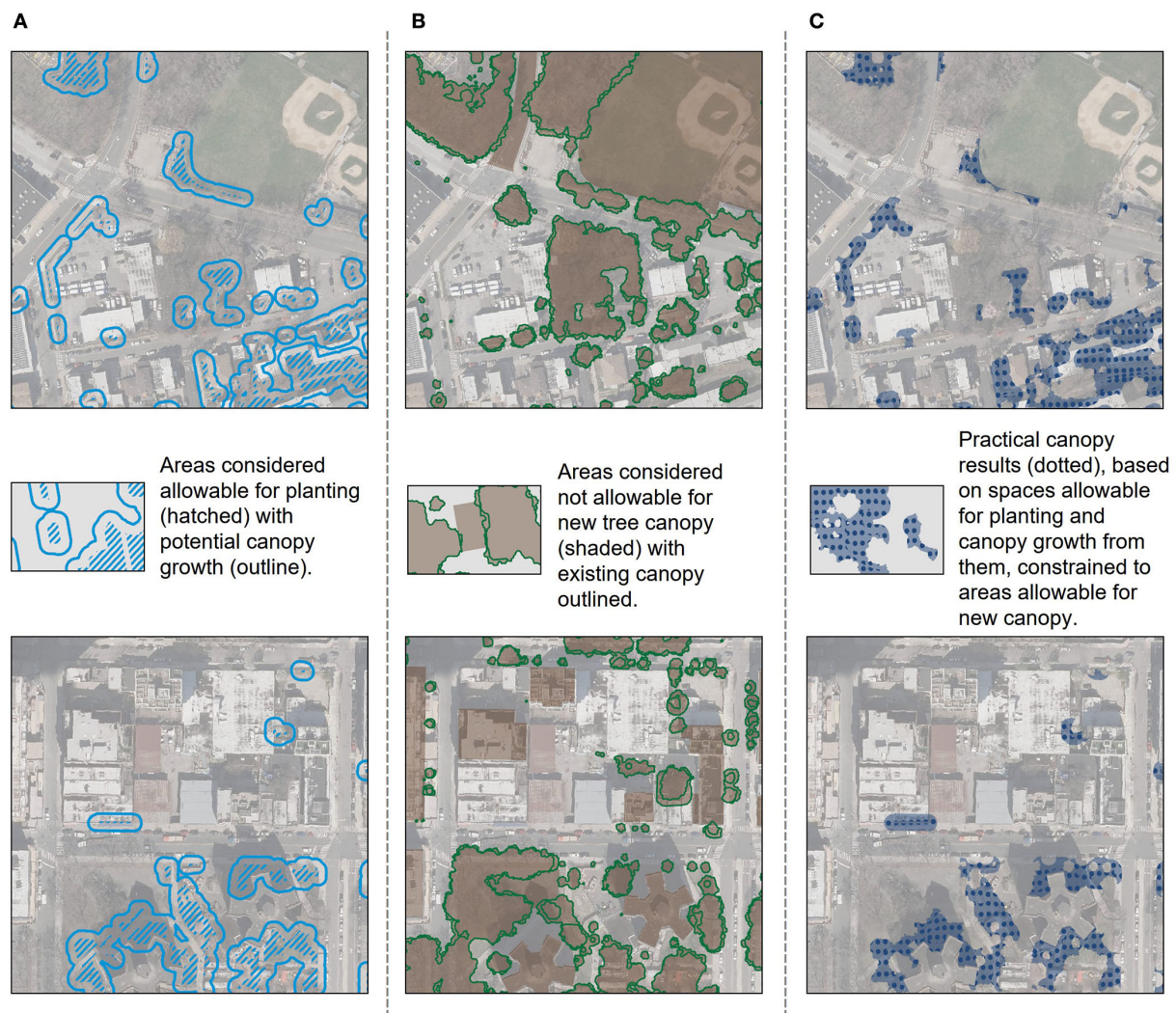


FIGURE 2

Illustrative maps representing the process of mapping practical canopy in New York City, including delineation of where the landscape was considered allowable for tree planting (A), where the landscape was considered allowable or not for canopy (B), and how the two were used together to map practical canopy (C). The concepts apply the same in the top and bottom images, but in areas of the landscape with different levels of development and complexity. Imagery is courtesy of the City of New York, Department of Information Technology and Telecommunications.

2016). The buffer employed was 4.11 m (representing a 8.22 m diameter canopy per tree). The model is not temporal in nature; thus, while myriad factors influence canopy size of individual trees, our approach is intended to represent a general average at any given time since young trees are typically planted as larger ones senesce through time. We attributed the canopy “grown” to new trees associated either with plantable areas or with the simulated new street trees. In instances where practical canopy from these sources could overlap (e.g., along boundaries between individual properties and rights of way), we attributed the area of overlap to street trees for accounting purposes, given they are all within the jurisdiction of a single entity (NYC Parks). The

spatial data, representing canopy “grown” in this step (restricted to exclude spaces considered not allowable for canopy) and those representing plantable area, together comprised the final practical canopy layer (depicted in Figure 2C).

## Characterizing practical canopy in New York City

Once the practical canopy layer was developed, we overlaid it with spatial data representing a suite of political, administrative,

and jurisdictional datasets to derive descriptive summaries for interpretation, to enable comparison with the distribution of existing canopy, and to support discussion with members of the NYC UFTF. We summarized practical canopy data citywide, and by the following units, in order of decreasing size: boroughs (each representing a single county, and with an elected representative, a Borough President); City Council Districts (each with an elected City Council Member); Community Districts (each with an associated board of community members); and Neighborhood Tabulation Areas (NTAs; a unit used for planning purposes designed to be smaller than City Council Districts, with  $\sim 15,000$  residents within each). Each is relevant to planning and decision-making in NYC, as they align with specific levels of governance, civic engagement, or serve as planning units. We focus our results herein on citywide, borough, and NTA scales, representing the largest and smallest scales, to help highlight overall trends as well as local nuance. NTAs also include aggregated areas that have unique, non-residential uses (e.g., large tracts of land dedicated to parks and airports), which we included in summaries and analysis. Though a set of newer NTA boundaries is available, updated after the 2020 decennial census, we used the previously developed layer, created following the 2010 decennial census, to support comparison with previous analyses, such as those of existing canopy (Treglia et al., 2021b). A detailed map of boroughs and NTAs is available in [Supplementary Figure 1](#).

We also delineated whether practical canopy was associated with street trees, plantable area, or the “growth” around plantable areas, and we characterized the distribution of practical canopy by general jurisdiction (e.g., City properties and rights of way (assumed to be City land), New York State, Federal, or private), and for private property, generalized land uses. Ownership data were generally derived from a parcel dataset available for NYC, MapPLUTO (version 20v6), or agency-specific datasets, described in appendices of Treglia et al. (2021a).

## Canopy comparisons

We compared the distribution of potential for canopy based on practical canopy by administrative or political unit to the distribution of existing canopy as of 2017, the most recent time point for which there is a robust, LiDAR-based canopy data layer, using the results from Treglia et al. (2021b). This comparison allows us to understand what the practical canopy means in terms of opportunities to expand the urban forest in different spaces across the city. At the scale of NTAs, both citywide and by borough, we examined Kendall's  $\tau$  correlations (Kendall, 1938) to understand the relationship between the percentage of each area covered by canopy as of 2017 and that which would be covered by canopy with the inclusion of practical canopy. This offers insight into whether, in general, adding practical

canopy would change the rank order of NTAs in terms of total canopy (positive correlations would suggest that, in general, practical canopy would not change which areas have the most and least canopy). We considered significance for Kendall's  $\tau$  correlations based on  $\alpha = 0.05$  and incorporated best-fit lines with scatterplots of the data to support interpretation. This analysis was conducted using the `cor.test` function in R version 4.0.2 (R Core Team, 2020). We also examined whether realizing practical canopy would reduce the disparity in tree canopy by comparing the ranges in canopy cover across NTAs by borough based on the existing canopy and the existing plus practical canopy.

We also compared the practical canopy to an estimate of “possible canopy” for NYC (*sensu* Grove et al., 2006; considered as a representation of where canopy is “biophysically feasible”). For this, we calculated the possible canopy using a comparable methodology to that described by Grove et al. (2006) and Raciti et al. (2006), as the land area that was not existing canopy, water, buildings, roads, or railroads (added as an available, relevant land cover class for this analysis). For this work, we leveraged the most recent high-resolution land cover data for NYC representing the landscape as of 2017. This comparison allowed us to better understand the differences between the existing typology of potential for new canopy and our proposal, “practical canopy.”

## Results

### Summaries by borough and Neighborhood Tabulation Area

The spatial data layer of practical canopy we developed for NYC represents 15,899 ha (20.31% of the NYC land area) that we estimate could likely be covered by tree canopy from planting and growth of additional trees while accounting for constraints associated with current land use, land cover, and the built environment. The resultant data layer from this work, as well as summaries by borough, City Council District, Community District, and Neighborhood Tabulation Area (2010) are available in a public repository at <https://zenodo.org/record/6547492> (Treglia et al., 2022).

The distribution of practical canopy among the five boroughs of NYC generally followed their rank order by land area, with Queens containing the largest share of all practical canopy in NYC (42.70%) and Manhattan containing the smallest (3.09%) (Table 1). Brooklyn and Staten Island were the only boroughs that did not follow this trend; Brooklyn is the second largest borough but has the third highest practical canopy area, and Staten Island is the third largest borough, but has the second highest practical canopy area. The trends in terms of practical canopy by borough align with trends in existing canopy, as of the most recently available canopy dataset for NYC. Staten

TABLE 1 Summary information of land area, existing canopy, practical canopy, and “possible canopy” (*sensu* Grove et al., 2006), by borough of New York City and citywide.

Borough	Land area (ha)	Practical canopy (ha)	Existing canopy 2017 (ha)	Practical canopy cover (%)	% of total practical canopy	“Possible canopy” (ha)	Mean NTA practical canopy (%) $\pm$ SD	Range of NTA existing canopy (%)	Range of NTA practical + existing canopy (%)
Bronx	11,024	1,948	2,733	17.67	12.25	4,294	17.03 $\pm$ 9.25	3.06–50.47	14.93–70.81
Brooklyn	17,968	2,591	3,165	14.42	16.3	7,300	14.17 $\pm$ 5.48	7.82–27.99	14.90–53.93
Manhattan	5,914	491	1,264	8.3	3.09	1,675	6.83 $\pm$ 3.38	2.90–39.51	7.87–59.67
Queens	28,280	6,788	5,344	24	42.7	12,811	26.60 $\pm$ 11.71	2.43–35.83	2.95–70.79
Staten Island	15,085	4,080	4,748	27.05	25.66	6,743	30.81 $\pm$ 8.54	19.67–48.46	31.81–75.22
Citywide	78,272	15,899	17,254	20.31	100	32,823	18.95 $\pm$ 11.56	2.43–50.47	2.95–75.22

Columns titled with ‘NTA’ contain aggregate statistics for the respective Neighborhood Tabulation Areas.

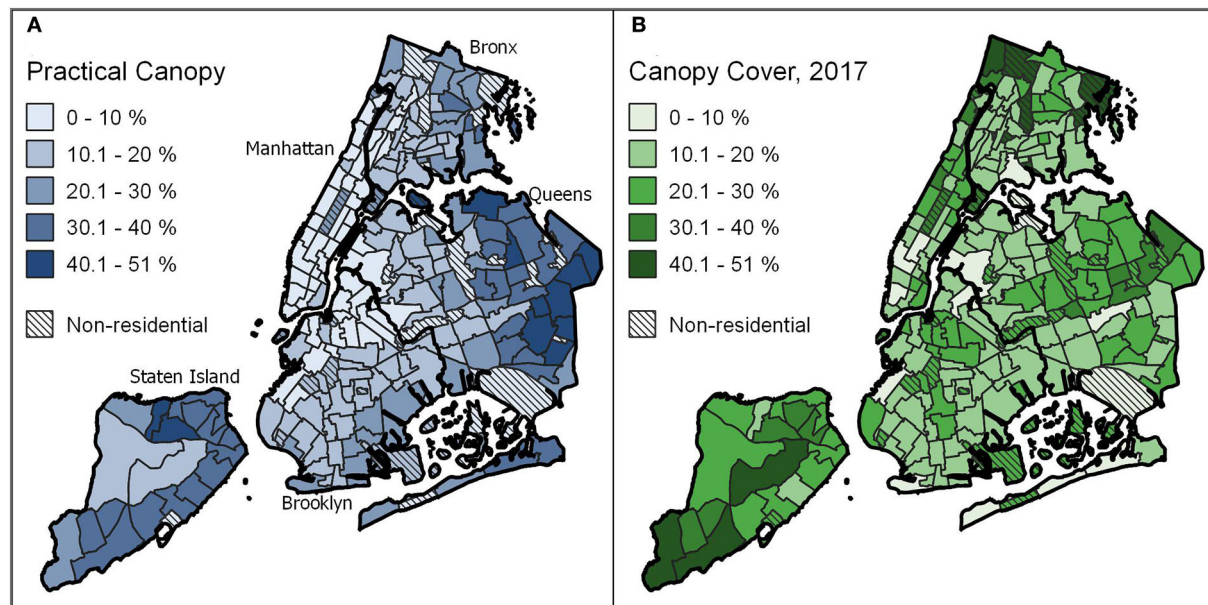


FIGURE 3

Maps illustrating the practical canopy (A) and existing canopy as of 2017 (B) as percent of land area by Neighborhood Tabulation Area. Thicker borders delineate the borough boundaries [with boroughs labeled on (A)]. Borough and Neighborhood Tabulation Area Boundaries are from the City of New York, Department of City Planning. Non-residential areas are generally aggregated by borough in those datasets and as presented here. Summaries of existing canopy cover are from Treglia et al. (2021b).

Island, followed by Queens, had the largest portion of its area identified as practical canopy (27.05 and 24.00%, respectively), with Manhattan having the lowest (8.30%) (Table 1).

Practical canopy within NTAs (Figure 3A) generally reflects the patterns within the respective boroughs, as the rank order for average percent of land area mapped as practical canopy by NTA within each borough was the same as the rank order for percentage of land area mapped as practical canopy by borough as a whole (Table 1). There is substantial variation in the percentage of each unit mapped as practical canopy at this more granular scale; the lowest value for an NTA was 2.74%, in the Clinton area of western Manhattan (MN15) and the highest

value was 49.87%, in Cambria Heights, eastern Queens (QN33). In terms of areas with special uses, the one representing JFK International and LaGuardia Airports (QN-98) had the lowest percentage of area with practical canopy (0.52%), and Riker's Island (BX-98) had the most (50.47%). The variation tends to be moderated within every borough except for Queens (Table 1).

Citywide, only 6.38% of practical canopy was attributable to street trees, with the remainder associated with spaces considered allowable for planting (34.57%) or the buffered area representing canopy growth from those spaces (59.05%). The Bronx and Queens both have about 6% of their practical canopy attributable to street trees, though Manhattan and Brooklyn



have substantially more (14.60 and 10.31%, respectively); Staten Island has less, only 3.42%. In terms of jurisdiction, the majority of practical canopy mapped (68.78%) was within private property, followed by city land (25.28%; primarily within rights of way, generally associated with canopy grown from plantable area within adjacent properties; see available results files), state (4.14%), and federal properties (1.80%) (Figure 4A). While this varied by borough, Manhattan was the only one not to have the majority of practical canopy within private property (the majority there, 56.97%, was within the jurisdiction of the city). Furthermore, the large majority of practical canopy mapped on private property was within 1–2 family residential properties, and this was true across all boroughs except for Manhattan, in which the majority of private property practical canopy fell within 3+ family residential properties (Figure 4B). These breakdowns by NTA are available in summary result files (Treglia et al., 2022).

## Practical canopy compared to existing (2017) canopy and “possible” canopy

The 15,899 ha of practical canopy mapped citywide is nearly the same area covered by canopy in NYC as of 2017, 17,254 ha (Treglia et al., 2021a), indicating the potential to nearly double tree canopy at this scale if all practical canopy were realized and existing canopy cover was maintained (achieving 42.35% canopy cover total). Given the variation in borough-level canopy and practical canopy (Table 1) the largest relative increases would be the greatest in Queens (127.04%), more than doubling its canopy, and the smallest would be in Manhattan (38.84%), with the potential relative increases in other boroughs ranging 71.27–85.93%.

Citywide and across all five boroughs, we found significant positive correlations between the practical canopy and practical plus existing canopy within NTAs (Figure 5). This indicates that, in general, the rank order of the NTAs in terms of canopy would not change if all practical canopy mapped in this analysis were realized. Furthermore, in all boroughs, the range of canopy cover across the NTAs would increase. Thus, while all NTAs would see at least some increase in canopy cover, realizing all practical canopy would lead to an increase in the disparity between areas with the most and least canopy; the ranges in canopy across NTAs would increase in all boroughs and citywide (Table 1).

Our estimate of “possible canopy” (*sensu* Grove et al., 2006) (32,823 ha) was more than double the area of practical canopy. The “possible canopy,” relative to practical canopy, was highest in Manhattan and Brooklyn (3.41 and 2.82 times higher, respectively) and lowest in Staten Island (1.65 times higher). “Possible canopy” covered 41.93% of the NYC landscape, and if added to existing canopy would suggest opportunity for a total of 63.98% canopy cover citywide.

## Discussion

Our estimate of practical canopy suggests the existing NYC landscape could likely support 15,899 ha of additional tree canopy. If all practical canopy were realized and the existing canopy is maintained, the canopy cover in NYC would nearly double, to 42.35% of the land area. The methodology we developed relies on making explicit assumptions of where trees could be planted, informed by local context and data, and thus enables deeper conversations or iterative analysis depending on the needs of those using the information. Comparing existing canopy cover, “possible canopy,” and practical canopy additionally provides a more complete picture of urban forest possibilities in a way that enables discussion of what may be required to address inequities in the NYC urban forest. Notably, the existing urban forest in NYC should not be taken for granted, as it is susceptible to loss from various challenges, requiring ongoing protection and stewardship (Treglia et al., 2021a). Protection and stewardship would also be required for newly planted trees to achieve the canopy simulated in the practical canopy analysis. It is imperative that future planning efforts take these dynamics into account. Ultimately, by promoting deeper conversation and a nuanced understanding of the landscape, the practical canopy analysis facilitates a framework for a “priority” canopy, which can then be acted upon. Our NYC practical canopy analysis grounded discussions around what a visionary and achievable goal could be in the current urban landscape. It not only informed the goal of at least 30% canopy cover by 2035 put forth in the *NYC Urban Forest Agenda*, but also has made clear that to achieve a more just urban forest, it will likely be necessary to create new spaces for planting, beyond what exists in the current landscape. Throughout this process, conversations have been in line with what is required to set forth a virtuous cycle (Morrison, 2015, 2016) where technical information and analysis, such as practical canopy mapping, support buy-in for planning and implementation efforts, in iteratively larger circles of stakeholders.

The concept of practical canopy is broadly transferable, and implementation can be adapted to a given place using locally relevant data and assumptions. Efforts for operationalizing it in small areas can potentially leverage on-the-ground mapping and knowledge, although robust analysis of for broader areas (e.g., citywide) requires reliable data on land use, land cover, and built infrastructure, for which availability varies substantially. Thus, we hope that as more data are generated for different cities, this type of work can be broadly replicated, but the analysis, as we have conducted it in NYC, may not be readily accomplished everywhere. As with any modeling effort, despite the local expertise and relatively rich data we incorporated into our analysis for NYC, there are limits in our results. In some cases, for example, we identify that the available data do not fully capture constraints in terms of where the urban



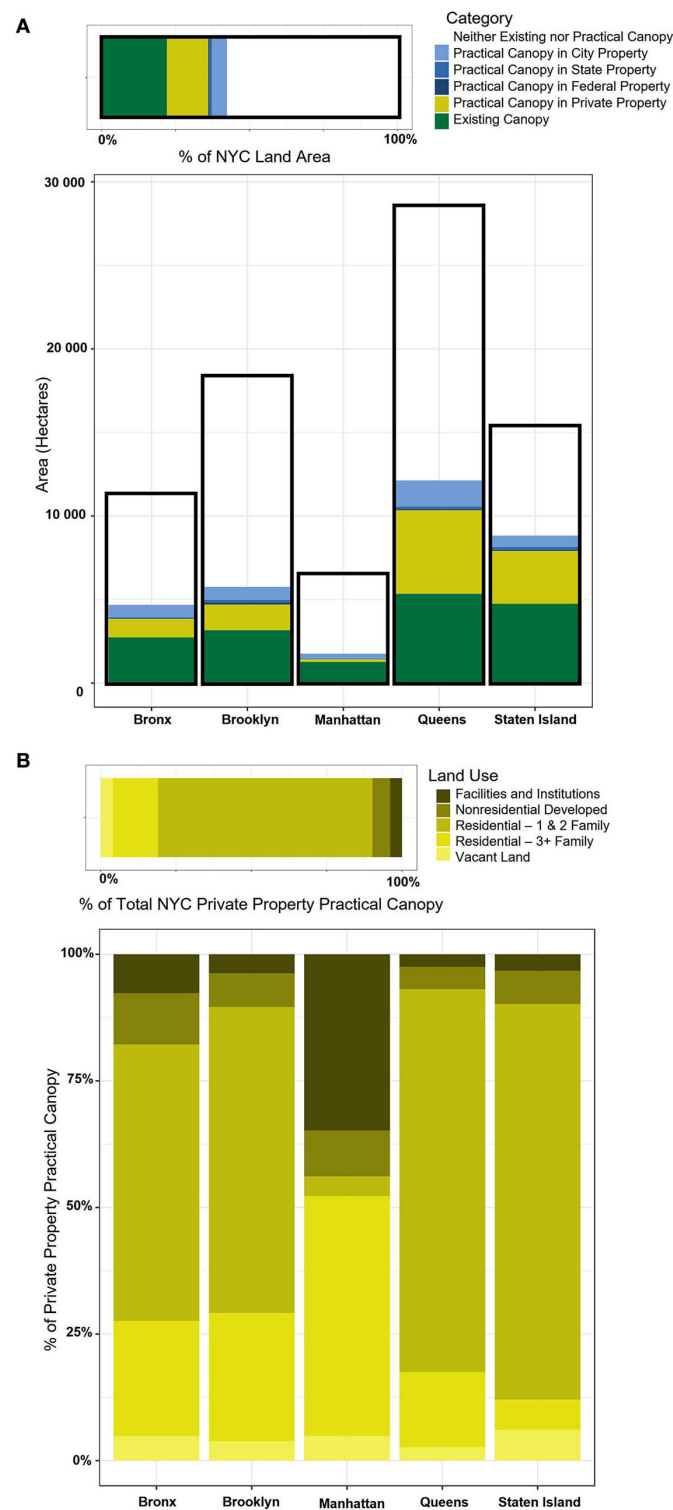


FIGURE 4

Stacked bar charts showing the distribution of practical canopy by ownership type, as well as existing canopy and land with neither canopy nor mapped practical canopy, both citywide and by borough (A), and the breakdown of practical canopy among different land uses of private property, citywide and by borough (B). For (A), City Property includes rights of way, generally within the jurisdiction of the City of New York; when State or Federal practical canopy is not discernable, it represented a small very small portion, if any, of the practical canopy. For (B), land uses are aggregated from parcel data for NYC (see [Supplementary material](#)).

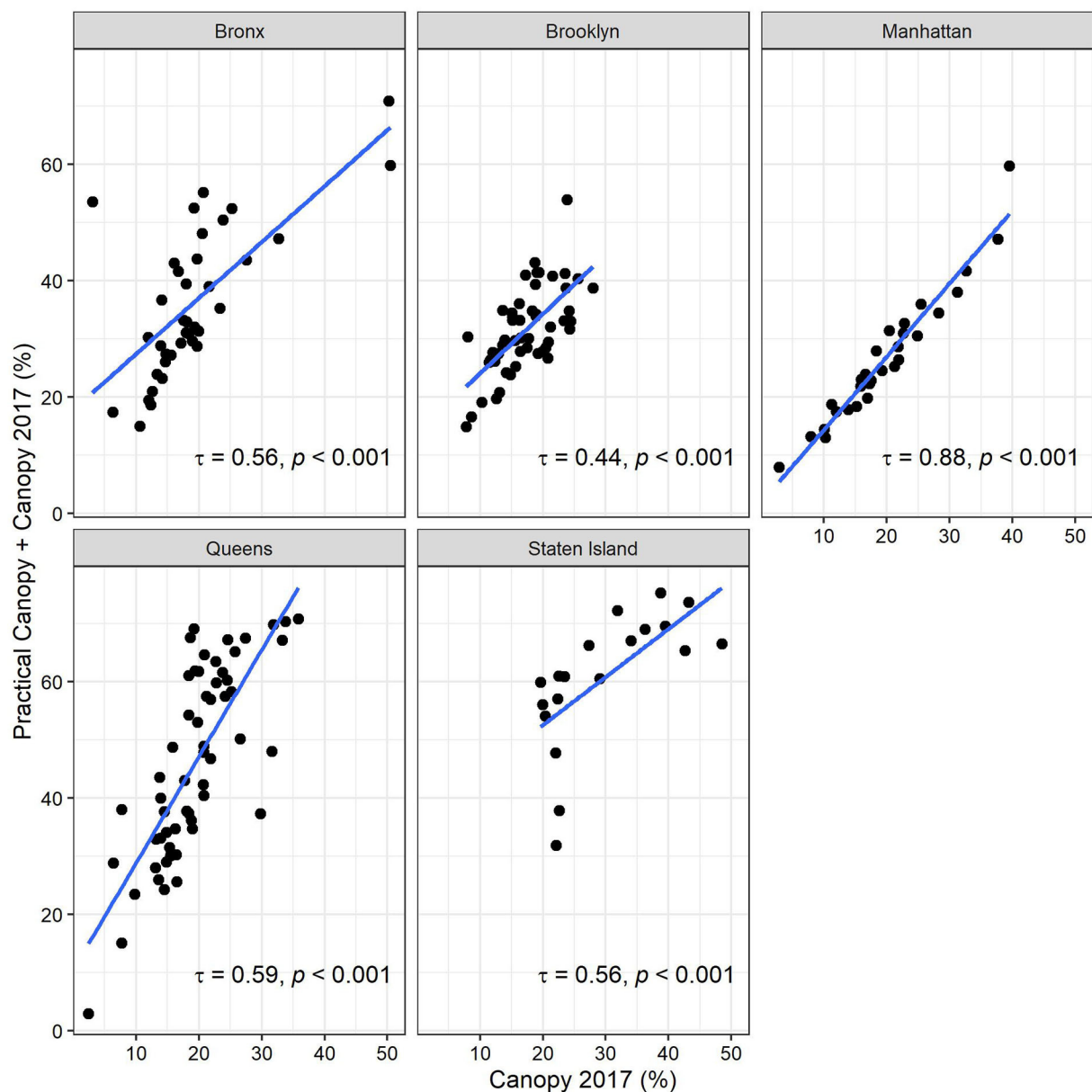


FIGURE 5

Scatterplots, by borough, showing existing canopy (as of 2017) and the combination of practical and existing canopy, both as percentages of land area for each Neighborhood Tabulation Area.  $\tau$  represents Kendall's  $\tau$  correlation coefficient, and  $p$  represents the respective  $p$ -value. Best-fit lines are displayed to support interpretation.

forest could be expanded, with practical canopy appearing in the infield of Kissena Velodrome in Queens, as that space is not entirely reflected as an active recreation space in the data, and while underground infrastructure can limit opportunities for planting, such data were not available. There may also be cases of underestimation of practical canopy, such as associated with our assumptions of limited opportunity for planting on cemeteries and within airport boundaries. Thus, more robust

data and even further refined assumptions could improve this analysis, and if applied in different places, different factors may need to be accounted for. Furthermore, future work can include sensitivity analyses to yield a more complete understanding of how different datasets and assumptions impact the results. In addition, the urban forest is also just one part of an urban system; other forms of greenspace and open space, such as green roofs, green walls, and gardens,

offer myriad benefits and could also be considered in a broadened scope.

We see the iterative process of considering data and assumptions together as a refinement of the three P's ("possible," "potential," and "preferable" canopy; Grove et al., 2006) as the general categories of each P, "biophysical," "economic," and "preferable," are not truly distinct. Instead, they inform each other and are dependent on the people making decisions, generally based on the data available. Their application then demands a step that is "practical," working explicitly to ground conversations and priorities without being prescriptive. Our effort to explicitly document the data and assumptions can enable researchers and practitioners to refine this work based on new information or different objectives. For example, while cemeteries were considered not suitable for tree planting in our analysis, we recognize there is variation in how cemeteries are managed. The Green-Wood cemetery, as a case in point, is an arboricultural leader, qualified as a Level III Arboretum (Treglia et al., 2021a). Thus, additional opportunities for new canopy can be explicitly incorporated with refined or targeted analyses and assumptions. Functionally, the practical canopy is a spatial model that does not necessarily incorporate local perspectives, all constraints at play, or the potential to fundamentally change the landscape to create new canopy or planting opportunities (e.g., un-paving land). However, it can ultimately inform where fundamental changes to the landscape may be needed to achieve expansion of the urban forest.

The comparisons between the practical canopy and both the existing and "possible" canopies for NYC elucidate how context dependent understanding of opportunities for urban forest expansion can be. We expected the "possible canopy" to be greater than practical canopy because the former focuses only on relatively coarse assumptions of where new canopy can go based on the biophysical landscape, without consideration for where trees from which that canopy would grow can be planted or what the actual land uses are (e.g., if land is used for active recreation). In early work, we explored applying the "possible canopy" methodology of Grove et al. (2006) for NYC. We recognized its utility in starting conversations, and we began to better understand its limits. It ultimately inspired development of the idea of practical canopy, particularly given the wealth of data available for NYC that enabled a more realistic model that can account for specific constraints and opportunities for the urban forest. For example, while "possible canopy" does not allow canopy over any buildings or roadways, we were able to incorporate potential for canopy over short buildings and surface roads into practical canopy.

In exploring the relationships between practical canopy and existing canopy, we observed that while all areas of the city had some practical canopy, many areas with little existing canopy also had little practical canopy. Examples include in midtown Manhattan and, to a more moderate degree, the South Bronx (Figure 3). While one might expect that places with low canopy

would have more opportunity for new canopy because they have not been paid attention to for urban greening, our results show that the existing landscapes, driven by various factors that shaped development history, have real constraints in terms of expanding the urban forest, as these areas have urban forms that are largely incompatible with broad greening efforts. Places with low canopy cover that have generally not had green space prioritized have often been paved over for other uses (Gould and Lewis, 2017) and are not simply "low-hanging fruit" for expanding the urban forest. We see this is indeed a general trend, as realizing practical canopy cannot counter the disparities in existing canopy across the city, though there are exceptions (see Figure 5).

Our results show that reducing disparities in tree canopy across NYC will require meaningful changes in the landscape that enable more planting of trees where there is little canopy. In general, urban forest goals are often established at a citywide level to improve access to benefits of trees and their canopy, and sometimes vegetation more generally, as in the case of efforts to mitigate urban heat challenges, particularly given warming temperatures associated with climate change (Eisenman et al., 2021). However, consideration of more granular spatial units is often needed to be relevant for the local impacts of challenges such as the urban heat island effect: in NYC, Johnson et al. (2020) identified a 32% vegetative cover threshold within a 12.6 ha area (approximately equivalent to a Manhattan block) before temperatures are cooled by vegetation, and in Madison, WI, USA, Ziter et al. (2019) suggested that 40% canopy cover in a 25 ha area is required before the cooling effects of increased vegetation are felt. When we consider our practical canopy results, neither the hottest areas (see Johnson et al., 2020) nor the areas with the most heat-vulnerable communities (mapped by the NYC Department of Health and Mental Hygiene) are among those with the most practical canopy (with a notable exception of Jamaica, Queens; Figure 3) or those that would see their circumstances substantially change in terms of canopy (Figure 6). This result may partially reflect that the driving force in the urban heat island effect is the differential rates of energy storage and release by different substrates, of which impervious surfaces (buildings and paved surfaces) store and release the most heat (Ward and Grimmond, 2017). Thus, the hottest areas (albeit not always the most heat-vulnerable ones) may inherently be some of those with the least practical canopy given the high densities of impervious surfaces. In addition, the findings of Ziter et al. (2019) and Johnson et al. (2020) suggest some of these interventions have to be considered at a scale as small as individual blocks, since at larger scales, cooling effects of trees may not be felt from one edge of a unit to another. Further research is needed to better understand how temperature reduction benefits of urban forests scale across the landscape and could inform more specific local goals, though expanding the benefits of the urban forest such as this can ultimately help increase support for the resource in the

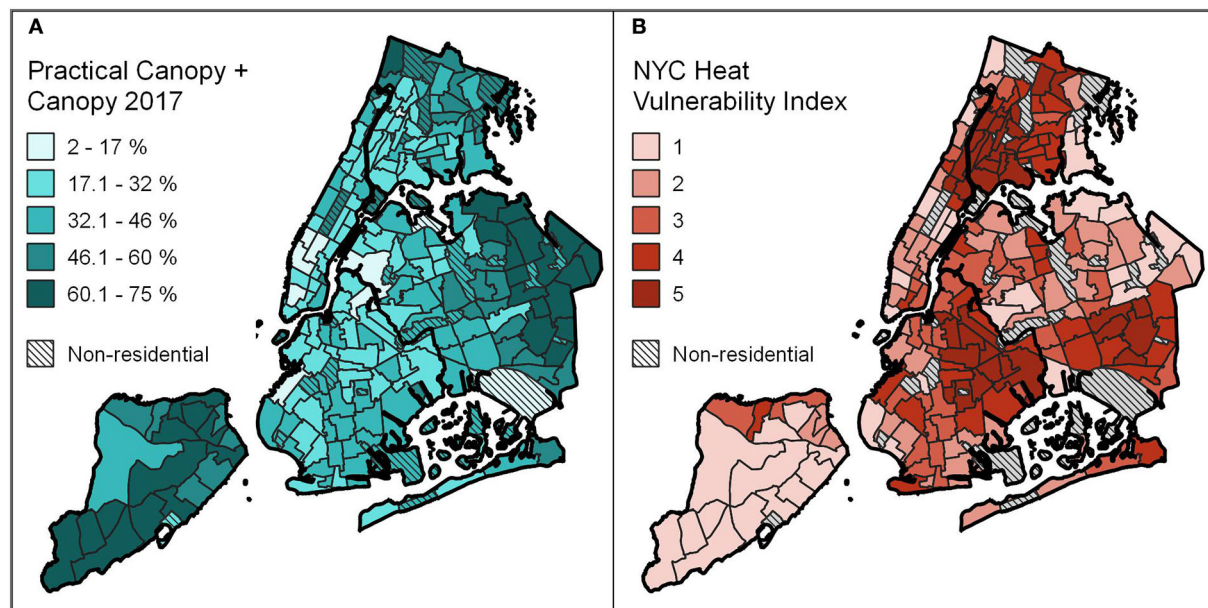


FIGURE 6

Maps illustrating what canopy cover (%) would be if all practical canopy mapped were realized, assuming maintenance of existing canopy as of 2017 (A), and the NYC Heat Vulnerability Index (2018 version), by Neighborhood Tabulation Area (B). Borough and Neighborhood Tabulation Area Boundaries are from the City of New York, Department of City Planning. Non-residential areas are generally aggregated by borough in those datasets and as presented here. Data on existing canopy used in (A) are from Treglia et al. (2021b); the NYC Heat Vulnerability Index is available from the NYC Department of Health and Mental Hygiene at <https://a816-dohbsp.nyc.gov/IndicatorPublic/VisualizationData.aspx?id=2411,719b87,107,Map,Score,201>.

positive feedback loop of the virtuous cycle. While increasing access to the urban forest and its benefits is important through lenses of equity, public health, and general climate resilience, it is important that communities affected are authentically engaged, with opportunities for their visions to be elevated to support their self-determination for a more just end result (Campbell et al., 2022). Further, such engagement, in concert with other policies, can help prevent consequences such as green gentrification, if the goal is to expand urban forest to those who stand to benefit the most (Gould and Lewis, 2012; Schell et al., 2020; Campbell et al., 2022; García-Lamarca et al., 2022).

Three examples of means by which the landscape can be changed to accommodate expansion of the urban forest are as follows: through broad changes in zoning regulations; rezoning specific neighborhoods; and redesigning streetscapes, within which street trees are generally planted. For example, in 2008, the City Planning Commission in NYC created a requirement in the zoning resolution that in almost all cases, new buildings and large alterations citywide have to either plant or protect a street tree for every 7.62 m of frontage on the building (Zoning Resolution of the City of New York, 2011). Furthermore, local areas can have more regulations or enabling conditions that support protection and expansion of the urban forest as part of zoning processes, and rezoning can result in future development (or redevelopment) that creates more opportunities for tree

planting and canopy growth; special purpose zoning districts can also be established with more specific urban forestry requirements (e.g., as with the Special Natural Area District; Treglia et al., 2021a). Finally, the COVID-19 pandemic and a citywide commitment to decrease dependence on fossil fuels have created space for conversations on re-envisioning the right-of-way (Freudenberg et al., 2021). Streetscapes can be designed to prioritize vegetation and permeable surfaces, often in concert with other sustainability and livability improvements, such as for pedestrians and cycling. This can ultimately support depavement and tree planting, and even daylighting of below-ground streams (that were once aboveground) with riparian vegetation buffers (Freudenberg et al., 2021). Deciding which strategy makes sense where and how to prioritize expansion of the urban forest requires coordination with those who will be affected by such decisions and landscape changes.

Deciding when and how to promote landscape change is a subsequent step from identifying the priority canopy, or where canopy is most desired and needed for its benefits, regardless of existing constraints. This can build on and perhaps incorporate existing prioritization approaches that strive to represent various perspectives from across a city (e.g., Locke et al., 2010, 2013), while centering on more local perspectives. Stakeholders and decision makers can inspect the results in dialogue within the context of other relevant initiatives, the policy landscape, and



priorities of the local communities. Specifically, high practical canopy but low existing canopy in an area can suggest the need to leverage available planting spaces; low practical and low existing canopy may suggest a need to re-envision the local landscape; and areas with high existing canopy, in general, may require tree preservation and stewardship efforts, and it is critical that these be considered more broadly in planning for the resource. Practical and existing canopy each reflect some dimensions of land use and social or natural histories that can be made more explicit, and preferences and needs for the future can be developed from there, by or with local communities.

Understanding dimensions of existing and practical canopy can also have implications for broader urban forest planning efforts, particularly when considered with jurisdictional and land use data. Based on our analysis in NYC, from a citywide perspective, it may be critical to prioritize engagement with private property owners, particularly those that own 1–2 family residential properties (Figure 4B), given the substantial practical canopy there. Yet, geographically targeted analyses, such as in heat-vulnerable areas with limited practical canopy, may guide local efforts involving the community and government agencies to ensure a robust urban forest in the public space (e.g., street trees) or to redesign the streetscape or rezone an area to create opportunities for additional tree plantings. In such local efforts, however, it is critical to ensure local stakeholders such as residents and community-based organizations are authentically engaged. Through dialogue with local communities (tenants, homeowners, workers, political and economic actors, identity affiliations, and others), at the scale of participation that is appropriate (Arnstein, 1969; Campbell et al., 2021), valuable additional information for the priority canopy framework can be included. The landscape of politics often defines this information, for example, to balance sometimes competing priorities and understand tradeoffs (e.g., increasing building height and density to promote an increase in housing density). The urgency of climate change also requires different information to be incorporated into urban forest decision-making, such that heat- and flood-tolerant tree species need to be considered at the same time as the mitigation effects of the urban forest. As urban forest goals are implemented, these complexities can be layered on top of the existing and practical canopies to create a priority canopy.

In NYC, our development of the practical canopy analysis was spurred by conversations with other stakeholders in the NYC UFTF, in part, as a means of informing the canopy goal in the *NYC Urban Forest Agenda* (NYC Urban Forest Task Force, 2021). The Task Force was composed of approximately 50 organizations that worked to collaboratively develop the *NYC Urban Forest Agenda* between 2019 and 2021. During this time, the NYC UFTF agreed they needed, among other things, a citywide goal that would support planning, guide policy initiatives, and to spark individual and collective action.

Canopy was agreed upon as preferred metric for goal setting for several reasons: it can be measured and compared through time using periodic LiDAR-based data (when available); its change over time reflects a collection of actions or events relative to the resource (including planting, protection or lack thereof, maintenance, and stochastic events); its extent may correlate to service provisioning; and it can be understood and compared at different scales relevant to policy-making and interest of local communities. Once canopy was selected for the goal metric, the leadership of the Task Force wanted a grounding in the potential for additional canopy, which led to our development of practical canopy. It was critical that the goal be set within the context of potential resources such as funds and availability of trees to plant, and guiding principles or values (e.g., increasing equity of the urban forest, particularly through lenses of health and climate resilience, per the *NYC Urban Forest Agenda*). Furthermore, it was desired for the goal to be visionary and achievable, and simple such that it could be digestible and galvanizing, in ways that could inspire and require policy improvements, increased investments, and an expanded urban forest workforce, while having potential to improve environmental quality and climate resilience. It was also important that the goal be time-bound, such that it could spur both immediate and sustained action, while allowing for sufficient time to measure progress. Achieving a more equitable distribution, in addition to higher citywide canopy cover, was a key part of the conversation. Thus, the development and exploration of practical canopy enabled such discussions, resulting in a citywide canopy goal of at least 30% by 2035.

Since the release of the *NYC Urban Forest Agenda* in June 2021, myriad stakeholders have taken on the goal to varying degrees. The applicability of the goal across geographic scales, and the potential for it to touch down in local communities that can see benefits of achieving it may enable this to be the start of a virtuous cycle (Morrison, 2015, 2016). While mapping practical canopy was highly technical work, it ultimately supported buy-in for a canopy goal and allowed those engaged in the process to see the opportunity and potential for broad engagement by others, in expanding the urban forest. The opportunity identified, to at least some degree throughout the city and across jurisdictions, to increase canopy was galvanizing. Perhaps the same quantitative goal could have been set without this consultative process of mapping practical canopy (or with a simpler analysis), but the effort created buy-in *via* participation and discussion. Furthermore, the practical canopy data layer itself serves as a tool for conversation that supports local engagement and visioning, and ultimately, it informs ways in which the goal of at least 30% canopy by 2035 might be achieved in ways that improve equity of the resource. As the *NYC Urban Forest Agenda* was released, the NYC Urban Forest Task Force launched Forest for All NYC a growing coalition composed of over 70 organizations at

the time of this writing, which is working to advance the canopy goal, among other actions detailed in the *Agenda* to support the NYC urban forest. While tree planting goals are still part of the conversation in NYC, with a “Million More Trees” campaign initiated by the five borough presidents, the coalition has effectively advocated for the campaign to incorporate the canopy goal, strengthening both initiatives simultaneously. The goal has also been adopted by other government officials such as the Chair of the NYC Council Committee on Parks and Recreation. Thus, a virtuous cycle for the NYC urban forest may be in its early stages, where the work of the NYC UFTF and this analysis created conditions where participating in the conservation of the urban forest reinforces the long-term commitment of an increasing number of local actors. If so, it was supported by technical information grounded in the landscape context, in the form of the practical canopy analysis, that can facilitate stakeholder engagement and planning for expansion of the resource with consideration of local priorities.

## Data availability statement

The datasets presented in this study can be found in the online repository, Zenodo, at <https://zenodo.org/record/6547492>.

## Author contributions

EM and MT developed the initial concepts of practical and priority canopy as described in the article, led meetings that resulted in input and feedback from partners, contributed to writing the first draft of the manuscript, and supervised the overall work. KL and MT developed and applied the novel methodology for mapping practical canopy. MT led the calculation of possible canopy. NP led the writing of the first draft of the manuscript and led refinement of the framing for practical and priority canopy as described in the manuscript. AV and MT conducted post-processing of the practical canopy data layer to summarize the results by different geographic and jurisdictional units. AV, KL, MT, and NP developed figures for the manuscript. All authors contributed to revision of the manuscript and read and approved of this submission.

## Funding

Funding for this work was provided in part by the Leona M. and Harry B. Helmsley Charitable Trust.

## Acknowledgments

We are grateful for input and feedback on both the methods employed for mapping practical canopy and the general framing of practical and priority canopy, provided by staff from the NYC Department of Parks and Recreation, Division of Forestry, Horticulture, and Natural Resources; Sarah Charlop-Powers, Crystal Crown, and Clara Pregitzer of the Natural Areas Conservancy; Lindsay K. Campbell, J. Morgan Grove, Richard Hallett, and Dexter Locke, USDA Forest Service, Northern Research Station; Jarlath O’Neil-Dunne, University of Vermont/USDA Forest Service, Northern Research Station; and Tami Lin-Moges, The Nature Conservancy, New York Cities Program. Kate Galbo of the Nature Conservancy, New York Cities Program provided analytical support in estimating canopy diameter of existing trees in the New York City urban forest. We also appreciate the time and constructive comments from reviewers, and the editors for the special issue of *Frontiers in Sustainable Cities* as this was submitted to be part of. The use of Esri products in this work was supported by license grants from Esri to The Nature Conservancy. The work described herein is also available in a preprint at <https://www.preprints.org/manuscript/202206.0106/v1>.

## Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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## Supplementary material

The Supplementary Material for this article can be found online at: <https://www.frontiersin.org/articles/10.3389/frsc.2022.944823/full#supplementary-material>

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## OPEN ACCESS

## EDITED BY

Michele Romolini,  
Loyola Marymount University,  
United States

## REVIEWED BY

Tischa A. Munoz-Erickson,  
Forest Service (USDA), United States  
Juheon Lee,  
Midwestern State University,  
United States

## \*CORRESPONDENCE

R. Patrick Bixler  
rpbixler@utexas.edu

## SPECIALTY SECTION

This article was submitted to  
Urban Greening,  
a section of the journal  
Frontiers in Sustainable Cities

RECEIVED 09 August 2022

ACCEPTED 07 November 2022

PUBLISHED 09 December 2022

## CITATION

Bixler RP, Coudert M, Richter SM,  
Jones JM, Llanes Pulido C, Akhavan N,  
Bartos M, Passalacqua P and Niyogi D  
(2022) Reflexive co-production for  
urban resilience: Guiding framework  
and experiences from Austin, Texas.  
*Front. Sustain. Cities* 4:1015630.  
doi: 10.3389/frsc.2022.1015630

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# Reflexive co-production for urban resilience: Guiding framework and experiences from Austin, Texas

R. Patrick Bixler<sup>1\*</sup>, Marc Coudert<sup>2</sup>, Steven M. Richter<sup>3</sup>,  
Jessica M. Jones<sup>1</sup>, Carmen Llanes Pulido<sup>4</sup>, Nika Akhavan<sup>4</sup>,  
Matt Bartos<sup>5</sup>, Paola Passalacqua<sup>5</sup> and Dev Niyogi<sup>6</sup>

<sup>1</sup>Lyndon B. Johnson School of Public Affairs, University of Texas at Austin, Austin, TX, United States, <sup>2</sup>Office of Sustainability, Austin, TX, United States, <sup>3</sup>East Carolina University, Greenville, NC, United States, <sup>4</sup>Go! Austin/Vamos! Austin, Austin, MN, United States, <sup>5</sup>Department of Civil, Architectural and Environmental Engineering, Cockrell School of Engineering, The University of Texas at Austin, Austin, MN, United States, <sup>6</sup>Jackson School of Geosciences, The University of Texas at Austin, Austin, TX, United States

The growing frequency and intensity of extreme weather events have placed cities at the forefront of the human, social, economic, and ecological impacts of climate change. Extreme heat, extended freeze, excessive precipitation, and/or prolong drought impacts neighborhoods disproportionately across heterogeneous urban geographies. Underserved, underrepresented, and marginalized communities are more likely to bear the burden of increased exposure to adverse climate impacts while simultaneously facing power asymmetries in access to the policy and knowledge production process. Knowledge co-production is one framework that seeks to address this convergence of disproportionate climate impact exposure and disenfranchised communities. Co-production is increasingly used in sustainability and resilience research to ask questions and develop solutions with, by, and for those communities that are most impacted. By weaving research, planning, evaluation, and policy in an iterative cycle, knowledge and action can be more closely coupled. However, the practice of co-production often lacks reflexivity in ways that can transform the science and policy of urban resilience to address equity more directly. With this, we ask what kind of co-production mechanism encourage academic and non-academic partners to reflect and scrutinize their underlying assumptions, existing institutional arrangements, and practices? How can these efforts identify and acknowledge the contradictions of co-production to reduce climate impacts in vulnerable communities? This paper presents a framework for reflexive co-production and assesses three modes of co-production for urban resilience in Austin, Texas, USA. These include a multi-hazard risk mapping initiative, a resident-driven community indicator system for adaptive capacity, and a neighborhood household preparedness guide. We establish

a set of functional and transformational criteria from which to evaluate co-production and assess each initiative across the criteria. We conclude with some recommendations that can advance reflexive co-production for urban resilience.

#### KEYWORDS

**social vulnerability and vulnerable populations, co-production and co-learning, multi hazard vulnerability, climate adaptation, urban resilience**

## Introduction

Research on urban resilience and urban systems has exponentially increased in recent years (Caldarice et al., 2019). This includes advancements in the fields of urban ecology (Rademacher et al., 2019), urban social-ecological systems (Crowe et al., 2016), and hazard and risk reduction (Xue et al., 2018). Global trends highlight the importance of understanding urbanization and climate change as converging issues that create multifaceted challenges that span multiple scales (Bai et al., 2017). Climate-related impacts—biodiversity loss, greenspace degradation, flooding, wildfire, extreme heat, among others—cause damage and loss to property, infrastructure, livelihoods, service provision and environmental resources. Climate change is likely to further increase the exposure in cities to climate impacts by affecting the magnitude, frequency and spatial distribution of disastrous events (Field et al., 2012; Orimoloye et al., 2019; González et al., 2021).

One promising path to mitigating climate-related hazard exposure is through knowledge co-production (Iwaniec et al., 2020; Cook et al., 2021; Amorim-Maia et al., 2022). The process of co-production is an increasingly utilized framework to generate usable knowledge by linking knowledge production and application by science, practice, and policy actors working together (Wyborn et al., 2019; Norström et al., 2020). More broadly applied, co-production is a way to produce new knowledge with a clear normative objective to support societal change (Wyborn et al., 2019). Extending a notion of reflexive governance put forth by Dryzek and Pickering (2019), we consider a “virtuous cycle” that includes three iterative phases—recognize, reflect, and response—as a positive feedback loop for urban resilience. This *reflexive co-production* can encourage actors to scrutinize and reconsider their underlying assumptions, institutional arrangements, and practices (Dryzek and Pickering, 2019; Van der Jagt et al., 2021) and move from “managing” the intersection of equity and urban resilience toward transforming community-academic-municipal government interactions. We use reflexive as a co-production adjective to emphasize a process for different actors to critically consider different ways of knowing and addressing specific problems and solutions. This is a deliberative effort to get closer to the cognitive and social patterns in the practice

of science and become more attuned to the nuances and assumptions brought from the different research, policy, and community perspectives (Merton, 1987; Latour, 1991). When climate modeling, social science, lived experience, city policy and nonprofit programs integrate, a reflexive approach to co-production is warranted.

This paper focuses on a framework for reflexive co-production and assesses three modes of co-production for urban greening and climate impact risk reduction in Austin, Texas, USA. In 2013, the Austin City Council passed a resolution (#20131121-060) that directed the city manager and staff to analyze climate change projections, determine how departmental planning efforts integrate future impacts of climate change, and identify a process for performing department vulnerability assessments. Numerous efforts since then—publishing a “Climate Resilience Action Plan for City Assets and Operations” (2018), establishing “Climate Ambassadors” (2020), publishing a “Climate Equity Plan” (2021), and hiring a Chief Resilience Officer (2022)—are demonstrable efforts toward climate mitigation and adaptation in the City. Over this time period, communities in southeast Austin experienced a sequence of consequential floods (2013, 2015, and 2017) impacting many homes, lives and livelihoods. Community groups, such as Go Austin/Vamos Austin (GAVA), responded by organizing the community to increase preparedness and resilience to climate impacts through engagement, advocacy, and public accountability strategies. Concurrently, the Austin Area Sustainability Indicators (A2SI, [austinindicators.org](http://austinindicators.org)) at the LBJ School of Public Affairs at the University of Texas-Austin began focusing on climate vulnerability and community resilience (Bixler et al., 2021b; Bixler and Jones, 2022).

These (eventually) intersecting efforts create a foundation for co-production of urban resilience in Austin. This manuscript traces the interactions and processes that intertwined researchers, city agency staff, and community groups through a lens of reflexive co-production. We structure this paper as follows. First, we lay out a conceptual framework for reflexive co-production and utilize existing co-production research to think critically about the “different modes” of co-production. Specifically, we describe three co-production initiatives in Austin (Figure 1)—multi-hazard risk mapping, adaptive

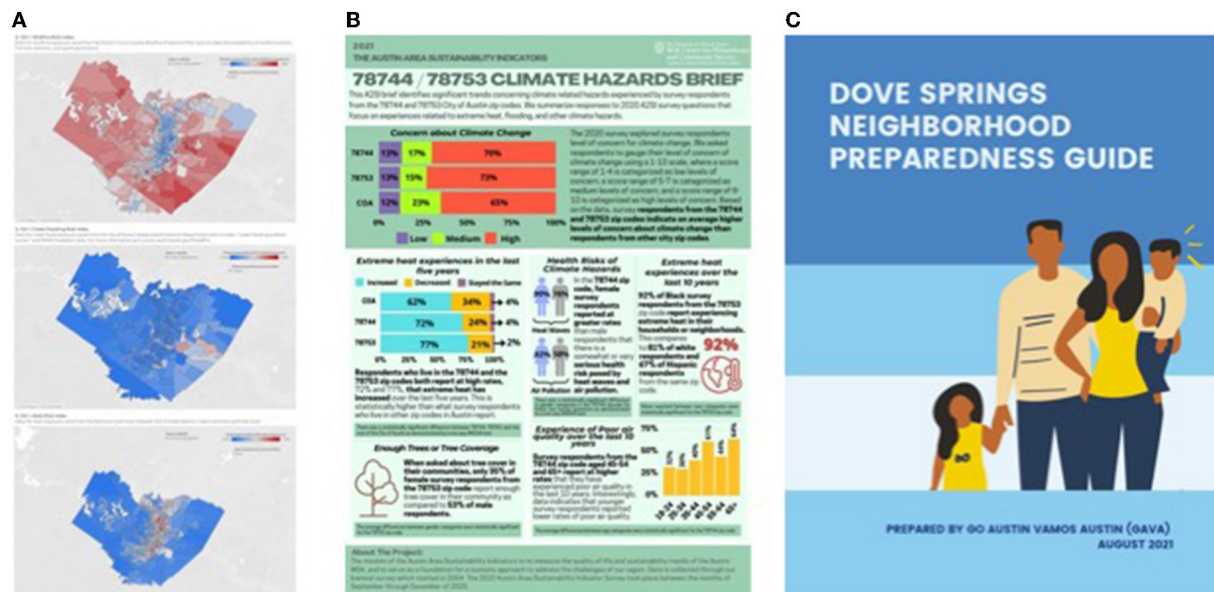


FIGURE 1  
Three co-production initiatives in Austin: (A) multi-hazard risk mapping, (B) adaptive capacity indicators, and (C) neighborhood preparedness guide.

capacity indicators, and neighborhood preparedness plan – and examine those cases through a set of criteria distilled from recent co-production scholarship. Next, we identify some co-production contradictions, as well and highlight insights through a lens of reflexive co-production that offer practical insights for urban resilience scholarship and practice.

## Study area

Austin is an economically diverse and growing city in central Texas at the edge of the Edwards Plateau and the Texas Hill Country. The 11th-largest city in the United States, Austin has an estimated population of 1,026,833 residents in 2021. The Austin Metropolitan Statistical Area (MSA), as defined by the U.S. Office of Management and Budget, includes five counties (Bastrop, Caldwell, Hays, Travis, and Williamson) and over two million people, making it the 29th largest metropolitan area in the United States. Robust population and economic growth since 2000 have increased the tax base and made Austin an attractive city for technology start-ups and established corporations alike. Major technology companies such as Facebook, Google, Apple, Tesla, Oracle, and Samsung have invested a combined >\$10 billion in new manufacturing facilities and office space since 2017. Economic opportunities are matched by increasing challenges like housing unaffordability, inequitable access to services and infrastructure driven by neighborhood displacement, and increasing consumption of water and land (Richter and Bixler, 2022). This is compounded

by climate projections that point to a higher intensity flood-drought regime in the region impacting human health and urban ecosystem services. Climate models show that average temperatures are increasing, the risks associated with extreme temperatures are more pronounced, and precipitation patterns are shifting, with an increase frequency in heavy precipitation and droughts (Banner et al., 2010). Historically underserved and economically marginalized communities are disproportionately impacted (Busch, 2017; Zoll, 2021).

As with many major U.S. cities, Austin's history of economic and housing segregation and broader systemic racism continues to shape the adaptation pathways and vulnerability of some neighborhoods to heat waves, drought, flooding, biodiversity loss, and wildfires. Historically marginalized communities – typically residing in a geography referred to as the “eastern crescent” of the northeast, east and southeast portions of Austin – are already stressed by limited resources, growth pressures, and higher rates of chronic disease. These social and institutional conditions define differential sensitivities and underpin disparate climate impacts across Austin's communities.

## Urban resilience and co-production

### Climate impacts and community resilience

Our research is situated in a literature base that is diversified, growing and evolving, and spread across many disciplines

focused on urban and community resilience (Aldrich and Meyer, 2015; Brunetta et al., 2019; Caldarice et al., 2019; Scherzer et al., 2019), adaptive capacity in relation to hazard preparedness (Pfefferbaum et al., 2013; Onuma et al., 2017; Siders, 2019; Bixler et al., 2021a), and vulnerability (Cutter et al., 2003; Adger, 2006; McDowell et al., 2016; Flanagan et al., 2018). The intersection of climate-related hazards, social vulnerability, and urban communities has become a central component of an international climate change adaptation research and policy agenda (Siders, 2019; Nalau and Verrall, 2021; Shi and Moser, 2021). Comprehensive frameworks for research and/or policy are lacking, but the common thread is clear: these areas of inquiry seek to increase community resilience by reducing climate impact exposure, decreasing sensitivity of households and communities to climate impacts, and/or increasing community adaptive capacity to mitigate the severity and intensity of climate-related disasters.

Climate extremes are increasing and intensifying loss of greenspace and biodiversity, heatwaves, droughts, wildfires and major flood events. To address this, researchers are “connecting climate extremes” (Raymond et al., 2020) through multi-risk assessments (Gallina et al., 2016) to improve understanding of disaster risk in all its dimensions (UNISDR, 2015). These concepts emphasize the increasing likelihood of climate-related compounding events, which are nonlinearly influenced by non-physical factors such as exposure and vulnerability and cut across decision-making levels from household, neighborhoods, informal and formal governance networks, and across society. Referred to as interacting, cascading, or multi-risk hazards (Pescaroli and Alexander, 2018), the framing emphasizes the interacting physical and social factors that cause their impacts to be amplified relative to the same hazard occurring separately (Raymond et al., 2020). Multi-hazard risk assessments and mapping are a tool to quantify hazard exposure and sensitivity of population to multiple climate related shocks and stressors (Adger, 2006; Pielke et al., 2021).

In addition to exposure and sensitivity, adaptive capacity is another dimension of vulnerability and urban resilience frequently considered in the literature (Pfefferbaum et al., 2013; Elrick-Barr et al., 2014; Bixler et al., 2021a; Shi and Moser, 2021; Bixler and Jones, 2022). Climate impacts are most acutely experienced at the household scale and thus increasing adaptive capacity of households is both a short-term necessity in hazard prone neighborhoods and a critical long-term hazard risk reduction strategy. Interdisciplinary frameworks and methods to measure adaptive capacity are accumulating and accelerating, but progress remains fragmented and lacking consensus (Siders, 2019). Generally speaking, attempts to operationalize adaptive capacity refer to a vector of resources and assets that can be economic, social, informational, and/or community oriented (Adger and Vincent, 2005; Norris et al., 2008; Elrick-Barr et al., 2014; Barnes et al., 2020). Adaptive capacity is a key part of the climate vulnerability equation (Adger,

2006) because increasing adaptive capacity can counteract population sensitivity and/or hazard exposure and increase community resilience.

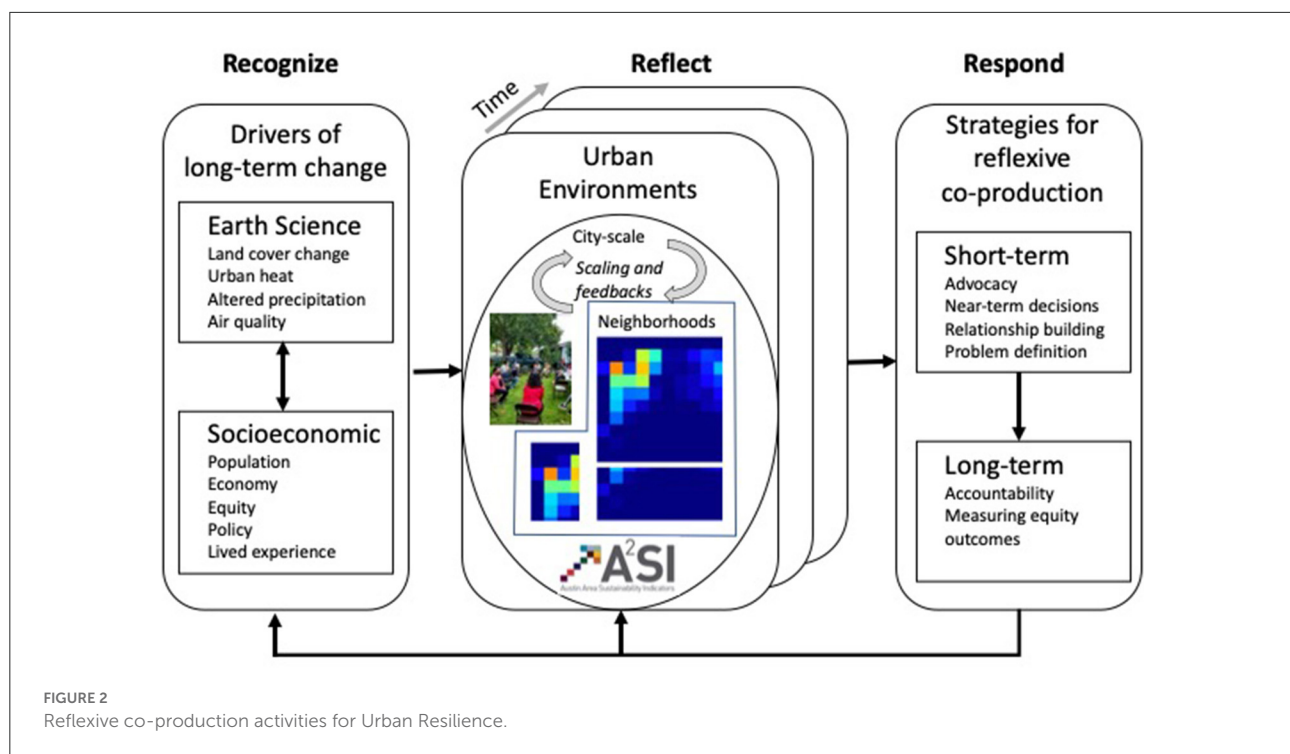
Beyond measurement, there is an increasing emphasis and reliance on a hyper-local scale, whole community approach for effective emergency management to occur (FEMA, 2011; LaLone, 2012; Jones, 2022). Preparedness plans at the neighborhood level are one example of hyper-local scale emergency management. These plans can encourage neighborhood mapping activities, support the identification of local resources, assets, and neighborhood vulnerabilities. In theory, neighborhood preparedness plans can create opportunities for shared understanding of community risks, needs, and capabilities (FEMA, 2011) in ways that strengthen a community's resilience to climate impacts.

## Co-production in a climate impact context

The thrust of community resilience and climate impact scholarship emphasize that cross-sector and interdisciplinary collaborations are critical for determining feedbacks between physical processes and societal decisions (Raymond et al., 2020) and that deep integration of knowledge bases, or convergent research, is necessary for addressing social, economic, environmental, and technical challenges of hazards (Peek et al., 2020). Co-production is a framework to address the complex nature of contemporary sustainability challenges by bringing together knowledge from academics and non-academics (Norström et al., 2020). It is a process to overcome the known barriers of knowledge use, in particular the lack of credibility, legitimacy, and relevance to decision making (Cash et al., 2003). The current concept – converged from public administration, science and technology studies, and sustainability studies – suggests that for knowledge to be actionable, the production of science should occur through scholars and stakeholders interacting to define important questions, identify relevant evidence, and co-create convincing forms of argument (Miller and Wyborn, 2020). More broadly applied, co-production is a way to produce new knowledge with a clear normative objective to support societal change (Wyborn et al., 2019). Norström et al. (2020) suggest focusing on four principles for successful co-production: context-based, pluralistic, goal-oriented, and interactive.

Urban resilience offers a somewhat unique context from which to assess the utility and impact of co-production. Earth system science that underpins climate impact research has a natural science tradition that, until recently, has had little community engagement or associated social science (Gill et al., 2021). Hazards research, particularly as it is related to climate change, can be politically polarizing and





the typical emergency frames used to discuss climate-related hazards have varied political effects (Patterson et al., 2021). Challenges of modeling uncertainty, risk communication, and risk perception further complicate how scientists from different disciplines and non-scientists interact (Lejano et al., 2021), but important frameworks have been developed that help us think co-production interactions and processes in urban systems (Frantzeskaki and Kabisch, 2016; Muñoz-Erickson et al., 2017; Iwaniec et al., 2020, 2021; Cook et al., 2021;). For example, Muñoz-Erickson et al. (2017) present a framework for a knowledge systems analysis that guides description and analysis of knowledge and governance interactions in cities, and Frantzeskaki and Kabisch (2016) show how policy and science learning was linked to governance capacity in Berlin and Rotterdam. There are efforts to empirically ground existing empirical frameworks at this intersection of hazards research, risk reduction, and co-production (see Davies et al., 2015; Lejano et al., 2021), as well as a growing interest in collaborative or participatory hazard modeling (Jordan et al., 2018; Minucci et al., 2020; Sanders et al., 2020).

To many, co-production has become 'gold standard' of engaged science, though not without critique (Lemos et al., 2018). Co-production often takes time and money to develop the necessary trust to not only for together in a knowledge generating process but also to act afterwards. Important and significant questions have been raised regarding the politics of co-production and questioning if processes reinforce, rather than mitigate or transform, unequal power relations (Jagannathan et al., 2020; Turnhout et al., 2020; Chambers et al., 2021). Moreover, non-academic partners may

experience partnership fatigue as scientists privilege familiarity over uncertainty of new partners or issues (Porter and Dessai, 2017). With these opportunity costs in mind, we ask what kind of co-production mechanism encourage academic and non-academic partners to reflect and scrutinize their underlying assumptions, existing institutional arrangements, and practices? How can these efforts identify and acknowledge the contradictions of co-production to reduce climate impacts in vulnerable communities?

## Reflexive co-production as a guiding framework for assessing co-production efforts

We emphasize a reflexive co-production process that iterates through three phases: Recognize, Reflect, and Respond. We outline the various activities that fit within these phases in Figure 2.

We set out functional and transformational criteria for assessing co-production in urban resilience context. Functional criteria are related to process and suggest (i) value-oriented indicators that include dimensions of being (ii) context-based, (iii) pluralistic, (iv) goal-oriented, and (v) interactive (Norström et al., 2020). Context-based suggests that co-produced science should be situated within the particular social, ecological, and technical (SET) context in which they are embedded (Bixler et al., 2019b; Chang et al., 2021). Pluralistic recognizes the multiple ways of knowing, whereas goal-oriented refers to

a clearly defined and shared goals. Finally, the interactive principle acknowledges that co-production requires frequent interactions among participants throughout the process, from framing the research problem to interpreting results (Bixler et al., 2019a). Interaction throughout the process builds trust between participants, which increases the likelihood that resulting knowledge is perceived to be credible, salient, and legitimate (Cash et al., 2003). These four normative principles, if successful, lead to pragmatic, proximate, and long-term outcomes such as expanding awareness, knowledge, increasing capacity, and overcoming the barriers to knowledge utilization (Wyborn et al., 2019). This is particularly relevant and true for hazards research where significant barriers exist to effective risk communication and explicit calls have been made for increased cultural competencies among disaster risk managers (Knox, 2020; Fakhruddin et al., 2022).

Transformational criteria move beyond functional outcomes to assess how power and politics are accounted for in co-production (Turnhout et al., 2020). As a result, co-production that is transformational will establish long-term changes beyond the single intervention and empower relatively marginalized groups in the decision-making process. This moves beyond recognition and integration of local perspectives into the knowledge process and toward establishing new institutions or systems within existing institutions (Chambers et al., 2021). Transformational co-production prioritizes marginalized social concerns over technocratic solutions, explicitly integrates social equity into a climate and hazard risk reduction agenda, and changes the relationship between science, policy, and practice (Lemos et al., 2018; Wyborn et al., 2019; Turnhout et al., 2020; Chambers et al., 2021).

## Applying different modes of co-production in Austin

### Three urban resilience co-production initiatives in Austin

In this section, we describe three co-production initiatives applied in the City of Austin these are (i) multi-hazard risk mapping, (ii) adaptive capacity indicators, and the creation of a (iii) neighborhood preparedness plan. Ahead in “Discussion and conclusion” we will examine those cases through a set of criteria distilled from recent co-production process. We identify some co-production contradictions as well and highlight insights that can inform co-production processes in hazard risk reduction scholarship and practice. The background on the different projects is outlined in Table 1 and discussed next.

#### Multi-hazard mapping

The multi-hazard risk mapping project was a collaboration between academic and non-academic researchers from City

of Austin agencies, as well as policy and program staff from the City of Austin (Bixler et al., 2021b). The aim of the project was to spatially map and aggregate multiple climate-related hazards – flood, heat, and wildfire – and combine those hazards with a measure of social vulnerability. The product of combining multiple types of climate impact exposure plus social vulnerability (population sensitivity) was a normalized multi-hazard risk score that City staff had for possible consideration in making resource allocation and community-engagement decisions. The activities were driven by City program staff who helped co-ordinate the data sharing between the academic research team and agency scientists.

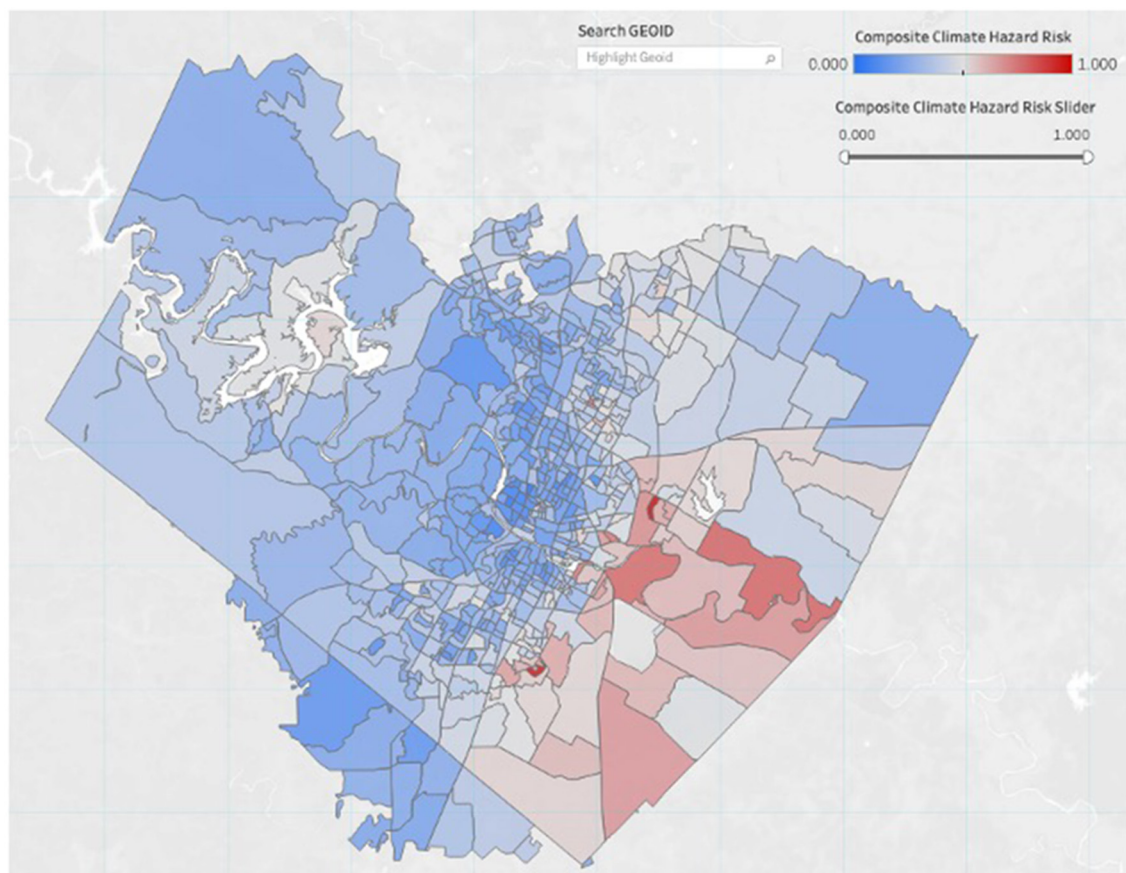
The academic research team conducted the analysis, which included utilizing 18 variables from the U.S. Census 2013-17 American Community Survey (ACS) to construct a unique social vulnerability index (SVI) solution for Austin. Our index, although specific to the Austin area, followed established workflows and principal component analysis techniques of the established SVI (Flanagan et al., 2011, 2018) and SoVI (Cutter et al., 2003; Cutter and Finch, 2008). The exposure indices for flood and wildfire were constructed with data from the City of Austin and used established techniques in the respective fields, whereas the heat exposure score was developed using the Urban Imperviousness and Tree Canopy layers of the 2016 National Land Cover Database (Yang et al., 2018). Upon completion of the analysis, the results were discussed and verified with the City of Austin scientists and program staff and subsequently shared through the Austin sustainability indicators portal [in Figure 3 the red indicates areas with a higher composite score of exposure (to flood, heat, wildfire)] combined with social vulnerability, and available online at: <https://tinyurl.com/2mme4krm>.

#### Adaptive capacity indicators

The community indicators for adaptive capacity effort were co-developed *via* collaboration between the academic research team and Go! Austin Vamos! Austin (GAVA), a grassroots community nonprofit. GAVA organizes and mobilizes community feedback to reduce barriers to health while increasing institutional capacity to respond to the people most impacted by historic inequities. GAVA works with Austin communities to build climate resilience, among other activities such as improving nutrition, increasing physical activity, and supporting neighborhood health. This project linked GAVA strategies and actions to community indicators around resilience and adaptive capacity collected by the Austin Area Sustainability Indicators (A2SI). A biennial community survey is conducted as part of A2SI, dating back to 2004 with subsequent waves of data collection in 2006, 2008, 2010, 2015, 2018, and most recently 2020. Prior to the 2020 data collection, the research team worked with GAVA staff to co-design indicators for adaptive capacity. These indicators were informed through an iterative and pluralistic process by residents in GAVA's service area zip

TABLE 1 Summary of the three urban resilience co-production initiatives.

	Phase of reflexive co-production	Project initiated by	Collaborating partners	Funding	Co-production Activities	End-users	Products
<b>Multi-hazard mapping</b>	Recognize	City of Austin, Office of Sustainability	City of Austin agencies: Office of Sustainability, Watershed Protection, Austin Wildfire	None	Model conceptualization; data sharing; analytical verification; reporting design	Austin City Council; City of Austin agency staff	A spatially explicit map, interactive visualization with information at Census Block Group.
<b>Adaptive capacity indicators</b>	Reflect	Academic researchers	Academic researchers, GAVA	Funding to academic researchers for data collection from a philanthropic funder of GAVA	Resident's input, verification	City of Austin staff, GAVA staff and other engaged nonprofits	Creation of survey items, indicators, and measurement strategies that are resident driven.
<b>Household preparedness guide</b>	Respond	GAVA	Academic research team, city staff, GAVA staff	Funding from the COA to support formatting and publication of guide	Information sharing	Residents	A digital and printed preparedness guide



**FIGURE 3**  
Multi-hazard risk index for Austin, Texas (for more on this see [austinindicators.org](https://austinindicators.org) and Bixler et al., 2021b).

codes: 78744 and 78753 (two historically underserved zip codes in Austin, Figure 4).

This interactive process began with the research team (1) conducting a literature review of community resilience indicators and principles. The research team presented to GAVA community organizers how to identify themes and indicators to support GAVA's mission and (2) GAVA community organizers, in collaboration with the research team, developed questions for GAVA staff to discuss with residents to identify what metrics are important to the community. GAVA community organizers then hosted 23 conversations with residents that took place in June of 2020. Notes from these conversations were translated (roughly two-thirds of the conversations occurred in Spanish) and transcribed. The research team coded the community conversations for key themes as they related to adaptive capacity and resilience, discussed those key themes with GAVA community organizers, cross-walked existing A2SI survey questions against those key themes, identified gaps, designed new and additional survey questions, and then brought the new survey items back to GAVA community

organizers for review and revision. From this collaborative work, twenty-eight additional survey questions were added to the survey representing approximately 30% of the survey questions asked (not including demographic questions). In 2020, the A2SI survey data collection utilized an oversampling procedure to secure a sufficient sample size in 78744 and 78753 zip codes to reduce the margin of error in those geographies to  $\pm 5\%$ .

### Household emergency preparedness guide

Concurrently, the academic research team coordinated with GAVA and the City of Austin to develop an emergency preparedness guide, in both English and Spanish, for the Dove Springs area (zip code 78744). The Dove Springs area has been historically impacted by major flood events. The research team served a dual role in the creation of the guide. They conducted background research, where they helped identify the types of content typically found in neighborhood preparedness guides. Additionally, the research team served as project manager, where



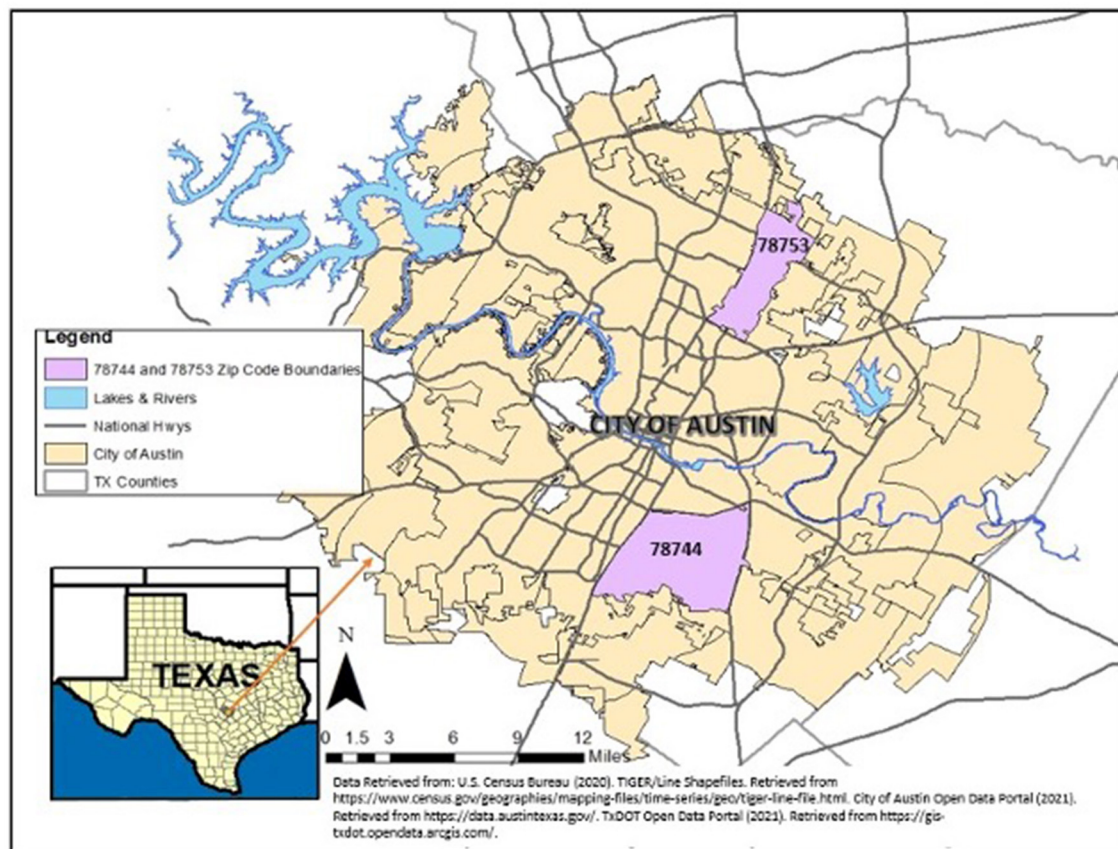


FIGURE 4

Zip codes where GAVA activities are focused and adaptive capacity indicators were developed.

they helped with the curation of the information in the guide and moving the document from draft to publication.

GAVA staff and community organizers helped identify content for the guide based on their trainings they have developed and implemented to grow a network of neighborhood “climate navigators.” Working with residents and researchers they also supported the guide by ground-truthing the guide’s content with residents, revised content accordingly, and revised the Spanish language version to make the guide more accessible/understandable. Meanwhile, City of Austin staff provided information about public resources available to assist in preparing for hazards and provided financial resources for publication, supported Spanish translation of the content, and helped design the guide. Although these efforts are clearly connected to the first two initiatives, how insights or information from those co-production efforts found their way into the guide was not explicit or systematic (guide available for viewing here: <https://tinyurl.com/2nptmrd2>). Since the creation of the Dove Springs guide, the City has used the guide as a template for a City of Austin-wide neighborhood preparedness guide.

## Discussion and conclusion

The different threads of co-production started in January 2019 and are currently ongoing as of October 2022. The time period of activities was significantly impacted by the COVID-19 pandemic in terms of mediums for interaction, methods of data collection and analysis. There were also shifting priorities of both individual personnel and respective organizations as the pandemic ebbed and flowed. We organize the discussion as a linear assessment of each of the co-production criterion, while acknowledging the non-linear interaction effects of the various projects and criterion on the interactions among participants. We highlight the complexities of co-production where activities serve multiple functional outcomes and then draw some insights for urban resilience reflexive co-production.

## Functional criterion

As described earlier, co-production scholarship supports the following four criteria for successful co-production: context-based, pluralistic, goal-oriented, and interactive (Wyborn et al.,

2019; Norström et al., 2020; Chambers et al., 2021; Zurba et al., 2021). We refer to these as functional criteria as they provide normative principles of what high quality and successful knowledge co-production “should be” (Norström et al., 2020). To varying degrees, all Austin initiatives intended to generate local, place-based information, was pluralistic, goal-oriented, and interactive.

By mapping the spatial variation across the city, the multi-hazard mapping project was context-based (focused on identifying the variation of exposure and sensitivity of census block groups within a specific municipal scale) and utilized city-generated data. The project was initiated by staff from the City of Austin Office of Sustainability, who openly acknowledge that municipalities can no longer rely solely on traditional public participation processes and data from historic climatic events to determine future impacts from extreme weather. The goal-orientation of this project was clear from the start – influence policy and steer community engagement interventions being designed by City staff, GAVA and other nonprofits, as well as through course-based work at the University.

In many ways, the intended outcomes of this effort matched the achieved outcomes of this project. City staff have found the maps a useful tool in highlighting geographical areas of concern that need more investigation and the mapping outputs have been used as an object around which on-going co-production occurs. For example, the GAVA-City-University team used the quantified and visualized multi-hazard risks as a focal point for responding to a request for proposals. Multiple proposals have received federal funding (NOAA, NASA) and the team is implementing a grant-funded, GAVA-led community engagement effort in areas of high social vulnerability and high hazard risk. Financial resources from the federal grants are also going to GAVA and the community to support the engagement efforts. In this sense, the maps served as a useful boundary/research object (Lang et al., 2012) providing a platform to scaffold and co-design new research and community engagement strategies.

This project, however, was less pluralistic and interactive than it *should be*. The hazard exposure and social vulnerability modeling utilized traditional disciplinary methods. Limited input from the community was provided in shaping measurement of hazard exposure, social vulnerability, or the multi-hazard index. By contrast, other efforts at mapping social vulnerability have documented pluralistic and interactive approaches with communities (Lavoie et al., 2018; Rickless et al., 2020).

By comparison, the adaptive capacity indicators project was context-based and more pluralistic and interactive than the mapping project, however, less goal-oriented. The iterative process employed a GAVA-requested and academic team led literature review, GAVA-led interviews to identify “what is important to measure” for residents, community organization and academic team co-design of new survey items, and then

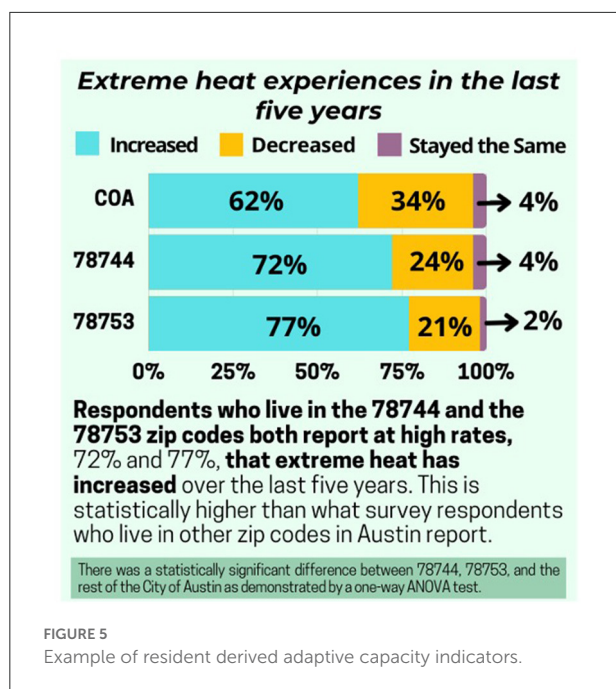
circling back to the community members for review of the new survey items language. This effort was pluralistic in that representatives from the community organization (GAVA) and residents, many of whom were from predominantly Spanish speaking households, directly defined what was important to measure, thus steering the data that were collected. The intention of this process was to empower the voices of relatively marginalized actors in shaping the indicators that pointed to adaptive capacity for those who experience frequent floods and extreme heat. The resident-generated questions broadly fit within three primary themes: gentrification and resident displacement, environmental quality, and barriers/opportunities for community organizing. An example question of each of these three themes include:

- Neighbors I’m close to have been forced to move away (four-point Likert response from strongly disagree to strongly agree);
- Trees or tree cover in my community (five-point Likert from “a considerable shortage” to “more than enough”)
- What are the barriers to getting organized in your neighborhood (open response)?

These and the related questions provide insight into the multiple dimensions of community resilience, broadening the scope of our previously identified and literature-based set of community indicators for adaptive capacity. Analysis of the information yielded interesting comparisons of residents of the zip codes of interest and identified strengths and gaps relative to other Austin residents. After data were collected from the 2020 A2SI survey, the research team worked with GAVA to co-design and co-develop research briefs and figures with the survey data. Once briefs were created, the information was reported back to GAVA and the City of Austin staff and interactive data sessions were conducted with community organizers and residents. Figure 5 provides an example of the data visualizations co-created and designed by the research team and GAVA, demonstrating a difference in experience of extreme heat in the specific underserved communities in relation to other zip codes in the City of Austin.

The community organization and academic team co-designed the problem frame and scope on community indicators for adaptive capacity initiative. However, the scope of resident participation was predetermined by the ongoing nature of the research and the problem framing of the project already established by the academic research team. These issues of uneven power relations have been previously identified in the literature (Turnhout et al., 2020).

In contrast to the multi-hazard mapping, the goal orientation of the adaptive capacity indicators was less well-defined. The broader framing of the project was set to establish baseline measurements as part of an ongoing, biennial, effort to track a broad range of sustainability and community resilience



indicators in the service area zip codes and across the city. The research team struggled to identify how to best represent the data as visualizations and how and when to test for statistical significance (and if it was important in this context). Additionally, making a direct connection between utilizing the survey data for program and organizing strategies has been challenging to implement.

Weaving components of both projects was the effort to develop a neighborhood preparedness guide. The resident hazard preparedness guide is context-specific information tailored for residents of one specific zip code – 78744 – that experiences frequent and intense flooding and extreme heat events. The effort was coordinated by the academic research team yet was pluralistic in that it compiled the most up-to-date resources from the city and cross-referenced with residents the knowledge needs as articulated by the residents. This effort was extremely context-based and goal-oriented. From project initiation to completion, the project aim was developing a resident-centered guide that GAVA could utilize in public information and training workshops that are conducted in that specific neighborhood. Since published, the guide has been distributed to residents through GAVA's climate navigator program. In 2022, the research team, GAVA, and the City of Austin collaborated to create a city-wide neighborhood guide.

## Transformational criterion

A meta-analysis of co-production identified two distinct ways that co-production efforts engage with politics:

empowering relatively marginalized groups or by influencing powerful actors (Chambers et al., 2021). The multi-hazard mapping project provided municipal officials science-based evidence to inform decision-making. Improved and refined technical modeling of hazards is of little use if not embedded in the policy, regulatory, institutional, and cultural factors in which hazard mitigation and preparedness occurs. Implicitly, the effort sought political engagement, with the intentions to highlight the unequitable distribution of hazards among historically marginalized neighborhoods in the city. The report was acknowledged by the Austin City Council and has shaped decisions and strategies at various municipal department levels. The initiative was an effort to reframe the solution set: city leadership and staff were challenged to move from resilience planning of municipal assets to communities made up of households with residents. This project generated a method and evidence to understand social vulnerability and the spatial relationship to various climate-related hazards.

The maps – the social vulnerability map in particular – highlighted the legacy of racial and economic disparities between east and west Austin institutionalized through racial segregation in the 1928 City Master Plan. Many of the once racially segregated neighborhoods are identified as “hot spots” for climate-related risk identified in the multi-hazard mapping, providing evidence and justification for ongoing City-led community engagement and climate adaptation efforts, a response to previous efforts being “color-blind” (Zoll, 2021).

The contradictions of the functional criteria, however, also created barriers for transformational policy and engagement. The information generated from the multi-hazard mapping project is “context-based” at the municipal level, yet too coarse for understanding street or household level variation within neighborhoods. The decision to map at the scale of the census block group was driven solely on the methodological considerations of census data availability used for the social vulnerability index. Social vulnerability and flood exposure may vary significantly within a census block group and our current approach, which accounts for geographical variation at one scale, does a poor job at finer scales. This has been a point of critique from the community organizations when conversations extend beyond researchers and city program staff. To this end, the project engaged with top decision-makers and advanced existing policy goals, although has not yet shifted institutional or management practices. This initiative did little to directly empower relatively marginalized actors, articulate, or mobilize the voices, knowledge or perceptions of different participants or address institutions of decision-making or governance.

The adaptive capacity indicators, in contrast, sought to integrate resident perspectives into the indicator design process and empower relatively marginalized voices to create more meaningful representations of what is important to measure and track. This effort sought to increase the knowledge base and issue awareness of resident-defined



metrics, thus creating opportunities for those most affected to redefine the range of climate adaptation solutions to include anti-displacement/gentrification, opportunities for political engagement, and broader environmental quality. That said, there was little space created to redefine the process and/or transform the broader system of governance, knowledge production processes, or strategies for delivering hazard mitigating related services. Moreover, the community indicators for adaptive capacity effort present another functional co-production contradiction. On the one hand, this initiative empowers resident voices to shape what outcomes are important and what should be measured and reported, yet the data collection and analysis of indicators treats the residents and resident information as the object of research through deductive data collection, analysis, and reporting. Other relativistic and/or systems thinking designs could bring resident voices closer to academic and city staff for more direct conversations, and transformations, of systemic governance issues.

The neighborhood preparedness guide brought City staff, community organization staff, academic researchers together to generate and compile information for residents. To date, there is little evidence this has shifted the strategies or priorities of decision-makers or led to changes in resident preparedness. There are plans for the guide to be a focal point in community workshops led by GAVA as part of their “climate navigator” efforts to increase neighborhood preparedness capacity. Similar to the hazard maps, this guide has the potential to serve as a boundary object for creating safe spaces to identify the governance barriers and opportunities for better climate preparedness at hyper-local scales.

## Conclusion

The three initiatives discussed were constituted by overlapping set of actors (academic, community, city partners) across the same period. The various threads of interaction have been necessary to build trust between the participants and provided opportunities to continue various co-production processes beyond the delivery of the final products from the projects reported here. Interestingly, early co-production scholarship focused on service delivery (Brudney and England, 1983), however the recent renaissance in science and technology and sustainability studies has significantly focused on knowledge creation and utilization. We find that functional and transformational co-production in a hazards context generates knowledge, reduces barriers to knowledge utilization in designing solutions or services, but importantly also should involve the co-production of public goods service delivery. This is the “respond” phase of the reflexive co-production cycle and points toward the iterative virtuous cycle of building urban resilience.

There are multiple pathways through which reflexively responding can occur: reducing hazard exposure, reducing

population sensitivity, and/or increasing adaptive capacity (Adger, 2006). In all cases, functional and transformational co-production needs to account for the mix of services and products as part of the output of co-production (Alford, 2014). What green or gray infrastructure services reduce exposure to hazards? What social services reduce population sensitivity? What program interventions increase adaptive capacity? These are future studies that currently underway by the academic, city staff, and community organization team. In all cases, municipal and community organization partners design and deliver climate services with the intended outcome to increase community resilience. Reflexive co-production, when applied to urban resilience initiatives, can more explicitly connect knowledge and service co-production through the recognize, reflect, and respond cycle.

## 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

RB contributed to writing, analysis, and research design. SR contributed to writing. MC contributed to writing, different initiatives, and work on the ground. JJ contributed to different initiatives and engagement with GAVA. NA and CL contributed to different initiatives and work on the ground. MB contributed to initiatives and community engagement. PP and DN contributed to research design and funding acquisition. All authors contributed to the article and approved the submitted version.

## Funding

This study was supported by the NOAA Climate Program Office's Extreme Heat Risk Initiative, Cooperative Agreement NA21OAR4310146; NASA's Earth Science Division Equity and Environmental Justice program (grant number 80NSSC22K1675); NASA IDS 80NSSC20K1262 and 80NSSC20K1268; NSF's Cultural Transformation in the Geoscience Community program (grant number 2228205); DOE's Integrated Urban Field Laboratory program; and the University of Texas Bridging Barriers initiative Planet Texas 2050.

## Acknowledgments

RB, MB, DN, and PP acknowledge support from Planet Texas 2050, a Bridging Barriers Initiative at the University of



Texas at Austin. RB and JJ acknowledge Auva Shariatmadari for her work with GAVA. RB acknowledges Euijin Yang for his work on the hazard mapping. DN acknowledges William Stamps Farish Chair endowment through the Jackson School of Geosciences at the University of Texas at Austin.

## Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships

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## OPEN ACCESS

## EDITED BY

Eric M. Wood,  
California State University, Los Angeles,  
United States

## REVIEWED BY

Neil Gilbert,  
Michigan State University, United States  
Gregory Blair Pauly,  
Natural History Museum of Los Angeles  
County, United States

## \*CORRESPONDENCE

Joscha Beninde  
✉ beninde@ucla.edu

†These authors have contributed equally to this work and share second authorship

## SPECIALTY SECTION

This article was submitted to  
Urban Ecology,  
a section of the journal  
Frontiers in Ecology and Evolution

RECEIVED 30 June 2022

ACCEPTED 15 March 2023

PUBLISHED 06 June 2023

## CITATION

Beninde J, Delaney TW, Gonzalez G and  
Shaffer HB (2023) Harnessing iNaturalist  
to quantify hotspots of urban biodiversity:  
the Los Angeles case study.  
*Front. Ecol. Evol.* 11:983371.  
doi: 10.3389/fevo.2023.983371

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# Harnessing iNaturalist to quantify hotspots of urban biodiversity: the Los Angeles case study

Joscha Beninde<sup>1,2\*</sup>, Tatum W. Delaney<sup>3†</sup>, Germar Gonzalez<sup>3†</sup> and  
H. Bradley Shaffer<sup>1,3</sup>

<sup>1</sup>La Kretz Center for California Conservation Science, Institute of the Environment and Sustainability, University of California, Los Angeles, Los Angeles, CA, United States, <sup>2</sup>IUCN WCPA Connectivity Conservation Specialist Group, Gland, Switzerland, <sup>3</sup>Department of Ecology and Evolutionary Biology, University of California, Los Angeles, Los Angeles, CA, United States

**Introduction:** A major goal for conservation planning is the prioritized protection and management of areas that harbor maximal biodiversity. However, such spatial prioritization often suffers from limited data availability, resulting in decisions driven by a handful of iconic or endangered species, with uncertain benefits for co-occurring taxa. We argue that multi-species habitat preferences based on field observations should guide conservation planning to optimize the long-term persistence of as many species as possible.

**Methods:** Using habitat suitability modeling techniques and data from the community-science platform iNaturalist, we provide a strategy to develop spatially explicit models of habitat suitability that enable better informed, place-based conservation prioritization. Our case study in Greater Los Angeles used Maxent and Random Forests to generate suitability models for 1,200 terrestrial species with at least 25 occurrence records, drawn from plants (45.5%), arthropods (27.45%), vertebrates (22.2%), fungi (3.2%), molluscs (1.3%), and other taxonomic groups (< 0.3%). This modeling strategy further compared spatial thinning and taxonomic bias file corrections to account for the biases inherent to the iNaturalist dataset, modeling species jointly and separately in wildland and urban sub-regions and validated model performance using null models and a “test” dataset of species and occurrences that were not used to train models.

**Results:** Mean models of habitat suitability of all species combined were similar across model settings, but the mean Random Forest model received the highest median AUC<sub>ROC</sub> and AUC<sub>PRG</sub> scores in model evaluation. Taxonomic groups showed relatively modest differences in their response to the urbanization gradient, while native and non-native species showed contrasting patterns in the most urban and the most wildland habitats and both peaked in mean habitat suitability near the urban-wildland interface.

**Discussion:** Our modeling framework is based entirely on open-source software and our code is provided for further use. Given the increasing availability of urban biodiversity data via platforms such as iNaturalist, this modeling framework can easily be applied to other regions. Quantifying habitat suitability for a large, representative subset of the locally occurring pool of species in this



way provides a clear, data-driven basis for further ecological research and conservation decision-making, maximizing the impact of current and future conservation efforts.

#### KEYWORDS

urbanization, green infrastructure (GI), environmental niche modeling (ENM), species distribution model (SDM), spatial conservation prioritization, nature based solutions, community science, iNaturalist

## Introduction

Increased urbanization in the Anthropocene has given rise to megacities, fundamentally transforming previously existing landscapes across much of the globe. The environment of cities is strikingly different from adjacent non-urban areas, with elevated levels of human population densities, impervious surfaces, roads, vehicular traffic, artificial light at night, pollution, urban heat, and many other factors shaping the microclimate, hydrology, and soil properties of cities (Groffman et al., 2014). Species living in urban areas must either cope with these altered environmental conditions (Johnson and Munshi-South, 2017) or be relegated to their fringing landscapes. The conservation of high levels of urban biodiversity has become a goal of many urban administrations (Waldrop, 2019) aiming to counteract the loss of human-nature interactions by city-dwellers (Soga and Gaston, 2016) and recoup the multifaceted benefits of biodiversity for human well-being (Fuller et al., 2007; Methorst et al., 2020), and more generally for sustainable urban footprints and biodiversity-ecosystem service synergies (Ziter, 2015; Schlaepfer et al., 2020).

To plan for effective biodiversity conservation, we must first understand the geographical distribution of as much of the regional species pool as possible. The central position of “place” in Morrison’s virtuous cycle framework for biodiversity conservation (Morrison, 2016) emphasizes this point—we need to know the places to focus conservation actions, followed by local community engagement. However, the available data on which conservation decisions hinge is typically restricted to only a small subset of all species in a given landscape (Hochkirch et al., 2021) and is often biased toward charismatic or endangered species. These species may serve as umbrella species, where the benefits of conservation efforts directed at them cascade across other, co-occurring species (Roberge and Angelstam, 2004). However, the umbrella functionality of a species can be difficult to quantify (Fleishman et al., 2001; Roberge and Angelstam, 2004) and may vary for species richness, abundance, or functional diversity (Branton and Richardson, 2011; Sattler et al., 2013). Despite these uncertainties, the traditionally limited availability of information on species distributions usually necessitates using a limited set of species as surrogates to guide decisions on the protection and restoration of habitats for conservation.

Urban areas are no exception to this general trend, and historically knowledge of patterns of distribution and abundance of local biodiversity in our cities has been limited (Kohsaka et al., 2013). However, this is changing. Emerging community-science projects, also referred to as citizen-science (Cooper et al., 2021),

have turned cities into hotspots of biodiversity monitoring, and this trend is only increasing over time (Devictor et al., 2010). Observations of species that are accessible on platforms such as eBird or iNaturalist frequently center on urban habitats and their surrounding landscapes, simply because those areas are the most accessible to large numbers of people (Spear et al., 2017). Unrestricted by the limited capacity of researchers for field observations, iNaturalist alone surpassed 33.7 million observations globally in 2022, up more than 10-fold from 3.3 million observations in 2017<sup>1</sup>, rendering such datasets an unprecedented resource for urban conservation prioritization (Li et al., 2019; Callaghan et al., 2020a). Observations on iNaturalist are taxonomically diverse and unstructured with respect to survey methodology. The nature of how users utilize iNaturalist, which relies on proximity, ease of access, and human esthetics, presumably biases these datasets in predictable ways according to human preferences for certain species/taxonomic groups, species detectability, the reduced chance to take photographs of small, distant or fast-moving species (Di Cecco et al., 2021), site accessibility (Zizka et al., 2021), and varying sampling effort at sites (Beck et al., 2014). Of course, these biases are inherent to most datasets, including herbarium and museum records, which contain similar spatial, environmental, temporal, and taxonomic biases (Newbold, 2010; Martin et al., 2012; Kling et al., 2018).

In contrast to these more traditional data sources, observations on community-science platforms have two important advantages: They accumulate data orders of magnitudes faster than conventional research data (Spear et al., 2017; Callaghan et al., 2020b), and they include observations from private land, which conventional research data frequently cannot sample (Martin et al., 2012). Given the knowledge of potential biases in occurrence datasets, methodologies for adequately addressing them have been proposed for models that use occurrence records and landscape predictors to describe and predict the environmental niche space of species (Warren et al., 2010). These techniques are widely used in biodiversity assessments (Araújo et al., 2019) and we refer to them as habitat suitability modeling (they are also commonly called environmental niche modeling, ENM, or species distribution modeling, SDM). When the input data are managed appropriately, these models allow users to correct for sampling biases via spatial thinning (to reduce over-representation at hotspots of observer activity; Steen et al., 2021), by scaling background locations to the distribution of sampling effort in the

<sup>1</sup> <https://www.inaturalist.org/stats/?year=>

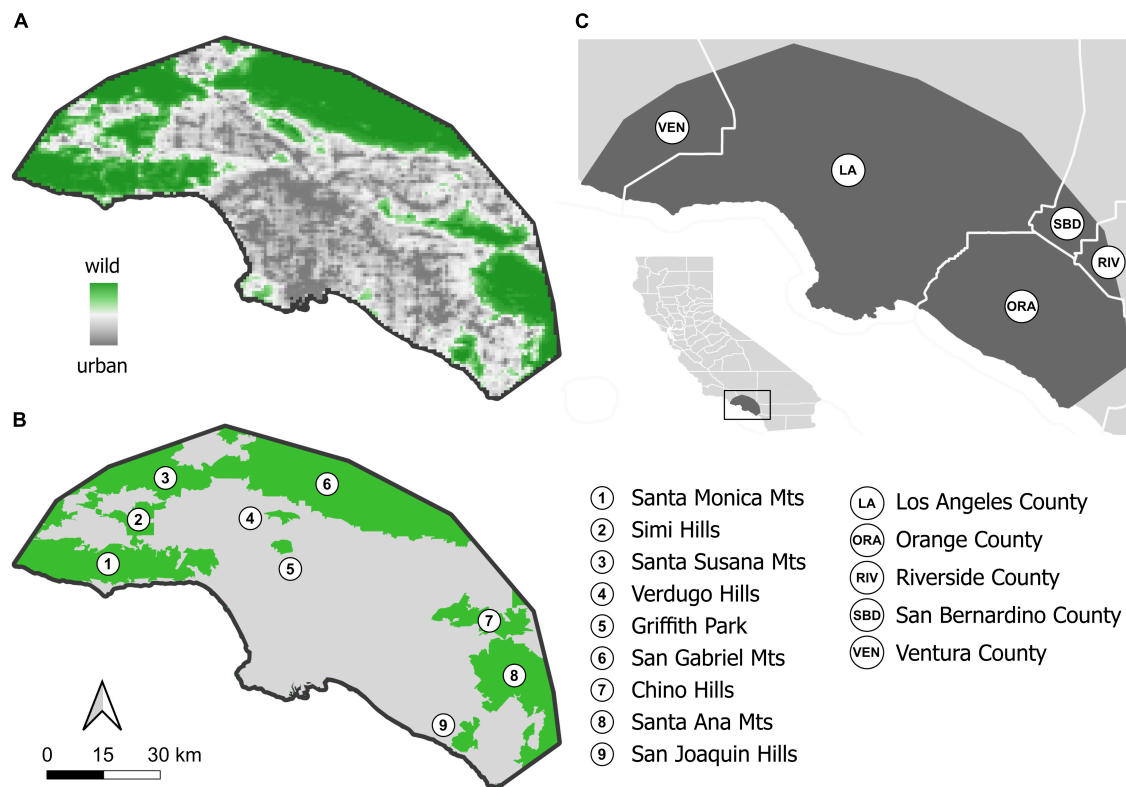


FIGURE 1

Study extent in the Greater Los Angeles area in Southern California, USA. (A) PCA-derived urbanization intensity is depicted as a gradient from wildland (dark green) to urban (dark grey). The urban-wildland interface received values close to zero in this urban PCA space (shown in white), which, spatially, closely resembles the delimitation of urban from wildland areas by the US Census Bureau (B); US Census Bureau "urban" consists of 88.5% of values above zero in urban PCA space, i.e., more urban cells, and 11.5% below zero, i.e., more wildland cells; US Census Bureau non-urban, or "wildland," consists of 4.4% values above zero, i.e., more urban cells, and 95.6% below zero, i.e., more wildland cells. The US Census Bureau delimitation is used in the modeling framework, which was applied to the full extent as well as separately to urban and wildland extents. (C) The study extent with respect to County boundaries and California (inset).

landscape (Kramer-Schadt et al., 2013; Merow et al., 2013; Kling et al., 2018), and/or by creating habitat-specific models (Fourcade et al., 2014). Following these precautionary measures to address spatial biases of occurrence datasets, bias-corrected models of habitat suitability can be parameterized and used to infer habitat suitability for sites that have not been sampled, which is frequently done across thousands of species simultaneously (Bradie and Leung, 2016; Kling et al., 2018). While many approaches to model habitat suitability exist, we use Maxent and Random Forests, which frequently rank among the top-performing modeling methods and require only a fraction of the computational resources of similarly high-performing methods, such as boosted regression trees or ensemble methods (Harrigan et al., 2014; Valavi et al., 2022).

We present a framework utilizing Maxent and Random Forest modeling in combination with the iNaturalist dataset for Greater Los Angeles (Figure 1), the largest metropolitan region in the US by area and the second largest by human population size. More than 1.5 million iNaturalist observations are available for this study extent (as of November 2021), including observations of 6,082 species whose identifications were verified by the iNaturalist community. Using only those terrestrial species with a minimum of 25 occurrence records, we generated habitat suitability models for 1,200 taxonomically diverse species composed of native taxa, ranging from velvet ants

(*Dasymutilla* spp.), swallowtail butterflies (*Papilio* spp.), tarantulas (*Aphonopelma* spp.), rattlesnakes (*Crotalus* spp.), coyotes (*Canis latrans*), mountain lions (*Puma concolor*), oak (*Quercus* spp.), maple (*Acer* spp.), sycamore (*Platanus racemosa*) and pine trees (*Pinus* spp.), poppy (*Eschscholzia* spp.) and Clarkia flowers (*Clarkia* spp.), manzanita shrubs (*Arctostaphylos* spp.), lichenized fungi (Lecanoromycetes), and non-native, human commensal species, including earthworms (*Lumbricus terrestris*), pill bugs (*Armadillidium vulgare*), honey bees (*Apis mellifera*), cockroaches (Blattidae), rats (*Rattus* spp.), house sparrows (*Passer domesticus*) and house cats (*Felis catus*; Supplementary Figure 1). By producing a mean model of habitat suitability of all species (also referred to as stacked models, Calabrese et al., 2014), as well as models for all native and non-native species separately, we created spatially explicit models at 1 km x 1 km raster cell resolution to summarize spatial biodiversity value. Land managers can use these models to guide spatial prioritization and protection, direct conservation and restoration efforts, or simply track current biodiversity, enabling ongoing efforts to create meaningful urban biodiversity indices (Kohsaka et al., 2013; Isaac Brown Ecology Studio and La Sanitation and Environment, 2018).

The question of what species to protect is controversial when discussing non-native species (Sax et al., 2022) and may be especially difficult to answer in cities (Gaertner et al., 2016). The

city of LA's ambitious goals toward a no-net-loss of biodiversity largely focuses on native species (City of Los Angeles, 2019) and some ecosystem services may be enhanced by focusing only on native species. For example, native California chaparral vegetation provides less fuel for fires and has better wind-stopping qualities than non-native vegetation (Keeley, 2020), and in the Greater Los Angeles area some species of native trees and shrubs attract higher densities of birds (Wood and Esaian, 2020; Smallwood and Wood, 2023) and insects (Adams et al., 2020) than non-native tree and shrub species. However, individuals of non-native species in urban areas also have recognized value when viewed through an ecosystem services lens; they provide habitat, disperse seeds, and take over the roles of native pollinators in some situations (Sax et al., 2022). Given the overarching role of urban biodiversity in conveying direct benefits to human health (Sandifer et al., 2015; Methorst et al., 2020), as well as generating an appreciation for biodiversity and avoiding the 'extinction of experience', non-native species in urban areas may sometimes surpass their native counterparts as important agents for developing a public understanding and personal motivation to conserve biodiversity (Schuttler et al., 2018). It has also been argued that urban individuals of endangered non-native species outside of their native range may qualify as a form of *ex situ* conservation; the endangered red-crowned parrot (*Amazona viridigenalis*) in Los Angeles is one example (Shaffer, 2018). We believe these to be important benefits of non-native species in urban areas which deserve consideration in urban conservation prioritization, as they are weighed against the equally important potential threats non-native species can impose on ecosystems and human health (Gaertner et al., 2016). As emphasized by Morrison (2016), identifying the benefits to biodiversity are often specific to place, species (including native versus non-native taxa), local human population and/or need; establishing a positive feedback cycle, or virtuous cycle in Morrison's language, between people and biodiversity conservation, is almost certainly different in urban Los Angeles than in the wilderness of the adjacent San Gabriel mountains. Our goal here is to provide the place-based biodiversity data, across types of species and geographic areas, that is required to establish such positive feedback cycles.

Given this variability in how to think about native vs. non-native biodiversity and the various modeling parameters that address sampling biases explored here, there are three key elements of our study: (1) identify the optimal habitat suitability modeling strategy to address the biases in iNaturalist occurrence data, (2) quantify mean habitat suitability across a gradient of urban intensity and identify hotspots of urban biodiversity, and (3) contrast mean habitat suitability for native and non-native species and different taxonomic groups along a gradient of urban intensity. We predict higher suitability for native species in wildland areas than in fully urbanized sites and the opposite for non-native species, following the conceptual framework of Cadotte et al. (2017) and based on observations across taxonomic groups that high levels of urbanization increase the ratio of non-native to native species (Celesti-Gradow et al., 2006; Ricotta et al., 2010). We further predict that the highest levels of mean habitat suitability in plants will occur where urban areas transition into wildland based on greater habitat heterogeneity, a pattern frequently found for plant species richness across urbanization gradients (Celesti-Gradow et al., 2006; McKinney, 2008). At the same time, the position of this peak likely

varies with the taxonomic group, as levels of species richness along urbanization gradients often vary across many groups of plants and animals (McKinney, 2008; Piano et al., 2020; Theodorou et al., 2020). We end with a discussion of the utility of our modeling framework and highlight its potential application in future urban biodiversity conservation and research.

## Methods

### Study extent

Our study extent is located in Greater Los Angeles, totals 7,797 km<sup>2</sup> and fully encompasses the City of Los Angeles, large parts of the Los Angeles-Long Beach-Anaheim, CA Metropolitan Statistical Area, plus parts of adjacent Ventura County in the west, small parts of Riverside and San Bernardino Counties to the east and into Orange County in the south (Figure 1). Developed land use types predominate (60.3%; Supplementary Table 1) followed by mostly vegetated areas (37.2%), while highly managed, working landscapes, such as agricultural areas, are uncommon (0.7%). Using the US census bureau delineation (U.S. Census Bureau, 2018; Figure 1B), 65.1% of the study extent is urban. We refer to the remaining 34.9% of the area collectively as wildlands, as they entail vast expanses of native vegetation, with little development (Supplementary Table 1), including parts or all of the Santa Monica Mountains, Simi Hills, Santa Susana Mountains, Verdugo Hills, Griffith Park, San Gabriel Mountains, Chino Hills, Santa Ana Mountains, and the San Joaquin Hills (Figure 1B). The study extent thus encompasses extensive urban areas that are home to a dense human population of 13.4 million (2,640 people/km<sup>2</sup>), framed by sparsely populated wildland areas with a modest combined population of 86,700 thousand humans and a density almost two orders of magnitude lower (32 people/km<sup>2</sup>; calculated based on data by Rose et al., 2017). This study extent is thus uniquely suited to study species' distribution patterns within an urban megacity and the immediately adjacent wildlands, separated by a sharp urban-wildland interface (Figure 1A). Given the extreme environmental differences between urban and wildland areas and the potential for contrasting habitat association of species in urban and wildland areas (Fourcade et al., 2014), we modeled (1) the combined urban and wildland habitat (which we refer to as the full study extent) and (2) the urban (5,074 km<sup>2</sup>) and wildland (2,723 km<sup>2</sup>) areas separately to assess habitat suitability (see details below).

### Occurrence datasets

Occurrence records from iNaturalist were downloaded via GBIF and directly from the iNaturalist site to include non-research grade observations, which are not integrated into GBIF, but necessary to control for observer bias (see section "Exploring different settings for occurrence and background point selection" in Supplementary material). iNaturalist is a rapidly-growing platform where the general public can submit observations of species and iNaturalist users can add species identifications to observations. As soon as two identical, independent species-level identifications of an observation are proposed, and the observation

contains a photo voucher, a location, and a date, it receives a “research grade” quality grade. This label, however, is dynamic and persists only as long as 2/3 of the proposed identifications agree; observations can toggle between research and non-research grade as a consequence.

We created two occurrence datasets from iNaturalist observations, one to train models and an independent dataset to test models. The raw training dataset contained all 1,537,123 iNaturalist occurrences falling within our Greater Los Angeles study extent and was downloaded using the search queries on the iNaturalist website. We filtered this dataset according to the following five criteria to include only: (1) research grade entries; (2) non-captive and non-cultivated individuals; (3) spatially unobscured records; (4) observations with a maximum inaccuracy of 100 meters, which equals 10% of the 1km raster cell edge length (or 1% of the area) used in analyses; and (5) species with 25 or more observations. This yielded a training dataset of highest quality observations (both spatially and taxonomically) with sufficient observations to train models accurately. It contained 388,793 occurrence records of 1,286 species with observations made between 1 January 2000 to 31 December 2021. We also excluded all fully marine and aquatic species, including all species of Actinopterygii, Elasmobranchii, and Bivalvia, and some species of Mollusca, Arthropoda, and Plantae (e.g., marine and freshwater slugs and snails, water striders, water scorpions, some Malacostraca, and the plant genus *Pistia*), reducing the number of species in the training dataset to 1,200.

The raw test dataset contained 130,640 iNaturalist observations downloaded from GBIF.org (2022) and was restricted to observations made in 2022 only, ensuring there is no overlap with the training dataset which ended in 2021. This test dataset was identically filtered using the same five criteria as the training dataset with the exception that we relaxed criterion (5) and included species with fewer than 25 observations. The final test dataset consisted of 113,729 occurrence records of 3,458 species, including occurrence data for 2,258 species that were not present in the training dataset. We think that using this test dataset is particularly suitable to evaluate the ability of mean models to predict habitat suitability of species not used for training. It serves to test whether our training dataset reasonably serves as a surrogate for the entire local pool of sampled and unsampled species (see details below).

For all 1,286 species of the training dataset, we determined California native or non-native status using Calflora (2022), iNaturalist species accounts, and the expert opinions of colleagues, primarily at the Natural History Museum of Los Angeles County (NHMLAC) and UCLA (see acknowledgments).

We ran the r-package “CoordinateCleaner” (v2.0-20; Zizka et al., 2019) on the training dataset to check for multiple errors in the coordinates using the function `clean_coordinates()`. This resulted in 26,580 coordinates flagged as being in the sea or too close to the coast and an additional 4,949 coordinates flagged because they were within a 10 km radius of recognized biodiversity institutions, such as museums or universities. We retained these flagged records in both cases. In the case of coast proximity, occurrence records that fall into non-terrestrial space would be dropped in the downstream modeling procedure as we only used terrestrial environmental predictors, and we wanted to retain all truly terrestrial records, including those close to shore,

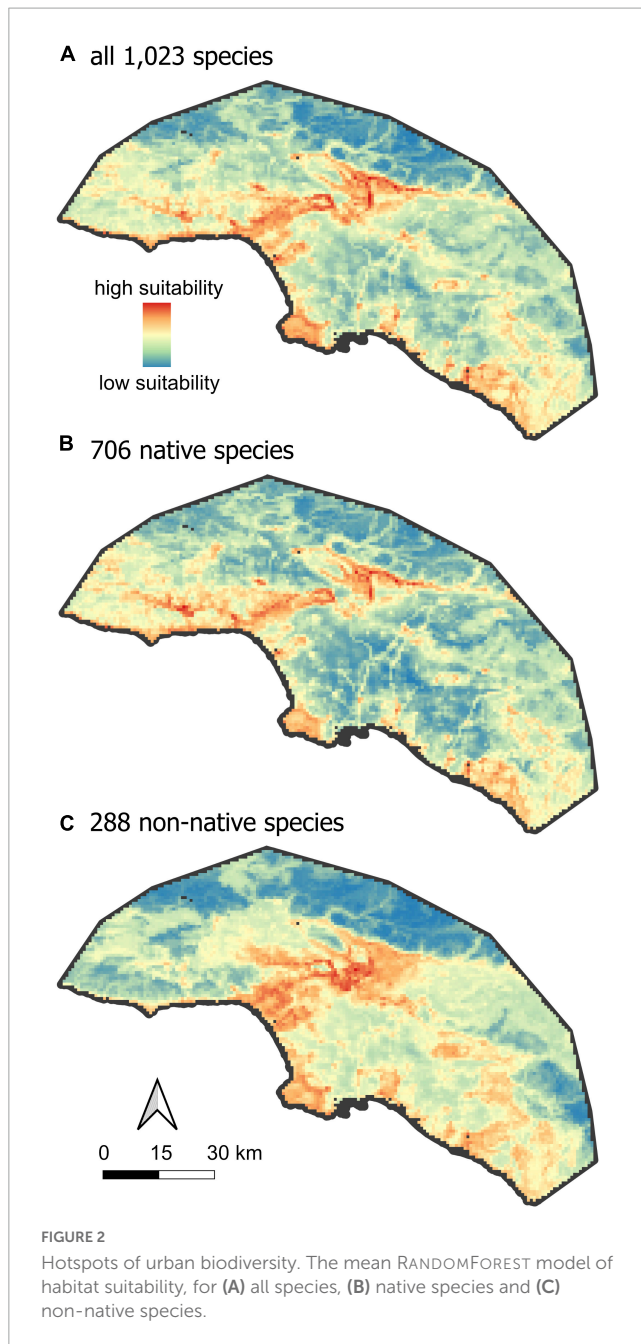
following recommendations by Zizka et al. (2019). For institution proximity, we visually checked flagged records and found them to be proximate to the NHMLAC (2,330 observations), Occidental College (1,045 observations), or UCLA (645 observations), but represented reasonable species observations from a museum or college-associated green space. Historically, some traditional museum samples have erroneously received coordinates of the biodiversity institutions where they are housed rather than the coordinates of the actual sampling sites (Zizka et al., 2019), but this does not seem to be the case in the contemporary iNaturalist dataset for the region.

To assess how well our training dataset covers the environmental predictor space, we calculated a standardized spatial PCA using “RStoolbox” (v0.3.0; Leutner et al., 2022) on the set of landscape predictors chosen for Maxent modeling (see Section “Landscape variables” below) and quantified the distribution of occurrences across the available environmental PCA space. Adequate distribution of occurrences along this environmental PCA space is a key prerequisite for accurate habitat suitability modeling because unsampled, environmentally unique areas can limit the model’s ability to correctly infer habitat suitability in such areas (Elith et al., 2011). We were particularly concerned about high-elevation (above 600 m) sites because these habitats make up only a small fraction of the entire sample extent, are located mostly in remote wildland habitats that may be undersampled by the public, and our initial analyses indicated that they had generally low multi-species predicted suitability. Given that these high-elevation sites often also had relatively few observations/species, we were concerned that undersampling may be contributing to this low suitability. We used a binomial test (`prop.test` function in base r-package “stats” v3.6.2) to compare the proportion of iNaturalist observations that were made in high-elevation habitats for species that were included in the training models (those with 25 or more observations) and species that were excluded from these models because they had fewer than 25 observations. This specifically tested whether species that are more closely associated with high-elevation habitats (those with a higher proportion of observations at higher altitudes) were more likely to drop out of the dataset due to an insufficient number of total observations (< 25), which could bias high-elevation inferences of mean habitat suitability.

## Habitat suitability modeling

Many techniques can build habitat suitability models, also referred to as environmental niche models (ENMs) or species distribution models (SDMs), from occurrence data and a set of landscape predictors (Valavi et al., 2022). These models quantify the distribution of environmental niche space at species’ presence locations and can be used to predict the suitability at unsampled locations, as a function of environmental predictors (Harrigan et al., 2014). The resulting predictions are best interpreted as depicting relative habitat suitability, circumventing issues of interpreting model output as the probability of presence or the relative occurrence rate, which is only valid in the rare cases of entirely random spatial sampling strategies for occurrence data or with complete knowledge of a species abundance in a





given landscape (Merow et al., 2013). To generate these models, environmental predictors at user-specified presence points (the occurrence data) are contrasted with environmental predictors at a variable number of background points, which can be chosen from within the study extent following different strategies. By default, some methods, including the default settings of Maxent, randomly sample background points, such that each 1-km raster cell locality has an equal probability of being chosen. This approach includes a random sampling of cells containing and lacking species observations. Alternatively, it is often recommended to scale the background point distribution to match the sampling intensity dedicated to each location in the study extent, as one way to correct for sampling bias (Kramer-Schadt et al., 2013; Merow et al., 2013). This strategy increases the likelihood of

locations being chosen as background points if they contain many observations and decreases the likelihood for sites with fewer observations.

The number of occurrences necessary to parameterize a model is an important consideration. While models can be parameterized with as few as three occurrence records (Proosdij et al., 2016), model predictions derived from such small sample sizes are often highly variable and should be treated with caution; predictions generally converge with 25–30 occurrence records (Wisz et al., 2008). Based on this convergence, we filtered our training dataset to species with at least 25 observations.

Finally, it is important to recognize that using habitat suitability modeling on a spatial extent as small as in this study does not allow one to quantify a species' full environmental niche space. Rather, the model reflects the (limited) environmental niche space available within the study extent, and can be used to understand habitat suitability within this study extent but not necessarily beyond, in space or time (Araújo et al., 2019).

We used two different methods to model habitat suitability, the maximum entropy-based approach of Maxent (Phillips et al., 2004), and the tree-based approach of Random Forests (Breiman, 2001). Studies comparing different modeling techniques frequently identify these two methods as among the top-performing (Harrigan et al., 2014; Valavi et al., 2022), and they are considerably less computationally intensive than other high-performance models, such as boosted regression trees or Ensemble methods (Valavi et al., 2022). Maxent and Random Forests allow the user to account for potential biases in the occurrence dataset (Kramer-Schadt et al., 2013; Merow et al., 2013; Fourcade et al., 2014), have high predictive power across sample sizes (Wisz et al., 2008), and are comparable in accuracy to other techniques (Kaky et al., 2020; Valavi et al., 2022). We implemented Maxent (v3.4.4; Phillips et al., 2022) through the r-package "dismo" (Hijmans et al., 2017) and Random Forests using the r-package "randomForest" (v4.7-1.1; Liaw and Wiener, 2022) using R Statistical Software (v4.1.2; R Core Team, 2021). Maxent modeling was replicated (Sillero and Barbosa, 2021) and run with 10-fold cross-validation (CV), using AUC values calculated on independent test datasets as a measure of model fit. Random Forest models were fitted with a down-sampling procedure as recommended by Valavi et al. (2022) which uses equal numbers of occurrence and background points, preventing class imbalance issues. Random Forest models were fitted using the default values for mtry and ntree = 1,000.

## Addressing biases in the modeling framework

We first explored different modeling strategies on a set of 15 plants and animal species using Maxent. For these species we made use of our knowledge of their distribution and relative abundance in the study extent, which we acquired during extensive fieldwork, to evaluate the performance of different modeling strategies to predict habitat suitability accurately. To address, and account for, sampling bias in the iNaturalist dataset, we tested several strategies to select occurrence records (Steen et al., 2021) and background points (Kramer-Schadt et al., 2013; Merow et al., 2013; Kling et al., 2018; section "Exploring different settings for occurrence and

background point selection” in the [supplementary materials](#)). To test for divergent habitat preferences and for species distributions that are in potential dis-equilibrium with the environment caused by rapid urbanization or recent introductions, ([Fourcade et al., 2014](#); [Searcy and Shaffer, 2014](#)) we modeled each species for the full study extent, and separately in urban and wildland habitats ([Supplementary Figure 2](#); section “Exploring separate modeling in urban and wildland extents” in [Supplementary material](#)). Based on our expert assessment of model performance for these 15 species, we identified the best model settings and spatial extents. We summarize the results here, as they are central to our methods (see comprehensive details in section “Exploring separate modeling in urban and wildland extents” in [Supplementary material](#)). We noticed considerable differences between models generated at the full, urban, and wildland extents but there was no extent that consistently returned the most realistic results, so we modeled in all three extents. The best models were generated with the following settings for occurrence records and background point selection: (1) THINNED, which uses a single occurrence record per species and raster cell as the occurrence dataset and a uniform prior for background point selection; (2) PHYLUMBIAS, uses all occurrence records of a species as the occurrence dataset and a prior for background point selection that scales to the number of iNaturalist observations of all species of the same Phylum as the target taxon; and (3) CLASSBIAS, also uses all occurrence records of a species as the occurrence dataset and a prior for background point selection that scales to the number of iNaturalist observations of all species of the same Class as the target taxon. These three model settings were adopted for all 1,200 species in our post-pruning dataset and all three spatial extents, resulting in a total of 6,953 models fitted for all combinations of species, model settings, and spatial extent that resulted in at least 25 occurrence records. All 6,953 single-species models were validated using a null modeling approach with randomly placed occurrences similar to [Merckx et al. \(2011](#); section “Null model validation”) to ensure that model performance was significantly improved over that expected by chance ([Raes and ter Steege, 2007](#); [Gomes et al., 2018](#)). Only models of species that exceeded null model expectations were used to calculate mean models.

We created mean models using the continuous predictions of habitat suitability, as recommended by [Calabrese et al. \(2014\)](#), and did so separately for each of the model settings and spatial extents. We rescaled all mean models to a 0–1 scale and evaluated them using the test dataset. Across THINNED, PHYLUMBIAS, and CLASSBIAS model settings, models generated at the full extent received higher  $AUC_{ROC}$  values than those generated separately for urban and wildland extents ([Supplementary Figure 3](#); see section “Creating and validating mean models” in [Supplementary material](#) for comprehensive details). Therefore, we limited all further analyses and modeling to the full extent. We created an additional COMPOSITE model at the full extent, composed of the best-performing model of each species, which we identified as the model with the highest  $AUC_{ROC}$  value among model settings.

In summary, this generated four mean Maxent models that were all modeled at the full extent and that we will refer to as the THINNED, PHYLUMBIAS, CLASSBIAS, and COMPOSITE models.

## Model evaluation

We used the test dataset (see above) to evaluate and rank the performance of each of the mean Maxent models using AUC. We randomly drew 5–100 occurrence records from the test dataset and used a varying number of background locations as absences. We chose the number of background locations to vary between 3 and 50 times that of the number of occurrence records, which means that the theoretically possible number of background locations varied between 15 and 5,000. This procedure was repeated 1,000 times for each of the mean Maxent models to generate a distribution of  $AUC_{ROC}$  and  $AUC_{PRG}$  values, which we calculated using the `evalmod()` function in the “`precrc`” r-package (v0.13.0; [Saito and Rehmsmeier, 2017](#)) and the `calc_auprg()` function in the “`prg`” r-package (v0.5.1; [Kull, 2016](#)). As outlined in [Valavi et al. \(2022\)](#),  $AUC_{ROC}$  and  $AUC_{PRG}$  are complementary and assess model performance either based on both presences and absences or based only on presences, respectively.  $AUC_{ROC}$  is computed using the number of true positives, or sensitivity, and the proportion of false positives, calculated as 1-the number of true negatives.  $AUC_{ROC}$  values vary between 0–1, where 1 indicates a perfect model, i.e., presence locations have higher habitat suitability values than absence locations and there is no overlap in their distributions. Values close to 0 indicate the unlikely, but theoretically possible, opposite case, where absence locations receive higher habitat suitability values than presence locations and there is no overlap in their distributions. A value of 0.5 indicates that a model is no better than randomly assigning habitat suitability values at presence and absence locations. However, there can be considerable variation around the  $AUC_{ROC}$  value of null models, which is why we included the additional null modeling step (see above and section “Null model validation” in [Supplementary material](#)).  $AUC_{PRG}$  is calculated based on the precision of predicted presences, calculated as the proportion of true and false positives, and the sensitivity, calculated as the proportion of true positives versus false negatives. The  $AUC_{PRG}$  metric is scaled so that perfect models also approach the value of 1, as in  $AUC_{ROC}$ , while negative values indicate that models are no better than randomly differentiating between presence and absence locations ([Valavi et al., 2022](#)).

Following these evaluations of mean Maxent models, we ranked them by  $AUC_{ROC}$  and  $AUC_{PRG}$  scores to identify the best-performing model. Using the same settings as for the best mean Maxent model, we generated new habitat suitability models for each species using a Random Forest modeling approach (see above), and created another mean model from these, which we refer to as the mean RANDOMFOREST model. In a final step, for comparison, we evaluated and ranked the mean RANDOMFOREST model in the same way as the mean Maxent models and identified the best-performing overall model.

## Landscape variables

We assembled environmental layers that are relevant to the study region’s biogeography, climate, and landscape. We assembled a total of 37 landscape variables, including all 19 bioclim variables (based on data compiled between the years 1960–1990; [Hijmans et al., 2005](#)), 9 soil variables ([Walkinshaw et al., 2021](#)), climatic

water deficit (Flint et al., 2013), elevation, slope (U.S. Geological Survey, 2017), surface imperviousness, land cover, tree canopy cover (U.S. Geological Survey, 2014), NDVI (ORNL DAAC, 2018), artificial lights at night (ALAN; The Earth Observatory Group, 2018), and water cover (U.S. Geological Survey, 2019). To reduce data collinearity, we reduced the number of landscape variables by checking pairwise correlation coefficients (calculated at the grid-cell level) and removing predictors until all correlation coefficients were below 0.7 (Dormann et al., 2013) using the `raster.cor.matrix()` function from the r-package “ENMTools” (v1.0.6; Warren et al., 2021). We chose the resulting final set of 10 landscape variables to preferentially include ones that, in our view, directly influence species across taxonomic groups, as recommended for creating habitat suitability models (Merow et al., 2013). These included: Bioclim02 (mean diurnal range), bioclim06 (mean temperature of the coldest month), bioclim09 (mean temperature of the driest quarter), bioclim16 (precipitation of the wettest quarter), soil bulk density, soil cation exchange capacity, climatic water deficit, imperviousness, NDVI, and percent water cover (Supplementary Figure 4). All of these data layers were available at a resolution of 1 km or higher and we rasterized all spatial polygons or lines objects, or resampled rasters, to a 1 km<sup>2</sup> resolution using the r-package “raster” (v3.5-15; Hijmans and van Etten, 2022). To evaluate the importance of bioclimatic predictor sets averaged over different periods, we compared model performance using Maxent (based on the THINNED settings only) for the WorldClim dataset averaged for 1960–1990 (Hijmans et al., 2005) and 1970–2000 (Fick and Hijmans, 2017), the CHELSA dataset for 1980–2010 (Karger et al., 2017, 2018), and the ClimateNA dataset for 1990–2020 (Wang et al., 2016; AdaptWest Project, 2021).

Using the spatial PCA function from the R-package “RStoolbox”, we generated a standardized spatial PCA based on only imperviousness and ALAN, the “urban PCA”, to generate a gradient of urban intensity, which we used for analyses of species responses to the intensity of urbanization.

## Further analyses

To evaluate and summarize the quality of the modeling framework presented here, we scored it following the 15 criteria established by Araújo et al. (2019), which provide minimum requirements of habitat suitability modeling for application in biodiversity assessments.

In addition to validation of mean habitat suitability models (see above), we compared Schoener’s D, Warren’s I, and rank correlation coefficients between mean models of habitat suitability using the `raster.overlap()` function in ENMtools (Warren et al., 2010). We provide all three metrics but place more emphasis on rank correlation coefficients for the interpretation of results because the other two measures tend to overestimate raster similarity (Warren et al., 2021). We categorized correlation coefficients > 0.9 as very similar, those between 0.7 and 0.9 as similar, and all coefficients below 0.7 as different (Dormann et al., 2013).

To quantify the influence of landscape variables on the highest-ranking mean model of habitat suitability, we used Random Forest models (Breiman, 2001) and the r-package “randomForest”, with `ntree` = 10,000 and `mtry` = the number of predictor variables/3

(Liaw and Wiener, 2022). We used the best mean model of habitat suitability for (1) all species, and subsets of (2) only native and (3) only non-native species as response variables, and all 37 landscape variables as predictors.

By quantifying the influence of the number of iNaturalist observations on the highest-ranking mean model of habitat suitability, we tested for signatures of sampling bias persisting in the mean models of habitat suitability. As above, we used Random Forest models for this and the best mean model of habitat suitability for (1) all species, (2) only native and (3) only non-native species as response variables and four different summaries of the number of iNaturalist observations as predictors (the sum of iNaturalist observations of all species, and separately, the sum of iNaturalist observations of plants, vertebrates, and arthropods).

In another effort to evaluate whether sampling bias is driving the results of the mean habitat suitability models, we used density plots across the urban PCA space of all iNaturalist observations and of the highest predicted mean habitat suitability. For the latter, we included only those raster cells of the highest-ranking mean habitat suitability model that fell within the highest quartile of predicted values.

We used loess regressions to visualize changes in mean habitat suitability values across the urban PCA space, separately for plants, vertebrates, and arthropods, as well as for native and non-native species. We used GAM (generalized additive models) models to quantify the strength of these associations using adjusted R<sup>2</sup> values (r-package “mgcv” v1.8-40; Wood, 2017).

## Results

### Native and non-native status of species of the training dataset

For the training dataset, we generated habitat suitability models for 1,200 species and were able to determine the native/non-native status of 1,183 (98.6%) of those species. Of that set of taxa, 835 species (70.6%) are native and 348 (29.4%) are non-native. The majority of the 17 species for which we could not determine the native/non-native status were Fungi (6) and Insecta (6), followed by Plantae (3), Myxomycetes (1), and Platyhelminthes (1). The proportions of native species were higher for vertebrates (88%) and arthropods (77%) than for plants (59.3%; Supplementary Table 2).

### Occurrence records across elevation

Testing the possibility that species restricted to remote, high-elevation sites may be underrepresented in our iNaturalist-based training dataset, we found a significantly higher proportion of observations from high-elevation areas (> 600 m) in species with a total number of observations below 25 than in species with a total number of records of 25 or more, for which habitat suitability models were created ( $\chi^2 = 131.24$ ,  $df = 1$ ,  $p$ -value < 0.001). This suggests that species more strongly associated with high elevation, may be underrepresented in our analyses, either because of true rarity, sampling bias, or the delineation of the study extent.



## Comparison of US Census Bureau defined urban and wildland regions to urban PCA

The US Census Bureau delimitation of urban areas was utilized to separate urban from non-urban (wildland) areas for separate Maxent modeling in these two study extents. However, urbanization also needs to be considered as a quantitative, rather than a qualitative, landscape attribute. To quantify the level of urbanization on a continuous scale, we generated a raster PCA of the levels of impervious surface and artificial lights at night, which we designate as the urban PCA. The first axis of this urban PCA explained 97.4% of the total variation in these two variables and is therefore a reasonable proxy for urban intensity as defined by artificial hardscapes and light. PC1 ranged from -0.7 – 4.7, and values at or near zero spatially resemble the borders of the US Census Bureau delimited urban areas (which is a two-state, rather than continuous, delimitation) at the urban-wildland interface. US Census Bureau urban areas consist of 88.5% of values above zero and 11.5% below zero on PC1, while wildland regions (that is, non-urban areas defined by the Census Bureau) consist of 4.4% of values above zero and 95.6% of values below zero on PC1 (Figures 1A, B). This indicates that there is a strong correlation between these discrete and continuous measures of urbanization. Consistent with this result, the proportion of NLCD (National Land Cover Dataset) land cover types was also significantly different between these two classes, with developed land-use types dominating in urban areas (> 87%) and shrub/scrub dominating in wildland areas (> 70%; Supplementary Table 1).

## Model performance using different sets of climatic variables

Comparisons of the WorldClim dataset for 1960–1990 and 1970–2000, the CHELSA dataset for 1980–2010, and the ClimateNA dataset for 1990–2020 returned very similar performances for Maxent (all comparisons were conducted using the THINNED settings; see section “Modeling using climatic variables across timespans” in supplements). Models using the 1960–1990 WorldClim predictors received marginally, but significantly, higher  $AUC_{ROC}$  values (Supplementary Figure 5) and we, therefore, conducted all additional modeling using this dataset.

## Mean models of habitat suitability

The four mean Maxent models were composed of different numbers of species, given the additional spatial requirements for occurrence records of THINNED models and due to null model validation (section “Null model validation”; Supplementary Figure 6): THINNED = 1,023; PHYLUMBIAS = 1,196; CLASSBIAS = 1,197; COMPOSITE = 1,199 species (derived from 399 THINNED models, 335 PHYLUMBIAS and 465 CLASSBIAS models; all based on the full extent modeling). The COMPOSITE model contained single-species models ranging in AUC score from

0.629–0.998 (median = 0.843; Supplementary Figure 7), built with 25–13,346 occurrence records per species (median = 99); Supplementary Figure 8). The mean RANDOMFOREST model of habitat suitability was created with the same settings, and for the same set of species, as the THINNED Maxent model (see next paragraph).

## Model evaluation

Using the test dataset, the THINNED Maxent model ranked higher for both  $AUC_{ROC}$  and  $AUC_{PRG}$  values than the three other Maxent models (CLASSBIAS, PHYLUMBIAS, and COMPOSITE models; Figure 3). Random Forest modeling was therefore conducted for the same 1,023 species modeled with Maxent using the THINNED settings, i.e., based on a single observation per species per raster cell as the occurrence dataset and a uniform prior for background point selection. Following the same model evaluation procedure as for the Maxent models, the mean RANDOMFOREST model yielded the overall highest-ranking model, outperforming all Maxent models on both  $AUC_{ROC}$  and  $AUC_{PRG}$  scales (Figure 3). Based on this, all further analyses, quantifying hotspots of urban biodiversity, and the responses of different taxonomic groups and native and non-native species to levels of urban intensity are based on mean RANDOMFOREST models.

Raster comparisons of the RANDOMFOREST model and the four Maxent models (THINNED, CLASSBIAS, PHYLUMBIAS, and COMPOSITE) were very similar, and pairwise comparisons using Schoener's D ranged from 0.899–0.983, Warren's I from 0.991–1 and correlation coefficients from 0.884–0.997 (Supplementary Table 3). Standard deviations between these five models, calculated at the level of each raster cell, ranged from 0.000698–0.165, with a median of 0.073 (Supplementary Figure 9).

## Sampling bias

The first three axes of the environmental PCA explained 72.5% of the variation (PC1 = 36.7%; PC2 = 22.1%; PC3 = 13.7%) and

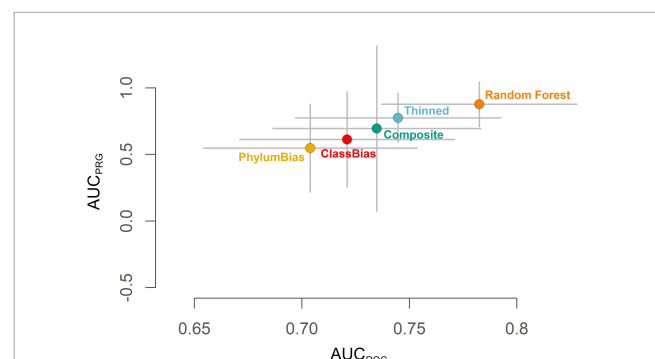


FIGURE 3

Evaluation of the mean RANDOMFOREST model of habitat suitability and the four mean Maxent models (THINNED, CLASSBIAS, PHYLUMBIAS, and COMPOSITE). Grey bars indicate standard deviations around the mean estimates of  $AUC_{ROC}$  and  $AUC_{PRG}$  following model evaluation using the test dataset.



iNaturalist observations covered this environmental space well (Supplementary Figure 10). Random Forest models using the summed iNaturalist observations per raster cell (separately for all species of arthropods, vertebrates, plants, and as a sum of all species) as predictor variables and the mean RANDOMFOREST habitat suitability model as the response variable explained 8.15% of the variation, indicating that mean habitat suitability was not strongly associated with the spatial distribution of iNaturalist observations. Density plots of the highest quartile mean RANDOMFOREST habitat suitability values across the urban PCA space show a similar pattern, with some similarities but also marked differences to the density of all iNaturalist observations across the urban PCA (Figure 4).

## Effects of the native status of species and different taxonomic groups

Using the mean RANDOMFOREST models of habitat suitability as the response variables and all landscape variables as predictor variables separately for all species, only native and only non-native species, Random Forest models explained 97.5, 97.4, and 98.6% of the variation, respectively. The landscape variables that explained most of the variation depended on the response variable being modeled (Table 1); water cover, soil bulk density, NDVI, and imperviousness explained most of the variation for the model containing all species as a response (Supplementary Figure 11).

Rank correlation coefficients showed the strongest differences in mean RANDOMFOREST habitat suitability models of only native and only non-native species (correlation coefficient = 0.505; Schoener's D = 0.81; Warren's I = 0.974). This difference persisted in the response of native and non-native species to urban PCA space; native species have higher mean suitability in wildland areas than non-natives, while in areas that are fully urbanized, non-native species have higher mean suitability than natives (Figure 5 and Supplementary Figure 12).

GAM models testing the association of habitat suitability of native and non-native species to the continuous urbanization PCA space were highly significant ( $p < 0.001$ ), with more variation explained for non-native species (adjusted  $R^2 = 0.28$ ) than for native species (adjusted  $R^2 = 0.07$ ). Within native species

(Figures 6A–C), GAM models testing the association of habitat suitability and urban PCA space explained most variation in plants (adjusted  $R^2 = 43.6\%$ ), while the models for vertebrates and arthropods had low predictive power (adjusted  $R^2 < 1\%$ ) and all models being highly significant ( $p < 0.001$ ). For non-native species (Figures 6D–F), GAM models testing the association of habitat suitability and urban PCA space explained less variation in plants (adjusted  $R^2 = 16.9\%$ ) than in vertebrates and arthropods (adjusted  $R^2 = 45\%$  and  $52.2\%$ , respectively), again, all models being highly significant ( $p < 0.001$ ).

## Discussion

Large, human-dominated ecosystems like Greater Los Angeles differ from more ecologically pristine landscapes in at least two important ways. First, they often contain steep environmental gradients between urban and wildland habitats. In many cases, a single roadway or chain-link fence may separate largely intact natural habitats from areas characterized by high levels of imperviousness, extremely high human population densities, and a correspondingly high concentration of human commensal species. This habitat heterogeneity, over scales of a few hundred meters, presents both challenges and opportunities for different groups of organisms. Second, urban ecosystems tend to have high concentrations of non-native species. Whether these taxa should be considered unwanted pests, integral parts of novel ecosystems (Sax et al., 2022) or valuable elements of *ex situ* endangered species recovery (Shaffer, 2018) depends on the urban context, the goals of urban planners, the preferences of a diverse array of residents, and the conservation status of the species in question (Gaertner et al., 2016). Rather than enter into a debate on the roles of non-native species in urban ecosystems, our goal is to provide the biodiversity distributional data upon which decisions depend, based on best-practice standards for biodiversity assessments (Araújo et al., 2019; Supplementary Table 4). We provide models identifying hotspots of urban biodiversity jointly for a broad set of 1,023 commonly observed species, as well as separately for native and non-native species (Figure 2). We hope that the modeling framework outlined here for Los Angeles will provide a baseline for future research, and that the resulting data will allow planners to include more comprehensive appraisals of the distribution of biodiversity in their assessments and management plans for other urban centers.

## Comparison of methods to model mean habitat suitability

Validation of models using the 2022 iNaturalist test dataset, which included records of more than 2,200 species that were not used to train our models, demonstrated that, across modeling settings, models generated at the full extent outperformed those that were combined from models generated separately in urban and wildland areas (Supplementary Figures 2, 3). We were surprised by this result, but it may indicate that, while significant differences between urban and wildland models persist at the species level (section “Exploring separate modeling in urban and wildland extents” in Supplementary material), modeling at the full extent

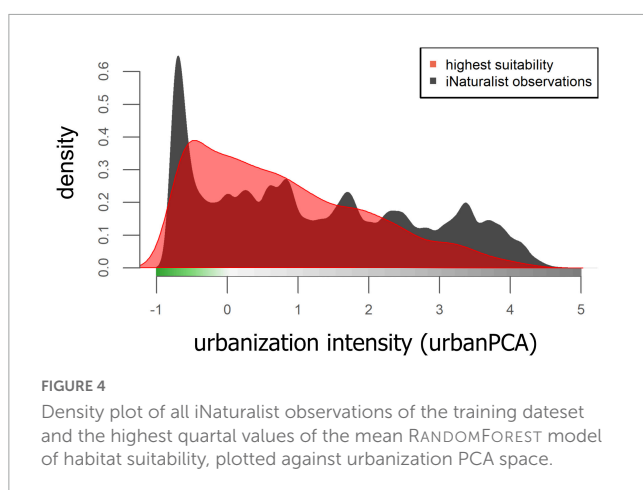
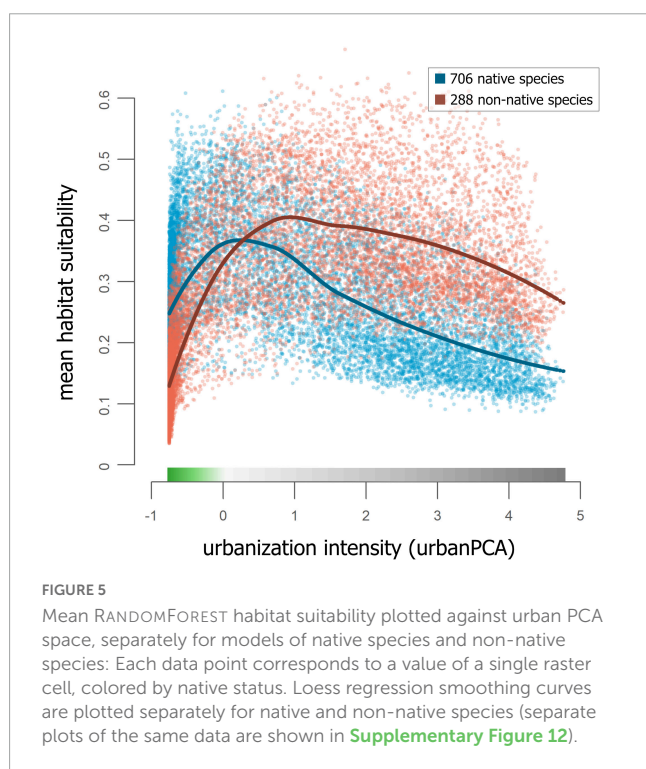


TABLE 1 Importance of landscape variables for mean habitat suitability (%IncMSE: percent increase in mean squared error).

All species model		Only native species model		Only non-native species model	
%IncMSE	Landscape predictors	%IncMSE	Landscape predictors	%IncMSE	Landscape predictors
31.3	Water cover	38.5	water cover	30.4	bulk density
27.8	Bulk density	27.3	bulk density	28.9	NDVI
24.8	NDVI	25.0	NDVI	20.2	cation exchange capacity
23.5	Imperviousness	24.2	imperviousness	19.8	climatic water deficit
19.3	Cation exchange capacity	19.7	cation exchange capacity	18.7	organic matter

Models used the mean RANDOMFOREST model of habitat suitability as the response variable and landscape variables as predictors and were run separately for all, only native and only non-native species (explaining > 97% of the variation in all models). Soil cation exchange capacity, soil bulk density, and NDVI are among the five most important landscape variables of all three models, with water cover and imperviousness among the five most important landscape variables of models for all species and only native species.



is the best (and fortunately, also the simplest) approach when generalizing across species.

Among those models generated at the full extent, the mean RANDOMFOREST model of habitat suitability, generated using the THINNED settings, ranked above all four mean Maxent models for both  $AUC_{ROC}$  and  $AUC_{PRG}$  metrics (Figure 3). Pairwise raster comparisons of all five mean models showed a fairly high similarity between models, with all values of Schoener's  $D > 0.89$ , Warren's  $I > 0.99$ , and correlation coefficients all  $> 0.88$  (Supplementary Table 3).

By using the test dataset to generate  $AUC_{ROC}$  and  $AUC_{PRG}$  metrics, we can ask whether our mean RANDOMFOREST model is an accurate representation of unsampled biodiversity for the Greater Los Angeles ecosystem. The high values of  $AUC_{ROC}$  (median = 0.783) and  $AUC_{PRG}$  (median = 0.877) demonstrate that the mean RANDOMFOREST model of habitat suitability is suited to accurately predict hotspots of urban biodiversity more broadly, especially for species that occur but were not part of the modeling framework. At the same time, we recognize the need to explore

model validation further, particularly at the species level, and ideally using structured survey datasets that contain true absences (Valavi et al., 2022).

Habitat suitability of single species can be closely associated with the observed abundance of that species (Weber et al., 2017; de La Fuente et al., 2021), but it need not be. There is conflicting evidence on this association (Boyce et al., 2016; Dallas and Hastings, 2018), and it has been suggested that predicted habitat suitability values more closely reflect the upper limit of a species' abundance (VanDerWal et al., 2009). Habitat suitability for multiple species, including the mean habitat suitability model constructed here, can be interpreted as the cell-by-cell probability of encountering many species, analogous to interpreting a single species' habitat suitability model as the probability of that species' presence (Elith et al., 2011). Put another way, mean models of habitat suitability are commonly interpreted as reflecting species richness, or alpha diversity, across a modeling extent (Calabrese et al., 2014).

## Can iNaturalist records be harnessed as valid indicators of species distributions?

A common, and reasonable critique of community science observational data is that it reflects where people go to observe nature, rather than the true distribution of biodiversity. To some extent, this must be true, just as it is for museum specimen records or sample sites in ecological studies—people tend to go where access is relatively easy (Newbold, 2010; Martin et al., 2012). However, four lines of evidence convince us that the post-filtered iNaturalist dataset paired with the analyses run here are a reasonable representation of true mean habitat suitability rather than a reflection of rates of human visitation. First, the vast majority of the raster cells encompassing the total environmental PCA space in our study extent contain iNaturalist observations (Supplementary Figure 10). While some cells have many observations and some relatively few, the iNaturalist dataset used in this study did not leave unique environmental conditions unsampled. Second, models using iNaturalist observations as predictors and the mean RANDOMFOREST model of habitat suitability of all 1,023 native and non-native species as the response variable explained a modest 8.15% of the variation. If visitation frequency and their associated iNaturalist observations were driving the mean model of habitat suitability, we would expect this to be much higher. Third, density plots of iNaturalist

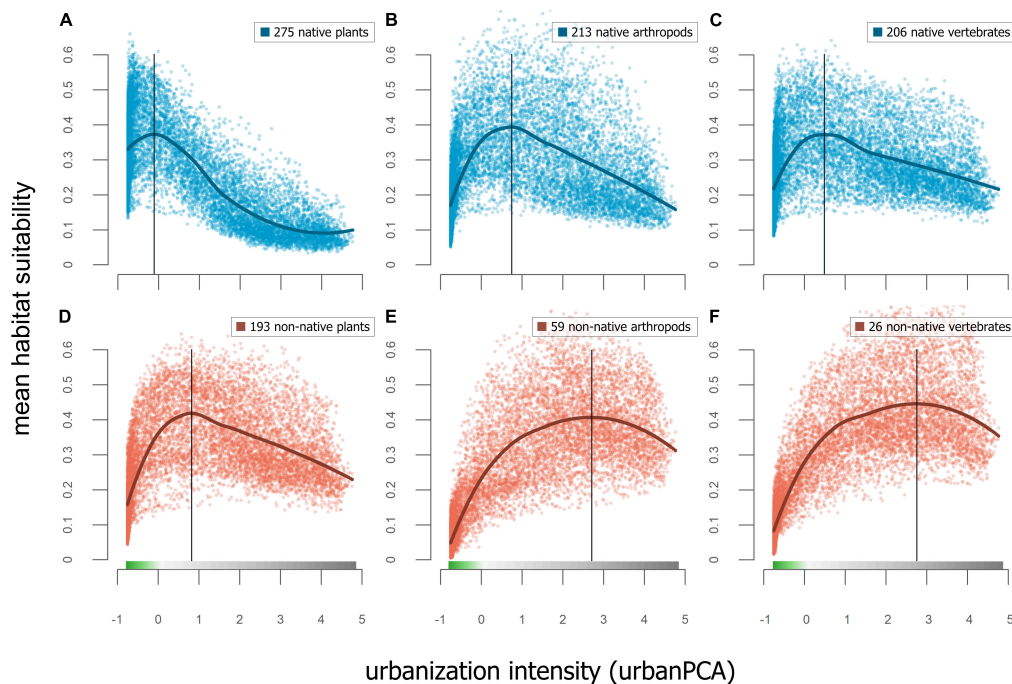


FIGURE 6

Top row shows mean RANDOMFOREST models of habitat suitability for (A) native plant species, (B) native arthropod species, and (C) native vertebrate species. The association to urban intensity were only strong in plants (adjusted  $R^2 = 43.6\%$ ) and low in vertebrates and arthropods (adjusted  $R^2 < 1\%$ ). Bottom row shows mean RANDOMFOREST models of habitat suitability for (D) non-native plant species, (E) non-native arthropod species, and (F) non-native vertebrate species. The association to urban intensity was less strong in plants (adjusted  $R^2 = 16.9\%$ ) and stronger for vertebrates and arthropods (adjusted  $R^2 = 45\%$  and  $52.2\%$ , respectively). Vertical lines show the level of urban intensity with highest mean habitat suitability.

observations and the highest quartal mean habitat suitability values showed considerable mismatch (Figure 4), indicating that habitat suitability models are not driven by the sheer number of iNaturalist observations at a given locality. And last, the independent test dataset used to evaluate all mean models of habitat suitability produced median  $AUC_{ROC}$  values  $> 0.7$ , similar to model validation in other studies (Valavi et al., 2022), indicating that it predicts the presence of many species accurately. Collectively, we interpret this as strong evidence that spatial variation in human sampling intensity was adequately addressed by the methodologies employed here, and that the resulting models can be interpreted as depictions of true multi-species habitat suitability, largely uninfluenced by the location and number of iNaturalist records alone. The one exception to this may be the low habitat suitability modeled for the highest elevation sites in our study extent, although this is likely due to the delineation of the study extent rather than the modeling itself, as discussed below.

## Hotspots of urban biodiversity in the Greater Los Angeles ecoregion

In Greater Los Angeles, areas of the highest mean habitat suitability are distributed in a pattern that is closely aligned to, but not identical with, the spatial distribution of wildland habitat (compare Figures 1, 2). The difference is a subtle offset, such that regions of high mean habitat suitability (orange-red in Figure 2) very closely align to the urban-wildland interface (the light gray

regions of Figure 1A), while both very urbanized and very wild areas receive lower values of mean habitat suitability (Figure 5). This general result is similar to findings for species richness of plants (McKinney, 2008) and birds (Vale and Vale, 1976), and likely reflects the greater habitat heterogeneity at the interface of this steep environmental gradient. Beninde et al. (2015) found a similar increase in species richness as a function of habitat richness across taxonomic groups in globally distributed cities. The lowest mean habitat suitability was detected in heavily urbanized areas, but also in some of the wildest areas within the study extent, including the San Gabriel and Santa Susana Mountains (Figure 1B). The San Gabriel mountains reach the highest elevation (3,069 m) within the study extent and harbor unique environmental conditions, but only make up a limited area within the total study extent. Further analyses indicated that species restricted to high elevation (above 600 m) may have been under-represented in the training dataset, with too few occurrence records of these high-elevation species to pass our 25-observation filter, and this may reduce the apparent suitability of these habitats. However, independent studies from other parts of the world show a similar decrease in species richness with increasing elevation across taxonomic groups (Lee et al., 2004; Nogués-Bravo et al., 2008). Thus, our inferred low habitat suitability in these ecologically intact high-elevation areas could reflect insufficient sampling, true low habitat suitability, or both; increased sampling efforts are necessary to resolve this question.

The landscape variables that stand out in their importance to explain the mean RANDOMFOREST model of habitat suitability were, in decreasing order of importance, water cover, soil bulk



density, NDVI, imperviousness, and soil cation exchange capacity (Table 1). The positive effect of water cover and all types of vegetation, as captured by NDVI, on species richness is well known across taxonomic groups in globally distributed cities (Beninde et al., 2015), and may be even more pronounced in the relatively xeric conditions that characterize most of our southern California study extent. Cation exchange capacity, a measure of soil nutrient availability, has a negative impact on mean habitat suitability, similar to findings from non-urban systems that showed reduced plant species richness in soils with higher cation exchange capacity (Huston, 1980; Le Brocq and Buckney, 2003; Palmer et al., 2003). The positive effect of bulk density on habitat suitability deserves further inquiry. Bulk density is an indicator of soil compaction and is rarely used in analyses of species richness. Rather, it is often considered indicative of ecosystem functionality, since high levels of compaction decrease the water storage capacity of soils (Wang et al., 2018). Imperviousness has a negative association with mean habitat suitability, which is highest at low levels of imperviousness and decreases rapidly between 30 and 70% of impervious surface cover, beyond which levels of mean habitat suitability remain consistently low. This pattern is consistent with many other observations across plant and animal species that impervious surface cover reduces species richness, diversity, and abundance (Sattler et al., 2010; Geslin et al., 2016; Gillespie et al., 2017; Choate et al., 2018; Souza et al., 2019; Yan et al., 2019; Piano et al., 2020).

## Responses to urbanization by native and non-native species

Many studies synthesizing data across taxonomic groups and scales have found very different responses to urbanization between generalist and specialist species (Callaghan et al., 2019), among taxonomic groups (McKinney, 2008), and between native and non-native taxa (Celesti-Grapow et al., 2006). In line with our most general predictions, native species had high mean habitat suitability in wildland areas, lowest habitat suitability in urban areas, and the highest suitability in the proximity of the transition from wildland to urban habitats (Figure 5). In contrast, and consistent with expectations (Cadotte et al., 2017), wildland areas received lower values of mean habitat suitability than urban areas for non-native taxa, and, like native species, non-native taxa peaked in mean habitat suitability at the transition from wildland to urban. Drivers of the difference in mean habitat suitability between urban and wildland areas for non-native species need to be explored further, as knowledge on this is limited (Cadotte et al., 2017; Spear et al., 2017). Such a difference in mean habitat suitability for non-native species may be higher in cities that have strongly seasonal and relatively arid Mediterranean climates, including Greater Los Angeles, than in less arid cities. Although this has not to our knowledge been explicitly examined, we suspect that supplemental watering may be a stronger environmental influence at urban-wildland interfaces in arid or extremely seasonal climates, leading to more severe environmental gradients and reduced spillover of non-native urban species into adjacent wildland areas. More generally, following the categorization of species based on their urbanization tolerance (Fischer et al., 2015), our results confirm findings from a global

analysis of urban bird and plant species (Aronson et al., 2014) and demonstrate that native species tend to be less urban tolerant than non-native species in Greater Los Angeles.

## Similar responses to urbanization by different taxonomic groups

Our findings indicate that most taxonomic groups have hump-shaped responses in mean habitat suitability with respect to urbanization, peaking in the proximity of the urban-wildland interface. Comparisons between taxonomic groups, conducted separately for native and non-native species, showed similar responses (Figure 6). Native plant, arthropod, and vertebrate species show peaks at similar levels of urban intensity, with maxima at the transition from urban to wildland. In non-native species, these peaks shift toward higher levels of urban intensity, although this shift is strongest in arthropod and vertebrate species. These findings are consistent with other studies that found the highest levels of plant species richness at intermediate levels of urbanization across cities (McKinney, 2008). The higher habitat suitability values of native plants in relatively more wildland areas, in comparison to that of native arthropods and vertebrates, may be explained by the unique positioning of our study within the California Floristic Province biodiversity hotspot (Myers et al., 2000).

The decline in habitat suitability with increasing intensity of urbanization is gradual rather than showing an obvious threshold or step-cline pattern. However, the strengths of these associations for taxonomic groups were variable. The strongest associations were found for native plants (adjusted  $R^2 = 43.6\%$ ) and for non-native vertebrates and arthropods (adjusted  $R^2 = 45\%$  and  $52.2\%$ , respectively). These results warrant further research into the responses of other taxonomic groups, for plants and animals, to levels of urban intensity. Highly variable responses in species richness of various insect taxa have been demonstrated by urban-rural comparisons in temperate European cities, with some taxa peaking in urban areas, others in rural areas, and some showing no significant differences between the two (Theodorou et al., 2020). A comparison of levels of avian species richness within multiple Mexican cities demonstrated that bird species richness was higher in green spaces than in areas with more impervious surfaces, although this varied with the functional group of species (MacGregor-Fors et al., 2021). Following this example, future studies could include the response of different functional groups, potentially including additional species traits such as aspects of life history or physiology, to explore the mechanisms underlying species' responses to urbanization across taxonomic groups.

## Conservation efforts in Greater Los Angeles

The City of Los Angeles has the ambitious, and admirable, goal of no net loss of biodiversity by 2050 (City of Los Angeles, 2019). Given that many species in the region are negatively affected and threatened by urbanization (Vandergast et al., 2009; Thomassen et al., 2018; Gustafson et al., 2019), many existing and pending plans focus on protecting and enhancing existing



wildland areas. Efforts to mitigate the risks of future urbanization include the Annenberg Wildlife Crossing in Liberty Canyon, the Wildlife Pilot Study (City of Los Angeles, 2014), and the Rim of the Valley Corridor (Zellmer and Goto, 2022), which together aim to protect large habitat patches and existing and constructed connections between them to allow wildlife to achieve long-term persistence. Spatially, the extent of the Wildlife Pilot Study covers a portion of the eastern Santa Monica Mountains, while the Rim of the Valley extends around the San Fernando Valley to include portions of the Santa Monica Mountains, Simi Hills, Santa Susana, and San Gabriel Mountains, and Verdugo Hills (Figure 1B), including the Annenberg Wildlife Crossing. Many of these areas have received among the highest mean habitat suitability scores from our models and are thus rightful candidates for protection. However, our models also emphasize other regions with high suitability values, including the urban-wildland interface regions along the southern flanks of the San Gabriel Mountains, and pockets of urban open space dotted across the region. Many of these regions are relatively modest in size compared to large wildlands; key regions include the Sepulveda Basin, Baldwin Hills, Ballona Wetlands, Dominguez Gap Wetlands, Coyote Hills, Whittier Narrows, and Upper Newport Bay. These regions emphasize the well-established importance of urban green and open spaces for urban biodiversity, including sometimes-isolated or small patches (Beninde et al., 2015; Wintle et al., 2019).

Many of these highly suitable areas also encompass the most affluent areas in the region, including Bel Air, Beverly Glen, and the Hollywood Hills, while extensive areas of low habitat suitability often fall in low-income neighborhoods including Downtown and South Los Angeles. Both formal policy and broadly accepted equity concerns demand that the positive effects of nature and biodiversity should be accessible to, and impactful for, all, regardless of wealth (Schell et al., 2020). Our models highlight that providing equitable access to areas with high mean habitat suitability presents a real challenge and needs to become an integral goal for future biodiversity planning in Greater Los Angeles. While the habitat suitability models presented here can identify areas that have particularly low levels of biodiversity, corresponding efforts to restore sites and provide green-space access also need to take into account threats of green gentrification, further complicating such efforts (Maantay and Maroko, 2018). To put this in the context of Morrison (2016), achieving virtuous cycles that enhance biodiversity conservation will require different inputs, given the very different human constituencies interacting with that biodiversity, in different parts of Greater Los Angeles. The places, people, and benefits from and for nature are different, and require different approaches, even if the consistent goal is increased biodiversity.

## Conclusion: Outlook on the application of iNaturalist for urban ecology and conservation

While data deficiency plagues biodiversity research globally (Hochkirch et al., 2021), and knowledge gaps in urban areas

persist that range from the identification of habitat patch size thresholds, to evolutionary trap and population sink dynamics and the best landscape configuration to facilitate dispersal (Aronson et al., 2017), the accelerating number of species observations from urban community scientists is unprecedented and encouraging (Callaghan et al., 2020a). With growing confidence in adequately addressing the biases inherent in community science datasets using habitat suitability modeling techniques, iNaturalist and similar datasets have become invaluable resources allowing in-depth comparisons of thousands of species across cities globally. In the future, such data should allow tracking of changes in distributional patterns of taxa along urbanization gradients. These data have already been used to document shifts over the last decades for Los Angeles birds (Cooper et al., 2020), and the recent displacement of the region's native urban black widow spiders by the introduced congeneric brown widow spider (Kempf et al., 2021). The modeling framework outlined here can and should be expanded upon to include biotic interactions (Dormann et al., 2018). This can include methodological approaches, including linking them to macroecological models and comparisons of inferences to multi-species occupancy models (Calabrese et al., 2014; Devarajan et al., 2020), and empirically by integrating presence-absence modeling techniques and data (Isaac et al., 2020). A key goal should be to corroborate habitat suitability modeling from iNaturalist datasets with other, independent data sources, such as scientific monitoring surveys (Prudic et al., 2018) and field validation studies (Searcy and Shaffer, 2014). Furthermore, the increasing availability of high-resolution observations and data layers could allow for modeling fine-scale impacts of smaller patches of urban green spaces (Beninde et al., 2015) or scale-dependent effects across cities (Alberti and Wang, 2022). At its core, our study creates a resource for use by urban planners in Greater Los Angeles and provides a framework that other cities can implement to generate a more comprehensive understanding of the spatial distribution of biodiversity value in their region. Using this framework can provide policymakers with a spatially explicit tool for implementing planning strategies that are most appropriate for biodiversity conservation. The data exist and should be used.

## Data availability statement

The raw data supporting the conclusions of this article will be made available by the authors, without undue reservation. The code to create Maxent and Random Forest models is provided at <https://github.com/jbeninde/HotspotsUrbanBiodiversity>.

## Author contributions

JB and HBS designed the study. GG and TWD performed analyses on 15 species to explore Maxent settings. JB performed all subsequent analyses. JB wrote a first version of the manuscript, with assistance from GG and TWD. All authors contributed to initial reviews and editing with JB and HBS completing data revisions and edits.

## Funding

Funding for JB was provided by the UCLA La Kretz Center for California Conservation Science and by the German Science Foundation (DFG: BE 6887/1-1).

## Acknowledgments

We thank the Joey N. Curti, Alison Lipman, Morgan Tingley (UCLA), Michelle Barton (LA City), and their students, as well as Brian Brown, Jann Vendetti (NHMLAC) and Michael Wood (MykoWeb) for invaluable help compiling the list of native and non-native species, and Ryan Harrigan (UCLA) for many inspiring discussions on Maxent and Random Forest modeling.

## Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships

that could be construed as a potential conflict of interest.

The handling editor EW declared a shared committee with the authors JB and HBS at the time of review.

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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.2023.983371/full#supplementary-material>

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