Legacy effects of redlining on the distribution of greenspaces in US cities

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We investigated how a discriminatory housing policy—redlining—has shaped the spatial patterns and configurations of greenspaces throughout 177 cities in the contiguous US. Housing segregation has been a long-term development practice that has sequestered communities of color to areas with elevated environmental and public health risks. Although the lasting environmental, social, and economic impacts of redlining are clear, the impact of redlining on landscapes is still unfolding. In neighborhoods that were historically redlined, we found that (i) less overall greenspace was available, (ii) individual greenspaces were smaller and less connected, and (iii) residents belonged predominantly to communities of color and/or households had lower income. Thus, the legacy of redlining can be seen in the modern spatial patterns of urban greenspaces, and ecosystem services provided by greenspaces have been systematically absent from redlined communities for decades.

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Urban greenspaces, such as natural areas, urban parks, and large residential yards, simultaneously provide habitat for biodiversity and access to nature for urban residents (Wolch *et al.* 2014; Aronson *et al.* 2017). The size and complexity of vegetative structure within greenspaces render a wide range of ecosystem services (Wolch *et al.* 2014) that individual components of green infrastructure cannot provide alone (eg street trees, rain gardens, green roofs). Thus, the social–ecological benefits provided by these complex natural landscapes justify the protection, restoration, and creation of urban greenspaces. However, it is important that greenspaces are distributed in an equitable and just way (Zuniga-Teran *et al.* 2021).

The lack of access to quality greenspaces can compromise resilient responses to environmental change, exacerbating vulnerabilities to climate-induced changes and environmental and public health risks (Hendricks and Van Zandt 2021). Ample research demonstrates that historical disinvestment in marginalized communities has not only reduced the quality and size of local greenspaces but also left them with insufficient stewardship, as compared to greenspaces in wealthier neighborhoods, creating inequitable access to nature for communities of color and low-income residents (Zuniga-Teran *et al.* 2021).

The underlying drivers of urban inequities are often tied to historical policies related to urban planning (Jennings *et al.* 2017). Historical development often disenfranchised low-income residents and communities of color, and public policies in the US have routinely failed to address these inequities in infrastructure (Jennings *et al.* 2017; Rothstein 2017). One of the most widespread and profound policies that enshrined distributional inequities in urban areas was "redlining", a racially discriminatory housing policy that was sanctioned by the US federal government during the 1930s (Rothstein 2017).

From 1934 to 1968, redlining limited access to homeownership and wealth creation for communities of color (Aaronson *et al.* 2021). In 1933, the US Congress created the Home Owners' Loan Corporation (HOLC) to assist and promote American homeownership in the wake of the Great Depression (Aaronson *et al.* 2021). HOLC created color-coded residential maps of US cities to distinguish neighborhoods ranging from "Best" to "Hazardous" for real-estate investments (Figure 1, a and b). Areas with predominantly US-born, white populations were often categorized as A ("Best") and B ("Still Desirable")—neighborhoods that were the "safest" places for banks to invest (Aaronson *et al.* 2021). Meanwhile, areas with immigrant communities were categorized as C ("Definitely Declining") and areas with a greater number of Black Americans and ethnic minorities were categorized as D ("Hazardous") (Aaronson *et al.* 2021). The latter areas were outlined in red on the residential maps, leading to the term "redlining".

Despite redlining being outlawed in 1968, building ordinances, zoning, and the clustering of public housing—developed even before redlining began—continued to perpetuate inequities in development patterns (Hendricks and Van Zandt 2021). Thus, the practice of redlining codified the valuation of neighborhoods based on class and race, limiting investments and influencing development patterns—ultimately creating adverse social, economic, and environmental outcomes that still exist today (Aaronson *et al.* 2021).

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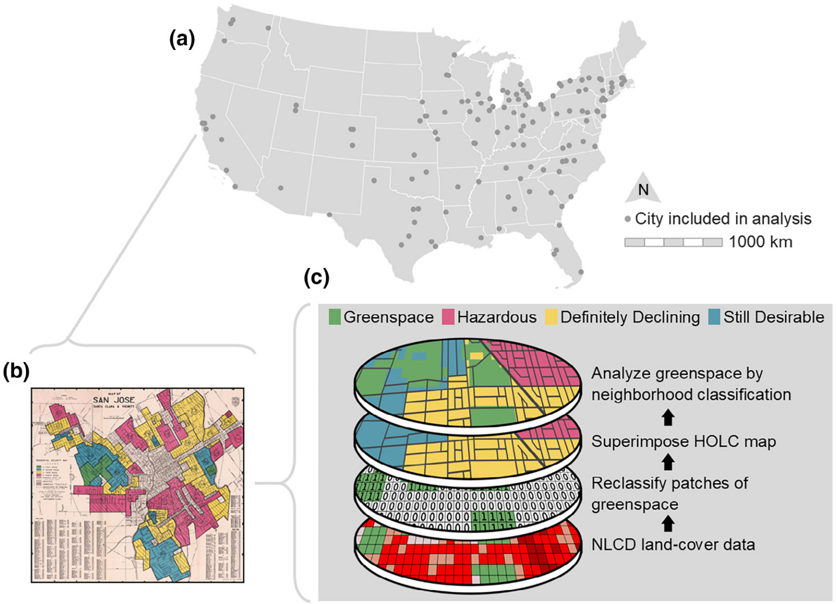

Figure 1. (a) Locations of the 177 cities within the contiguous US that were included in the analysis. (b) An example of a historical Home Owners' Loan Corporation (HOLC) map for San Jose, California. Maps like these were available for all 177 cities and digitized by the Digital Scholarship Lab at the University of Richmond (Nelson *et al.* 2021; https://dsl.richmond.edu/panorama/redlining/data). Image source: City Survey Files (1935–1940), US National Archives and Records Administration (NAID: 417204462). Image credit: Federal Loan Agency, Federal Home Loan Bank Board, HOLC. (c) Procedure for data processing. NLCD = National Land Cover Database.

While environmental justice leaders have called attention to socially driven heterogeneity in the "green" characteristics of cities, only recently have environmental researchers tried to quantify the connection between redlining and the distribution of green infrastructure in urban areas. Several studies have reported that neighborhoods that were once redlined have less present-day tree cover (Hoffman *et al.* 2020; Namin *et al.* 2020; Locke *et al.* 2021; Nowak *et al.* 2022; Salazar-Miranda *et al.* 2024), lower tree diversity, and smaller trees (Burghardt *et al.* 2023), and are generally less "green" (Nardone *et al.* 2021) than neighborhoods that were not redlined. Newly emerging research has also found that redlined neighborhoods have lower biodiversity (Schmidt and Garroway 2022; Ellis-Soto *et al.* 2023; Wood *et al.* 2024; Estien *et al.* 2024). While the ecological consequences of discriminatory land-use practices begin to emerge, it remains unclear whether these practices have had lasting effects on larger landscape patterns in cities. This body of research aims to look beyond individual components of green infrastructure and assess how historical redlining correlates with the configuration and connectivity of contemporary urban greenspaces.

Here we examined differences in the size, configuration, and connectivity of greenspaces located in US neighborhoods once categorized by the HOLC. We hypothesized that (1) neighborhoods formerly deemed "Hazardous" (D) will, to this day, have less overall greenspace and smaller and less connected individual greenspaces than those neighborhoods categorized as "Best" (A); and (2) the present-day demographics of neighborhoods categorized as "Best" (A) will continue to be predominantly white and wealthy (Aaronson *et al.* 2021), with those residents continuing to receive disproportionately more ecological benefits provided by greenspaces. Uncovering the legacy effects of discriminatory and exclusionary land-use policies on current urban landscapes will provide guidance for equitable and just urban planning and environmental design.

Methods

Landscape variables

To assess whether greenspaces differed between the four HOLC neighborhood classifications, we used the 2019 National Land Cover Database (NLCD) 30-m resolution land-cover dataset and the University of Richmond's Digital Scholarship Lab's "Mapping Inequality" database (Nelson *et al.* 2021). After downloading HOLC maps, we filtered for cities that contained all four HOLC classes ("Best", "Still Desirable", "Definitely Declining", and "Hazardous"), leaving 8527 HOLC polygons across 177 cities (Appendix S1: Table S1).

If present in a given NLCD raster cell, five land-use and land-cover (LULC) categories—Developed, High Intensity; Developed, Medium Intensity; Developed, Low Intensity; Pasture/Hay; and Cultivated Crops—were reclassified to 0 (not greenspace); all other LULC categories (including Developed, Open Space) were reclassified to 1 (greenspace).

We included the category Developed, Open Space because it contains areas where impervious surfaces account for less than 20% of total cover; areas within this category commonly include large-lot single-family housing units, parks, golf courses, and vegetation planted in developed settings (Dewitz 2021). We excluded the category Developed, Low Intensity because those areas can contain up to 49% impervious surfaces and rarely capture continuous and accessible greenspaces (Dewitz 2021). Although the effectiveness of using 30-m resolution land cover in urban and suburban areas to accurately define tree canopies has been questioned (Nowak and Greenfield 2010), we focused on the spatial patterns of larger continuous patches of vegetation and more extensive patches of greenspace, both of which can be captured by the 30-m resolution NLCD LULC categories (Grove *et al*. 2014). All spatial analyses were performed using the *terra* (Hijmans *et al*. 2024) and *sf* (Pebesma *et al*. 2024) packages in R (Figure 1c).

To create greenspace patches, raster cells classified as 1 (greenspace) that shared a common edge or common corner (Queen's case) were aggregated together as a "patch" using the *landscapemetrics* R package (Hesselbarth *et al*. 2019). We then calculated mean patch area (ha), mean core area (ha), mean perimeter-to-area ratio, and effective mesh size (ha) within each HOLC polygon (Appendix S1: Table S1). Core area was defined as the area of cells that do not have an edge with a non-greenspace cell. Perimeter-to-area ratio described the shape and edge composition of a patch; as the ratio increases, the edge per unit area expands, creating less core area. Effective mesh size was used to measure the fragmentation of patches across a landscape and is well-suited for measuring connectivity between individual patches in a landscape (Jaeger 2000). To calculate total greenspace in a HOLC polygon, we summed the number of raster cells classified as 1 within each HOLC polygon.

City-specific population density

To calculate the population density of each city, we calculated the full spatial extent of the HOLC polygons within the city and buffered that extent by 1 km. Within each city extent, we extracted the 2020 block-level population data (US Census Bureau 2022) using the *tidycensus* package (Walker and Herman 2023) in R. We then divided the total population by the area of the city extent.

Demographic variables

Using US Census 2021 American Community Survey block group data (US Census Bureau 2021), we extracted median household income from each block group overlapping a HOLC polygon. We then divided the median household income by the overall mean gross rent for each census block group to standardize cost of living. To calculate racial demographics, we extracted the following four metrics—total population; white alone, total population; Black or African American (hereafter Black) alone, total population; and Hispanic or Latino Origin (hereafter Hispanic) alone, total population—for census blocks that overlapped HOLC polygons using 2020 census data (US Census Bureau 2022). When a census block overlapped a HOLC polygon partially, we divided the population data by the proportion of the respective census block that was within a HOLC polygon (Goodchild *et al*. 1993).

Statistical analysis

We used hierarchical linear models to estimate the association between historical HOLC categorization and present-day landscape and demographic response variables. For g in $1,\ldots,G$ HOLC categories, j in $1,\ldots,J$ cities, and n in $1,\ldots,N$ neighborhood polygons across all cities, the linear predictor for all models was $\mu_n = \alpha_{j[n],g[n]} + \beta1_{g[n]}x_j$. The first term in this linear predictor, $\alpha_{j[n],g[n]}$, represented a random intercept for each HOLC category where $j[n]$ and $g[n]$ indicate the respective city and HOLC category associated to neighborhood polygon n, respectively. In the second term, $\beta1_{g[n]}$ represented a slope term for the HOLC category g of neighborhood polygon n and x_j was the log 2020 population density of each city extent j. We included this interaction term because HOLC categorization and city planning decisions may have varied as a function of a city's population size. We drew the random intercepts from a normal distribution such that $\alpha_{j,g} \sim N(\beta0_g, \sigma_g)$, where $\beta0_g$ was the overall average for a HOLC category across cities and σ_g was the estimated standard deviation of $\alpha_{j,g}$ around $\beta0_g$. Priors for each parameter or hyperparameter were $\beta0 \sim N(0, \sigma_{\beta0})$, $\beta1 \sim N(0, \sigma_{\beta1})$, $\sigma_g \sim U(0.001, 100)$, $\sigma_{\beta0} \sim U(0.001, 100)$, $\sigma_{\beta1} \sim U(0.001, 100)$.

To account for variation in the area and total population of different neighborhood polygons, we added a log-offset term, O_n, to the linear predictors for the total area of greenspace model and for the population density of white, Black, or Hispanic residents' models. For the total greenspace model, O_n was the log total number of raster cells in neighborhood polygon n. For the human population density models, O_n was the log total population of neighborhood polygon n.

Our data then arose as a normal random variable, $y_n \sim N(\mu_n, \sigma)$, where y_n was the log response variable for each neighborhood polygon n. Here we used a vague uniform prior for σ, $\sigma \sim U(0.001, 100)$. Models were fit using a Markov Chain Monte Carlo (MCMC) algorithm implemented in Nimble version 0.12.2 (de Valpine *et al*. 2017) using the *nimble* package in R. Four parallel chains were each run from random starting values. The first 10,000 iterations from each chain were discarded and every third iteration was retained to reduce autocorrelation among the samples. A total of 40,000 iterations were obtained for each model. Model convergence was assessed by checking that the Gelman–Rubin diagnostic statistic for each

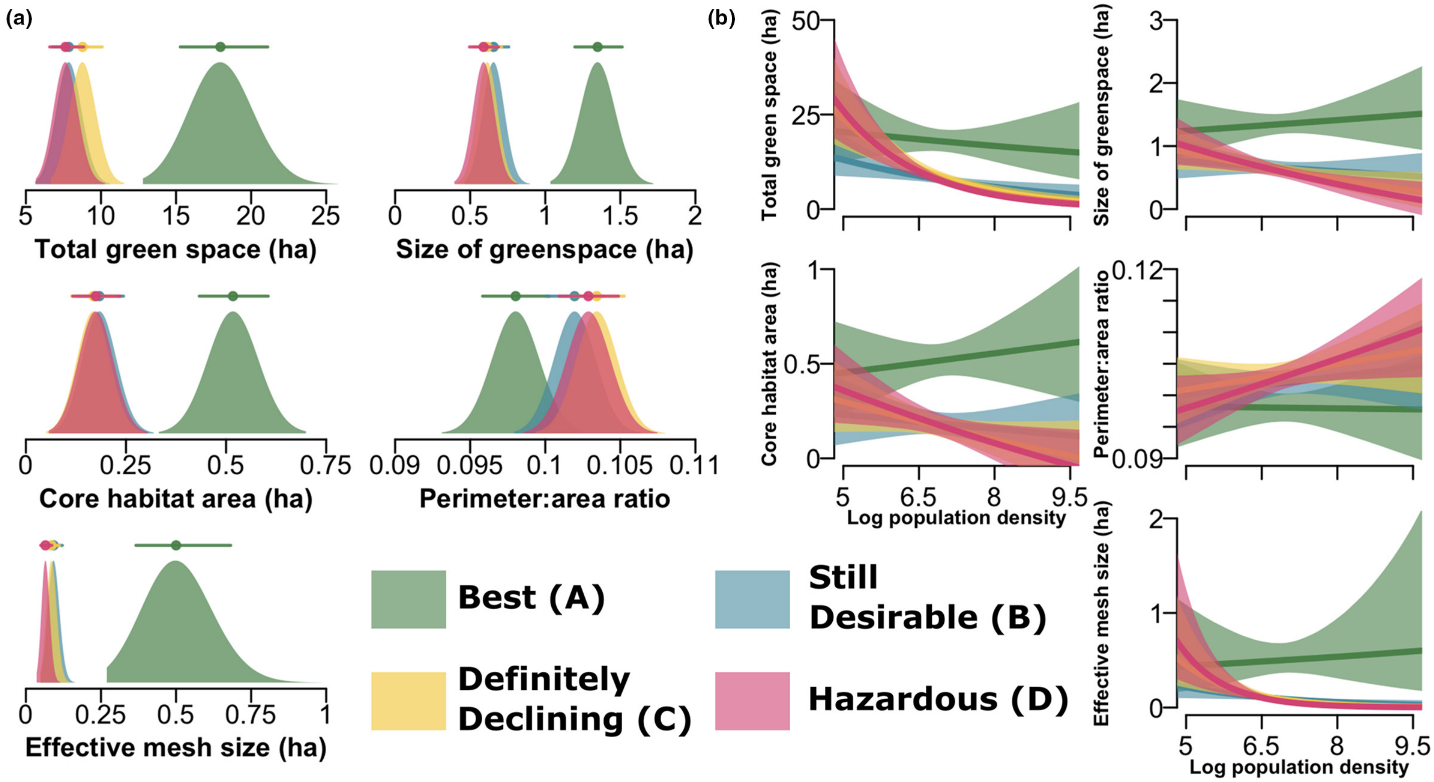

Figure 2. To the present day, neighborhoods categorized as "Best" by the HOLC (category A neighborhoods) have more overall greenspace and larger, less irregular, and more connected individual greenspaces. (a) Posterior distributions of the average for each response variable in the four HOLC categories (A, B, C, and D) and (b) the relationship between the population density of a city and each response variable. In (a), horizontal bars represent the 95% credible intervals (CIs), and circles represent the median value of each category. A colorblind-friendly version of this figure is available in the Appendix (Appendix S1: Figure S1). For city-specific results, see the CitySpecificModelResults.pdf in Gallo *et al.* (2026) at https://doi.org/10.5061/dryad.qv9s4mwp3.

parameter was <1.1 (Gelman and Rubin 1992) and by visually inspecting trace plots of the MCMC samples. We considered variables to be statistically different if the 95% credible intervals did not overlap 0.

Results

On average, category A neighborhoods had more total greenspace than any other neighborhood category (Figure 2a). We also found that greenspaces in category A neighborhoods were larger, contained more core habitat, had a lower perimeter-to-area ratio, and were more connected (Figure 2a). In more densely populated cities, each metric worsened in category D neighborhoods (Figure 2b; Appendix S1: Table S2). For example, when the human population density increased by one standard deviation (1725 per square kilometer) over the mean for a specific city, our model predicted a 60% decrease in total greenspace area, a 29% decrease in greenspace size, a 57% decrease in core habitat area, a 1.8% increase in perimeter-to-area ratio, and a 128% decrease in connectivity in category D neighborhoods (Figure 2b; Appendix S1: Table S2). Finally, on average, category A neighborhoods had significantly fewer residents who were Black and Hispanic while category C and D neighborhoods had more residents who were Black and Hispanic; likewise, category A neighborhoods had higher densities of higher-income residents while category C and D neighborhoods had higher densities of lower-income residents (Figure 3).

Discussion

Our results demonstrate that patterns of nature inequity match patterns of historical redlining. Neighborhoods that were once redlined have less greenspace and those greenspaces are smaller, less natural in shape, and less connected. Furthermore, residents who currently live in formerly redlined neighborhoods belong predominantly to communities of color with less income. These findings demonstrate that the wide range of ecosystem services provided by greenspaces has been systematically absent from redlined communities.

We found marked differences in the spatial patterns of urban greenspace correlated to HOLC categorizations of neighborhoods. We also found that these inequities were most severe in cities with high human population densities. More specifically, greenspace size and core habitat increased in category A neighborhoods as population density increased

(Figure 2b). Yet these same metrics declined precipitously in category B, C, and D neighborhoods as cities became larger (Figure 2b). On average, category C and D neighborhoods had the highest overall population densities, likely creating higher needs for development and thus reducing the available area for greenspace. Category D neighborhoods (redlined neighborhoods) and category A neighborhoods (non-redlined neighborhoods), however, had similar population densities. Nevertheless, in more densely populated cities, category D neighborhoods were still characterized by significant declines in greenspace metrics. These results suggest that when decisions were made to protect, maintain, or create greenspaces in the face of urban development, residents in formerly redlined neighborhoods continued to lose the opportunity for greenspace, while residents in category A neighborhoods may have had enough political capital to protect or increase greenspace (Swope *et al.* 2025). Urban centers and populations are projected to rapidly grow (Batty 2011), and we provide evidence that if historical inequities are not addressed—specifically in regard to greenspace—then environmental injustices borne from systemic racism and segregation will worsen as cities rapidly densify.

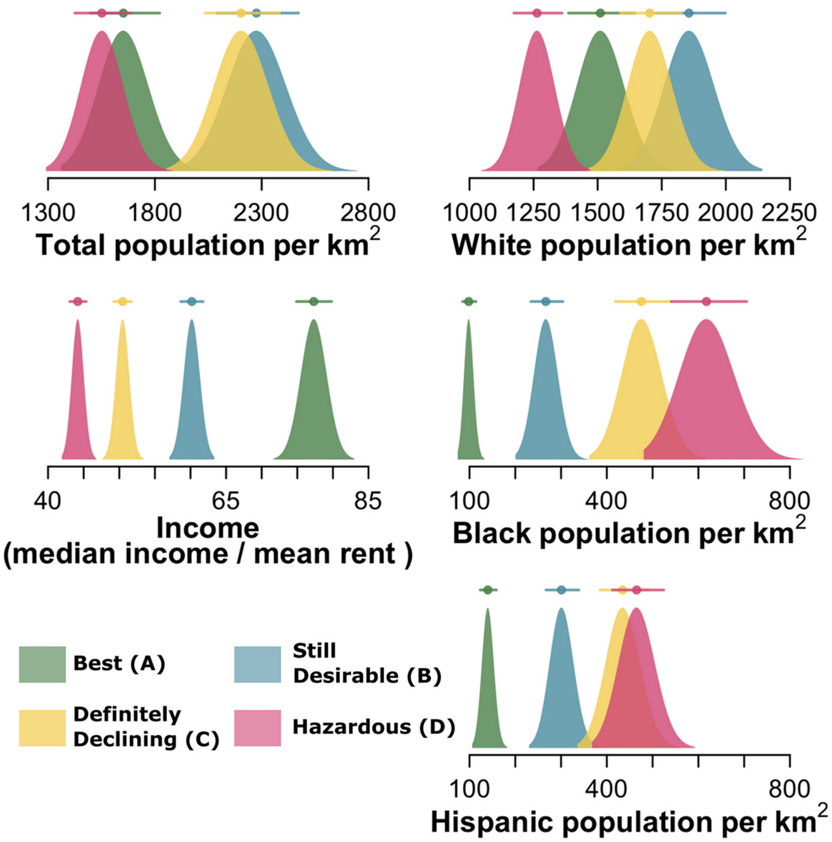

Figure 3. Neighborhoods that were once predominantly wealthy and white, on average, remain higher income in the present day. Neighborhoods that were once redlined have higher population densities of Black and Hispanic residents and those residents have lower incomes. Posterior distributions of the average for each response variable in the four HOLC categories. Horizontal bars represent the 95% CIs, and circles represent the median value of each category. A colorblind-friendly version of this figure is available in the Appendix (Appendix S1: Figure S2).

Differences in demographic access to greenspaces

Historical migration patterns, efforts to desegregate schools and neighborhoods, and gentrification may have changed the demographics of once highly segregated neighborhoods (Woldoff 2011), including historically redlined neighborhoods, which today remain highly segregated (Aaronson *et al.* 2021). However, we found that Black and Hispanic residents remain at the highest population densities in category D neighborhoods. These results further underscore how communities of color are more likely to have limited access to larger and more intact greenspaces, limiting the ecosystem services gained from more natural landscapes. Historical inequities in homeownership and housing quality have persisted over decades and contribute to the ever-widening racial wealth gap in the US (Figure 3; Woldoff 2011; Aaronson *et al.* 2021). Here, we provide further evidence that redlining has created additional contemporary harms to marginalized communities, by also simultaneously constraining access to natural capital.

Looking to the past to plan for the future

For years, environmental justice scholars have demonstrated that communities of color and low-income neighborhoods are more polluted (eg Bullard 1994), are hotter (eg Hoffman *et al.* 2020), have declining infrastructure (Hendricks and Van Zandt 2021), and are more vulnerable to environmental and public health risks (Hendricks and Van Zandt 2021). Mounting evidence suggests that urban greenspaces provide a degree of environmental protection such as improved air quality (Escobedo *et al.* 2011), protection from large storm events and flooding (Hendricks and Van Zandt 2021), and urban cooling (Cady *et al.* 2020). In addition, accessible greenspaces can support positive health outcomes such as physical, mental, and social well-being through direct physical activity and the enjoyment of biodiversity (Fuller *et al.* 2007; Wolch *et al.* 2014), and public greenspaces act as focal points around which community well-being, social inclusion, and social capital can grow (Frumkin *et al.* 2017).

Incorporating more greenspaces into urban design has been proposed as a mechanism to mitigate grand environmental and public health challenges (Hendricks and Van Zandt 2021). Clearly, well-distributed networks of accessible greenspaces can simultaneously support individual and community health and wellness (Egerer *et al.* 2024) and increase biodiversity (eg increased habitat and more connected animal movement corridors; McKinney 2002; Kong *et al.* 2010). However, our results show a clear disparity in access to nature through inequitable distributions of greenspaces in US cities. These inequities have left marginalized neighborhoods more susceptible to environmental and public health risks for more than 90 years, demonstrating that injustices and inequalities forged into the modern design of cities will take generations to rectify. Unless efforts are made to infuse equity into modern greenspace planning, restoration, and management practices, environmental inequities will continue, and these vulnerabilities will be intensified in the future.

Addressing a major challenge

While our results demonstrate a clear need for more greenspace in marginalized neighborhoods, this is an overly simplistic solution. Plans for urban greening will inevitably intersect with affordability and gentrification (Pearsall 2018; Quinton *et al.* 2024). In many cases of urban "greening", the urban poor are financially displaced from their homes, livelihoods, and social communities through increased property values and rent (Wolch *et al.* 2014) or physically displaced to make room for green infrastructure projects (Zuniga-Teran *et al.* 2021). In such cases, the benefits of nature continue to be unequally distributed to the affluent and powerful (Anguelovski *et al.* 2020).

Combating the negative feedback loop between urban greening and displacement requires strategies that embed social equity and reconciliation as their foundation. Several programs in cities throughout the US (including Baltimore, San Antonio, and Los Angeles) have worked to improve green infrastructure while also guarding against displacement through targeted housing legislation (Rigolon and Christensen 2019). However, the long-term success of such programs may be dependent on the ability to divorce greenspace equity from current US real-estate valuation and practices (Mullenbach *et al.* 2022). Future research should analyze these specific programs to better understand the gentrification processes associated with coupled green infrastructure improvements and anti-displacement policy.

Self-reflections on our own fields of research

A compounding challenge is within individual fields of practice. For example, we (the authors) are ecologists, conservation scientists, and landscape planners, and our work often focuses solely on ecological attributes, such as high levels of existing biodiversity or the presence of rare and underrepresented ecotypes (eg Snep and Clergeau 2020). When considered in our fields, human interests often fall under the very broad auspices of protecting ecosystem services (Snep and Opdam 2010). However, championing or justifying only areas of "high-quality" nature for preservation, management, or restoration exacerbates these inequities. Our findings add to the list of studies showing that human decisions influence the landscapes that we observe, study, and then advocate on behalf of (Warren *et al.* 2010). By ignoring the historical justifications of those land-use decisions, we have been complicit in perpetuating inequities in nature access. Thus, the fields of ecology, conservation, and landscape planning must recognize the influence that institutionalized racism and human decisions have had on ecological and evolutionary trajectories (Schell *et al.* 2020). Collectively, we must refocus our efforts toward preserving, restoring, and creating natural spaces that benefit community needs.

Conclusions

By uncovering the lasting effect of historical inequities, cities can begin to develop justice-centered planning and designs. Two core tenets of environmental justice are that no group of people—especially marginalized communities—should bear a disproportionate share of environmental harms and that environmental protections and benefits should be distributed equitably (Bullard 1994). Our results highlight that past, present, and future policies will have lasting effects on the structural and natural composition of cities. Thus, urban planners and conservation practitioners must center environmental justice principles in contemporary landscape planning (Zuniga-Teran *et al.* 2021; Mullenbach *et al.* 2022). To reduce nature inequities in cities, those who can influence urban design (eg urban planners, policymakers, land managers) must rectify past transgressions (focusing on the most marginalized first), support urban policies written through an equitable and anti-racist lens, and interrogate biases in planning and design procedures. It is our moral and civic responsibility to reduce inequities in the face of global environmental change, and we hope that this analysis spurs future research that further explores the nuances of our findings, thereby making cities a better place for people and biodiversity.

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Data Availability Statement

All data and scripts (Gallo *et al.* 2026) are available in Dryad at https://doi.org/10.5061/dryad.qv9s4mwp3.

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Supporting Information

Additional material can be found online at http://onlinelibrary.wiley.com/doi/10.1002/fee.70056/suppinfo