URBAN ECOLOGY AND HUMAN HEALTH

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frontiers Research Topics

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URBAN ECOLOGY AND HUMAN HEALTH

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Editorial: Urban ecology and human health

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Editorial on the Research Topic
Urban ecology and human health

Introduction

The character, design and biodiversity of urban ecosystems have both beneficial and negative impacts on human health and wellbeing (Flies et al., 2019). However, the ways in which urban ecosystems influence human health—positively and negatively—remain unclear. Current knowledge is dominated by research in large cities and the global north, and therefore is incomplete and biased (Kendal et al., 2020). The diverse chapters in this special issue contribute to the "Urban One Health" and "Ecology with Cities" frameworks (de Leeuw, 2021; Ellwanger et al., 2022) that bring together the ecological, social, political, and community engagement aspects of ensuring public health among the intricacy of interactions, both between organisms and within complex, changing, and diverse urban environments.

Greenspace ecosystem services and disservices

Urban green and blue spaces provide many diverse beneficial ecosystem services (ES; Flies et al., 2017; Lai et al., 2019; Mavoa et al., 2019). They play an important role in mitigating air pollution from exhaust and industrial emissions, and health-impacting haze from wildfires, as in Malaysia (Jaafar et al.). An examination of urban ES in China found that urban agglomerations typically had lower ES than their surroundings (Shao et al.), but among other variables, the presence of green infrastructure (GI), like woodlands increased urban ES.

Urban greenspaces and GI nevertheless can also provide ecosystem disservices (Lyytimäki and Sipilä, 2009) for example, as a mixing ground for native and introduced species and between human and non-human animals. These novel urban interactions may change disease patterns and adaptation by vector species and their pathogens. The Chagas disease vector, *Triatoma dimidiate,* may have adapted to the urban environment (de Oca-Aguilar et al.) as indicated by changes in the thorax and antennal phenotype of this species between urban and rural areas, suggesting adaptation leading to altered sensory and locomotion performance. However, as these differences seem not to have altered the insect's fecundity/fitness, the impact of the insect's urban adaptation on human health is unclear.

Urban sustainability, urban greening, and human wellbeing

Less tangible are the impacts of urban environments on human mental health and subjective wellbeing. For example, biodiversity can be supportive of human health and wellbeing (Taylor and Hochuli, 2014; Flies et al., 2017; Mavoa et al., 2019; Schebella et al., 2019) and urbanization can homogenize animal and microbial biodiversity (Johnston et al., 2014; Morelli et al., 2016; Flies et al., 2020). However, the biodiversity-wellbeing mechanisms and how they impact urban greenspace benefits remain unclear (Lai et al., 2019). Untangling the role of biodiversity in the greenspace-wellbeing connection will require the type of interdisciplinary effort for which Hedin et al. make a plea and provide a framework.

Further exploration of the relationship between greenspace and wellbeing in two cities in Kenya and Thailand found wellbeing was most strongly influenced by availability of basic infrastructure (waste removal, accessible clean water; Cinderby et al.). Once these amenities were in place, social (crime and tenure) and environmental (noise and air quality) issues became important for community wellbeing. Spending time in urban greenspaces could mitigate city-living stresses even for residents of informal neighborhoods. Cinderby et al. demonstrate the need for diversity and equity in public realm space provision to ensure social and spatial justice.

The COVID-19 pandemic lockdowns and restrictions changed most people's use of urban greenspace. During a COVID-19 lockdown in Brisbane, Australia, greenspace use patterns changed, varying by both individual and greenspace characteristics (Berdejo-Espinola et al.). Places with access to blue spaces and good accessibility (carparks/public transport) experienced increased use; but places with foliage height diversity had decreased use. More females than males changed their greenspace visitation frequency during COVID-19. Females had increased reliance on greenspaces for social and family interactions and spiritual reasons, while males for nature interactions and mental health benefits during COVID lockdowns. Clearly, different times of stress and national crisis lead to major behavioral changes during which urban greenspace is a great asset for human wellbeing and morale. Understanding such changes during crises will help develop more resilient urban greenspace planning and policies.

Private and community vegetable gardens were a refuge for many during COVID lockdowns (Marsh et al., 2021) are key parts of urban greenery. Community gardens support social, physical and dietary health (Egli et al., 2016) and urban agriculture is a possible nature-based solution to socio-ecological challenges in cities (Kingsley et al., 2021). The ways people interact with gardens differ between countries, with few studies in the global south. In a South Africa, Du Toit et al. examined food security in a community garden scheme designed to encourage small garden plots in a community where 39% of participants reported hunger affecting the entire household and 51% were at risk of hunger. Although 72% participants planted fruits and vegetables, the gardens contributed little to food security; Du Toit et al. explore the reasons, including cultural and food-purchasing practices.

Planning and design for urban nature to improve health and wellbeing

Understanding the ways different communities, ethnic groups, and age cohorts use urban greenspaces is an interdisciplinary effort helping toward Urban One Health. Children's views are seldom sought when designing greenspaces (Vidal and Seixas). Often urban children would rather play in a seminatural local creek area than in a formal municipal park playground. Vidal and Seixas argue that community plans ought to include special "Children Green Infrastructure" designed to link children to nature where they live, learn, and play.

Such provision for individual and groups extends into the design of other green infrastructure (GI) like green roofs and walls. Adequate planning, design, and management, especially for water in changing climates, can maximize ES health benefits of GI (Sang et al.). Sometimes trade-offs occur in planting decisions: aesthetic choices aligning with human preferences can result in greater wellbeing benefits, while opting for native species or biodiverse combinations may result in greater ES. This trade-off may be circumvented by using color theory to create aesthetically pleasing, biodiverse designs for living walls (Thorpert et al.).

Urban greenspace and the resultant ecosystem services benefit human health everywhere, but values, perceptions, uses and risks remain diverse, between and within nations and communities. The more we learn about them, the more we can help communities and individuals prosper and enjoy urban GI.

Author contributions

ID wrote the initial draft of this Editorial summarizing the Research Topic articles. EF revised the draft and contextualizing the Research Topic articles in the broader literature. All authors revised subsequent drafts and finalized the article for submission. All authors contributed to the article and approved the submitted version.

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Assessing Inequalities in Wellbeing at a Neighbourhood Scale in Low-Middle-Income-Country Secondary Cities and Their Implications for Long-Term Livability

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To ensure future sustainability, cities need to consider concepts of livability and resident wellbeing alongside environmental, economic and infrastructure development equity. The current rapid urbanization experienced in many regions is leading to sustainability challenges, but also offers the opportunity to deliver infrastructure supporting the social aspects of cities and the services that underpin them alongside economic growth. Unfortunately, evidence of what is needed to deliver urban wellbeing is largely absent from the global south. This paper contributes to filling this knowledge gap through a novel interdisciplinary mixed methods study undertaken in two rapidly changing cities (one Thai and one Kenyan) using qualitative surveys, subjective wellbeing and stress measurements, and spatial analysis of urban infrastructure distribution. We find the absence of basic infrastructure (including waste removal, water availability and quality) unsurprisingly causes significant stress for city residents. However, once these services are in place, smaller variations (inequalities) in social (crime, tenure) and environmental (noise, air quality) conditions begin to play a greater role in determining differences in subjective wellbeing across a city. Our results indicate that spending time in urban greenspaces can mitigate the stressful impacts of city living even for residents of informal neighborhoods. Our data also highlights the importance of places that enable social interactions supporting wellbeing–whether green or built. These results demonstrate the need for diversity and equity in the provision of public realm spaces to ensure social and spatial justice. These findings strengthen the need to promote long term livability in LMIC urban planning alongside economic growth, environmental sustainability, and resilience.

Keywords: wellbeing, equity, urban, planning, livability, greenspace (Min5-Max 8), global south

INTRODUCTION

With the global transition to urban living, cities need to become sustainable in the broadest sense, which increasingly includes concepts of wellbeing and quality of life alongside environmental and economic considerations (Leach et al., 2014). The Habitat III New Urban Agenda (NUA) includes a recognition that to maximize the benefits of urbanization we need to promote environmentally sustainable and resilient urban development (WHO, 2016; Habitat III Secretariat-United Nations, 2017). How to balance the need for urban environmental sustainability (which encompasses concepts of circular economies, resource conservation, and energy efficiency) which typically leads to densification, with the need for resilience (ability to withstand shocks and disasters), which entails diversity, remains an ongoing challenge (Elmqvist et al., 2019). The inclusion of considerations of wellbeing in urban sustainability entails that residents should not only live in a clean, safe and healthy spaces but should also have equity of opportunity to act and move around in health-promoting environments. In fast-changing cities, urban development can mean the loss of landcover supplying ecosystem services which provide multiple benefits in terms of the resilience to disasters, climate adaptation and support wellbeing (Derkzen et al., 2017). For future sustainability we need to better understand what ability different urban forms have for delivering these multi-functional benefits of promoting human-wellbeing, being environmentally sustainable and supporting resilience (Grimm et al., 2008; Hansen et al., 2017). This evidence is particularly lacking from the Global South where cultural and environmental conditions make their challenges and potential solutions distinct (Nagendra et al., 2018; Pauleit et al., 2021).

Rapidly developing cities in low-middle- income countries (LMIC) represent unique challenges and opportunities for the delivery of such sustainable development. The current rapid urbanization of sub-Saharan Africa is putting pressure on natural resources and the environment, increasing environmental- and climate change-related vulnerabilities, urban poverty and the proliferation of informal settlements (AFDB, 2013; UN-Habitat, 2015). These challenges are exacerbated by weak urban planning and management institutions, and inadequate urban governance (UN-Habitat, 2015; Smit, 2018). South East Asia is 49% urban while South Asia is at 36% (United Nations: Department of Economic and Social Affairs Population Division, 2019). However, these percentages are increasing faster than urban infrastructure provision leading to over 130 million South Asian living in informal settlements (Ellis and Roberts, 2016) and facing problems of inadequate housing, poor air quality and sanitation. Meanwhile, the Asia Pacific region is one of the most exposed to the changing climate, and is projected to see extremes in precipitation, temperature, and sea level rise, with the associated economic, social and physical costs (Asian Development Bank, 2017) including in urban contexts.

Unfortunately, such unplanned growth often outpaces infrastructure provision and occurs at the expense of a city's ecological foundations, undermining resident's wellbeing and the city's sustainability (McPhearson et al., 2016). Addressing this entails moving beyond concentrating on only meeting basic needs towards enabling residents to achieve their aspirations. Developing more sustainable cities does not merely concern the improvement of infrastructure and systems mediating urban life, but also needs to consider the social aspects of city living, such as people's satisfaction, experiences and perceptions of their everyday environments (Corburn, 2017; Shackleton et al., 2021). Achieving this requires that city authorities take sympathetic care of residents (Winkler, 2012) meaning decision makers need to have a greater understanding of a cross-section of their people's needs and wants, aiming for ideals of equity, equality, social and spatial justice most relevant in LMICs (Soja, 2010; Bai et al., 2018; Zuniga-Teran and Gerlak, 2019).

Wellbeing supporting environments that promote mental health allow individuals realize their own abilities, cope with the normal stresses of life, work productively and fruitfully, and contribute to their community (World Health Organization (WHO), 2014). Research is increasingly demonstrating the importance of immersion in nature for health including both mental and spiritual wellbeing and physical health (both from direct opportunities afforded for recreation and socializing (Bertram and Rehdanz, 2015; Ahirrao and Khan, 2021) but also urban agriculture). Studies, including some from the Global South, indicate using urban greenspace can reduce residents stress (Roe et al., 2013; Adhikari et al., 2019), improving cognitive performance (Berman et al., 2008; Dadvand et al., 2015), decreasing depressive symptoms (Bratman et al., 2015) and increasing relaxation (Neale et al., 2019). Even if not directly accessing natural spaces, all urban residents can reap the benefits of regulating services provided by green and blue infrastructure such as shade cooling, air quality improvement, noise buffering or flood mitigation that again connect to physical and mental health. However, rapidly developing and changing urban environments, driven by desires to maximize land use, means that urban greenspaces are often converted into built and paved areas. The negative impacts of reducing urban nature are long-term; difficult to reverse; and increasingly important as cities develop. This link between human and ecosystem health is conceptualized as "Ecological Public Health" which represents the complex interactions between humans and the urban biosphere. A recent review concluded "better informed decisions using neighbourhood-level health determinants datasets stand to improve the environments and societies in which we live, particularly in LMICs" (Thomson et al., 2019) supporting calls from previous studies (Nero, 2017).

This paper explores these multiple dimensions of city developments impacts on resident's wellbeing in LMIC contexts. We present results from two complementary LMIC cities exploring the interaction of urban form on wellbeing. Our findings address knowledge gaps that call for greater granularity of data to explore interactions with income, gender and environment (Patel et al., 2017). Our analysis considers the equity implications of this relationship contributing to recommendations for future city development pathways in LMIC settings to maximize sustainability that incorporates concepts of livability and wellbeing.

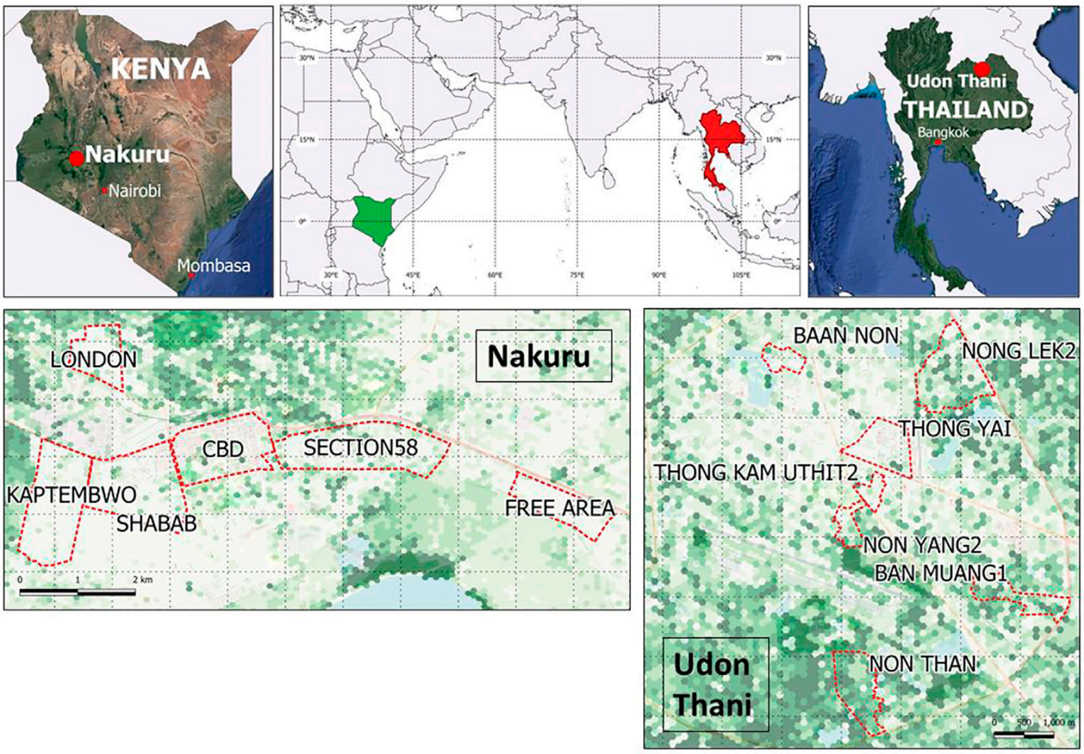

FIGURE 1 | case study city locations and surveyed neighbourhoods. Base maps indicate 100 m width hexed grid relative greenness derived from Landsat imagery processed to show Normalised Difference Vegetation Index (NDVI).

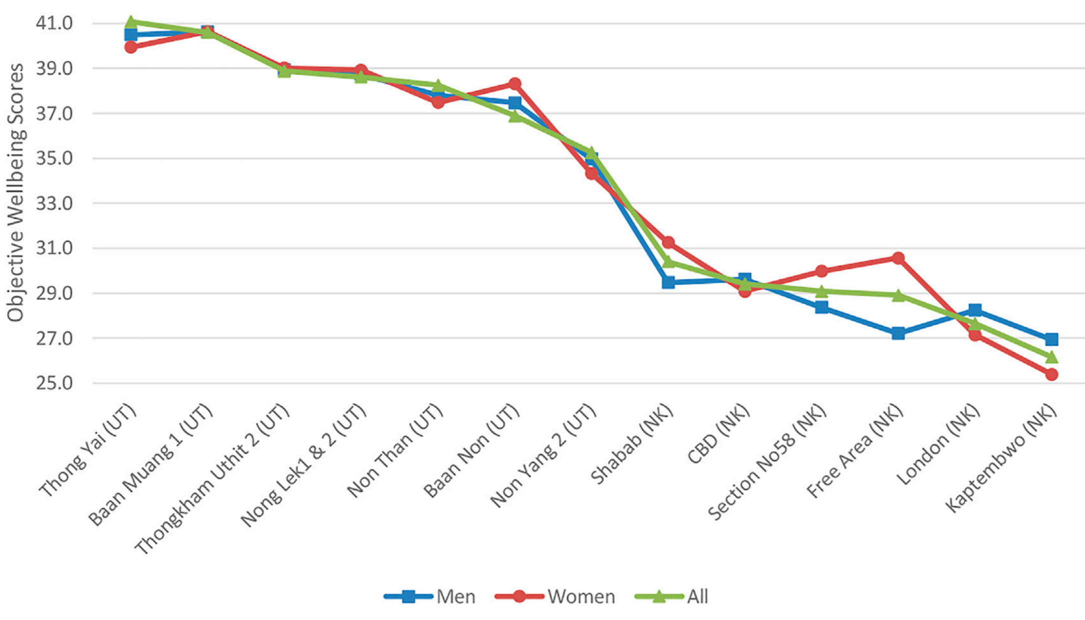

FIGURE 2 | Effects eleven surveyed dimensions of socio-environmental conditions have upon objective wellbeing summed by neighbourhood (**Note:** UT indicates Udon Thani; NK indicates Nakuru).

Research Questions

We addressed these topics in relation to three interlinked questions:

1. How are objective aspects of wellbeing (distributed according to socioeconomic and sociodemographic characteristics) related to subjective assessments of wellbeing (life satisfaction)?
2. How is the relationship between subjective wellbeing mediated by the quality of urban environments?
3. What are the implications for urban development to achieve equitable wellbeing improvements?

METHODS

Case Study Site Selection

To investigate these questions in real world settings two comparable but contrasting secondary cities of the Global South were selected (based on criteria including population and growth rates, mix of formal and informal growth, range of environmental concerns, relatively under researched) as representative examples in which to explore these concepts (see **Figure 1** for details).

Nakuru, located within the Great Rift Valley, 160 km northwest of Nairobi, is the fourth-largest city in Kenya (after Nairobi, Mombasa and Kisumu) and the county capital. Nakuru lies at an altitude of 1,850 m and has a Mediterranean climate (Köppen-Geiger climate classification is Csb) remaining temperate throughout the year with no annual dry season. According to the County Integrated Development Plan 2018-2022, Nakuru town had an estimated population of 405,000 in 2018 which is expected to reach 458,000 by 2022 (a 13% increase). It has a mixture of built environments, including informal and unplanned settlements and both green and blue spaces. Rapid growth in Nakuru is putting development pressure on the public realm including greenspace.

Udon Thani in northeast Thailand is a small city of 130,000 residents facing rapid development due to its strategic location near the Laotian border. The city has a tropical savanna climate (Köppen-Geiger classification Aw) with warm dry winters followed by a 6-month monsoon season. Through the Udon Charter for 2029, a multi-stakeholder vision for the city, the city is committed to achieving six policy points, driven by the objective of becoming a green city focused on MICE (Meetings, Incentives, Conventions and Exhibitions). It seeks to have a walkable urban core, invest in green transport and green infrastructure including parks and public realm spaces.

Surveys

Wellbeing can be considered a key component for a person's quality of life and encapsulates both objective and subjective elements. The objective dimensions define wellbeing in terms of quality-of-life indicators including access to basic needs resources (e.g. food, housing, income) and social attributes (education, health, political voice, social networks). The subjective dimension emphasizes people's own life evaluations including satisfaction (a cognitive evaluation) and happiness (relative emotional state) (Western and Tomaszewski, 2016). Subjective wellbeing encompasses hedonic functions such as pleasure attainment and pain avoidance, and eudemonic linked to a meaningful existence related to personal functioning (within individuals own mental and physical constraints) (Nordbakke and Schwanen, 2013).

This paper reports on the findings from two surveys detailed below: a bespoke neighbourhood survey investigating dimensions of socio-economic, environmental and wellbeing conditions (see 2.2.4.1 and 2.2.4.2 below); and a validated scale questionnaire exploring individual mood effects in different urban settings (see 2.2.4.3). All the survey tools received individual ethical approval via the relevant University of York, UK committee and participants gave informed consent. To facilitate accurate completion surveys were translated into local languages appropriate for each city.

Neighbourhood Wellbeing Survey Recruitment and Data Collection

The wellbeing survey was carried out across diverse neighbourhoods (six in Nakuru (during November 2018 dry season) and seven in Udon Thani (during December 2018 warm season)), identified in collaboration with city officials and local project partners, which represented a cross-section of local environmental, social and economic conditions ranging from central to suburban locations, including fully to partially serviced areas in terms of public utilities neighbourhood (see supplementary materials: two Assessment of socio-economic conditions). Adults (over the age of 18) were recruited through on-street intercepts in each neighbourhood aiming for a gender balanced sample.

Urban Settings Survey and Data Collection

To assess the impact of different types of urban spaces upon mood, a young (18–30 years) gender balanced, self-reported healthy, cohort of residents were recruited. This cohort was purposively selected to control for impacts of ageing on mobility and wellbeing as these participants also undertook recordings of heart rate variability (reported on in an upcoming paper) in different urban locations. Participants undertook transect walks between a busy built public realm space (market) and a quieter greenspace (public park) via other important infrastructure (e.g. bus interchange). These start and end points were selected to maximise the contrast in terms of type of public realm space–green vs grey; busy vs quiet. To control for the effects of direction the cohort was randomly sub-divided to undertake the walk in opposing directions (see supplementary materials part 2: Assessment of socio-economic conditions). Transect walks and mood surveys were undertaken in april 2019 during Udon Thani's hot season and Nakuru's wet season. Walks were only undertaken on dry days and in early morning to avoid high temperatures.

Assessments of Objective Wellbeing

The neighbourhood survey included questions on the impact on respondent's wellbeing of eleven different environmental and social factors. The impacts ranged from large (scored 1) to no impact (4) on a forced four-point Likert scale. By summing the participant's response scores across the eleven variables, a composite indicator of objective wellbeing was created. The raw data was scaled for graphing to range between greater than zero and the range maximum by subtracting the integer value of the minimum objective wellbeing score for each city. This improves visualization but means the graphed values are city specific and should not be directly compared (see supplementary materials:

To assess the relative affluence of the different surveyed neighborhoods the calculated mean sum of the ranked values for homeownership, employment status and job description were used. Job description was rated from employee upwards through managerial to business owner or professional. Two independent variables were used to validate the composite indicator of affluence, namely relative access to sanitation and access to water.

Assessments of Subjective Wellbeing

Wellbeing

The neighbourhood survey (translated into local languages as appropriate) utilized the Short Warwick Edinburgh Mental Wellbeing Scale (SWEMWBS) that assesses subjective wellbeing through seven questions rated on a five-point Likert scale which have been validated for construct validity (Stewart-Brown et al., 2009). The scale has revealed national wellbeing averages in the UK (McFall and Garrington, 2011) and successfully used in Europe (Koushede et al., 2019), Asia and Africa (Neale et al., 2019). This scale asks respondents to consider dimensions of life related to their wellbeing over the past 4 weeks.

Perceived Stress

Stress is inevitable and healthy factor of life. However, the duration and frequency of stress as well as someone's belief and ability to return to a non-stressed state has significant implications for overall health and wellbeing. The Perceived Stress Scale (PSS), is a measure of sub-chronic stress (Cohen, 1983) which evaluates subjective levels of stress over the previous 2 weeks. Survey questions were designed to measure how unpredictable, uncontrollable, or overloaded respondents find their lives. The PSS has been used successfully in African and Asian contexts (Cohen, 1983; Neale et al., 2019) making it appropriate for this cross-cultural assessment. Higher scores on the PSS refer to higher stress (which is problematic) and on SWEMWBS to higher wellbeing (which is beneficial).

Mood

The urban settings survey used the Acute Subjective Mood measured by the University of Wales Institute of Science and Technology (UWIST) Mood Adjective Checklist (MACL) to determine acute subjective mood changes between our two key locations (market and park). MACL is a 24-item checklist that gives an acute psychometric measure of hedonic tone (valence), stress and (physical) arousal, shown as three scores. Respondents are required to complete the questionnaire before and soon after completion of activity to ensure measurement of momentary shifts in mood. The arousal scale measures feelings of subjective energy. The stress scale measures feelings of subjective tension and the hedonic tone scale measures overall pleasantness of mood and is associated with feelings of somatic comfort and wellbeing. Scores are obtained from summation of individual item scores pertaining to each of the three mood components.

The age and gender distribution of survey participants in each city can be seen below in **table 1** (also see supplementary materials part 1: Detailed breakdown of participant numbers by neighbourhood).

Assessments of Urban Infrastructure

Natural urban spaces, often referred to as urban greenspace (UGS), have been defined as vegetated urban spaces (Taylor and Hochuli, 2017). Whilst this definition is not globally appropriate as it prioritizes green–for our case study locations climatic-ecological settings it remains relevant for our analysis. To evaluate the impact that urban infrastructure availability and use has on wellbeing two data sources were utilised. Firstly, the participant's response to questions on accessibility (do you live within walking distance of . . .) and how much time they spend in these location (how many hours do you spend in these spaces (both within and beyond walking distance)) of greenspace and built public realm spaces. A walking 'distance' of 15 min was given as a guide to the participants in answering the accessibility question. Secondly, to quantify greenspace, satellite imagery pre-processed to indicate mean normalised difference vegetation index (NDVI) values for the year our survey was undertaken was obtained from Climate Engine which uses Google's Earth Engine for on-demand processing of satellite data.

Spatial Analysis

To assess the quantity of greenspace satellite imagery processed to depict vegetation (normalized difference vegetation index (NDVI)) was accessed. Landsat imagery was processed by Climate Engine (climateengine.org) to determine the mean NDVI values for the 12 months prior to the survey period to assess the most recent variations in greenness that could affect wellbeing. These images were clipped to official neighbourhood boundaries for both cities and the distributions of 29 m pixel values determined for input to statistical tests.

Statistical Analysis

One-way ANOVA and Chi2 tests were utilized in IBM SPSS Version 26 to assess the differences between variables based upon age, gender and location. Tukey and Cramer V post-hoc tests determined the significance of any emerging associations or differences. Linear regression analysis was used to assess the explanatory strength of relationships between variables. Kruskal-Wallis H test was used to assess the differences in the distribution of NDVI pixel values by neighbourhood.

RESULTS

Objective Wellbeing Dimensions

The following sections results present findings relevant to our initial research question of 'how are objective aspects of wellbeing (distributed according to socioeconomic and sociodemographic

TABLE 1 | Survey participant demographics (Note: Thailand median age 40.1 yrs vs Udon Thani Neighbourhood wellbeing survey median age 46.26 years; Kenya median age 20.1 year vs Nakuru Neighbourhood wellbeing survey median age 41.96 years (country demographic information from worldometers.info Sep 2021). The mean age of the UWIST surveys was in Nakuru, 22.8 years for women, 24.6 years for men; Udon Thani, 24.1 year for women; 24.7 years for men).

Survey		Participant demographics (W=Women/M = Men)							Survey description
		Neighbourhood	Age						
			18–30	31–45	46–60	61–75	76+	Total	
Neighbourhood Wellbeing Survey	Nakuru	CBD Total:57	M:8 W:8	M: 13 W:10	M:7 W:4	M:5 W:1	M:1 W:0	M:34 W:23	Likert scale questions; Short Warwick Wellbeing; Perceived Stress; Use of green and public realm space
		Free Area Total:78	M: 13 W:11	M: 12 W:14	M:9 W:9	M:2 W:6	M:2 W:0	M:38 W:40	
		Kaptembwo Total:130	M: 19 W:20	M: 22 W:23	M: 14 W:16	M:9 W:4	M:2 W:1	M:66 W:64	
		London Total: 97	M: 16 W:19	M: 15 W:17	M:8 W:14	M: 9 W:10	M:4 W:1	M:52 W:45	
		Section No58 Total:100	M: 15 W:13	M: 16 W:13	M: 14 W:12	M:7 W:5	M:3 W:3	M:55 W:45	
		Shabab Total:50	M:7 W:7	M:7 W:10	M:7 W:5	M:3 W:1	M:1 W:2	M:25 W:25	
		TOTAL:528	M: 78 W:78	M: 85 W:87	M: 59 W:60	M: 35 W:26	M: 13 W:7	M: 270 W:258	
	Udon Thani	Baan Non Total:64	M: 7 W:10	M:9 W:9	M:6 W:12	M:8 W:6	M:0 W:0	M:27 W:37	
		Thong Yai Total:136	M: 16 W:8	M: 19 W:20	M: 27 W:22	M: 8 W:13	M:8 W:3	M:70 W:66	
		Baan Muang 1 Total:90	M:8 W:7	M: 11 W:14	M: 12 W:20	M: 7 W:10	M:1 W:0	M:39 W:51	
		Thongkham Uthit 2 Total:91	M:9 W:8	M: 19 W:16	M:13 W:9	M:7 W:7	M:1 W:2	M:49 W:42	
		Non Yang 2 Total:39	M:5 W:5	M:3 W:4	M:3 W:10	M:1 W:6	M:0 W:2	M:12 W:27	
		Nong Lek1&2 Total:80	M: 8 W:10	M:7 W:12	M: 10 W:26	M:5 W:1	M:0 W:1	M:30 W:50	
		Non Than Total:87	M:6 W:7	M:6 W:12	M: 18 W:14	M: 18 W:3	M:3 W:0	M:51 W36	
		TOTAL:587	M: 59 W:55	M: 74 W:87	M: 89 W:113	M: 51 W:46	M:5 W:8	M: 278 W:309	
Transect Walk		City	W	M	Total				UWIST Mood Adjective Checklist
		Nakuru	58	64	122				
		Udon Thani	58	57	115				

characteristics) related to subjective assessments of wellbeing (life satisfaction)?' All the significant statistical analysis presented in this results sections are in **table 2** below.

Inter-city Comparison

GDP per capita in 2019 (data.worldbank.org) varied from $7808 for Thailand to $1816 in Kenya indicating significant overall differences in average living standards and economic prosperity between the two countries. Our indicators of socio-economic conditions (relative affluence) confirmed these key differences for our two case study cities. Our Kenyan city has a statistically significant higher number of self-employed and tenants compared to Thailand where more residents were employees and homeowners. Access to basic services were reported as unproblematic across Udon whereas there were significant impacts from lack of access to infrastructure including water in Nakuru. The objective wellbeing scores represent a continuum from the most affluent neighbourhood in Nakuru having similar scores to the least affluent in Udon (see **Figure 2**).

Nakuru Economic and Socio-Environmental Conditions

Employment status (employed versus self-employed) determined which neighbourhood residents can afford to live in. There was no significant difference in objective wellbeing except between the extremes of the best serviced district (Shabab) and the least affluent (semi-informal Kaptembwo). Whilst this indicates similar infrastructure conditions across the majority of Nakuru neighborhoods analyzing by gender reveals significant differences in women's objective wellbeing scores (whilst men's do not vary significantly). Access to water, water quality and solid waste

TABLE 2 | Statistical analyses underpinning the results.

Esults section	Variables compared	Statistical result	
3.1.2 Nakuru economic and socio-environmental conditions	Chi2 test of Employment status and Neighbourhood	$\chi2$ (10) = 28.191, $p = 0.002$	
	One-way Anova comparison of Objective Wellbeing Score between Kaptembwo and Shabab	F (5, 506) = 3.282, $p = 0.006$	
	One-way Anova comparison of Women's Objective Wellbeing Score by Neighbourhood	F (5, 242) = 3.396, $p = 0.006$	
3.1.3 Udon Thani economic and socio-environmental conditions	Chi2 test of Tenancy status and Neighbourhood	$\chi2$ (6) = 26.810, $p < 0.001$	
	One-way Anova Objective Wellbeing Scores and Neighbourhood	F (6, 299) = 10.817, $p < 0.001$	
3.2.2 Udon Thani Subjective Wellbeing	One-way Anova comparison of SWEMWBS by Neighbourhood	F (6,271) = 2.16, $p = 0.047$	
	One-way Anova comparison of Older (61 + yrs) and younger people's Perceived Stress Scores	Difference in mean PSS of +1.6. F (3,583) = 5.59, $p = 0.01$	
	One-way Anova comparison of Older (61 + yrs) and younger people's SWEMWBS	Difference in mean SWEMWBS of -1.9. F (3,583) = 7.35, $p = 0.01$	
3.2.3 Inter-city comparison	One-way Anova comparison of PSS between Nakuru and Udon Thani	F (11,136) = 194.33, $p < 0.0005$	
	One-way Anova comparison of SWEMWBS between Nakuru and Udon Thani	F (11,136) = 1.039, $p < 0.308$	
3.3.1.1 Nakuru Greenspaces	Kruskal-Wallis test of difference in NDVI pixel values by neighbourhood	Pixel Range	Sig
		Values 0–9	0.086
		Values 10–19	0.000
		Values 20–29	0.000
		Values 30–39	0.000
		Values 40–49	0.000
		Values 50–59	0.000
		Values 60–69	0.000
		Values 70–79	0.000
		Values 80–89	0.001
	One-way Anova correlation between NDVI values and neighbourhood affluence	F (11,136) = 1.039, $p < 0.308$	
	Chi2 association between neighbourhood and living within walking distance of a greenspace	χ^2 (5) = 21.951, $p = 0.0005$	
	Chi2 association between neighbourhood and use of greenspace by surveyed residents	$\chi2$ (5) = 2.980, $p = 0.703$	
	One-way ANOVA comparison of change in SWEMWBS with more than 2 h s time spent in greenspace	F (1,290) = 4.677, $p = 0.031$	
	One-way ANOVA comparison of change in PSS with average greenness of neighbourhoods from NDVI pixel values	F (5,252) = 3.417, $p = 0.005$	
3.3.1.2 Nakuru Public Realm Spaces	Chi2 association between neighbourhood and public space walking distance accessibility	$\chi2$ (5) = 19.189, $p = 0.002$	
	Chi2 association between availability of walking distance public space and use	$\chi2$ (5) = 21.951, $p = 0.0005$	
3.3.1.3 Nakuru Environments Effects on Mood	*t*-test of change in men's hedonic tone pre- and post- transect walk for those who ended in the public park (pre-mean = 21.41; post-mean = 22.38)	(t (31) = −2.142, $p = 0.040$)	
3.3.1.4 Udon Thani Greenspaces	Kruskal-Wallis test of difference in NDVI pixel values by neighbourhood	Pixel Range	Sig
		Values 0–9	0.154
		Values 10–19	0.013
		Values 20–29	0.001
		Values 30–39	0.102
		Values 40–49	0.080
		Values 50–59	0.002
		Values 60–69	0.002
		Values 70–79	0.034
		Values 80–89	0.999
	Chi2 association between neighbourhood and living within walking distance of a greenspace	$\chi2$ (6) = 103.845, $p = 0.000$	
	Chi2 association between neighbourhood and use of greenspace by surveyed residents	$\chi2$ (6) = 37.056, $p = 0.000$	
3.3.1.5 Udon Thani Public Realm Spaces	Chi2 association between neighbourhood and public space walking distance accessibility	$\chi2$ (6) = 65.664, $p = 0.000$	
3.3.1.6 Udon Thani Environments Effects on Mood	*t*-test of change in all participants hedonic tone pre- and post- transect walk for those who ended in the public park (pre-mean = 23.74; post-mean = 22.32)	t (64) = 3.908, $p = 0.000$	

pollution were the most important differences in basic services and environmental conditions identified between the semi-informal neighborhoods and planned, more affluent locations.

Udon Thani Economic and Socio-Environmental Conditions

Tenancy status varied by neighbourhood indicating differences in home ownership levels across the city. Objective wellbeing varied significantly by neighbourhood. These results confirmed our sample neighborhoods had varying levels of affluence. Of the socio-environmental factors assessed, only traffic congestion was perceived to be having a 'somewhat negative' impact on wellbeing.

Subjective Wellbeing

Our subjective wellbeing metrics varied within our case study cities by neighbourhood and with gender.

Nakuru Subjective Wellbeing

Perceived stress tracks with affluence and objective wellbeing metrics and varied significantly between the most (Section 58) and least affluent (Kaptembwo) neighborhoods. This indicates that the absence of basic infrastructure and employment uncertainty has a significant psychological impact on daily life.

Our objective wellbeing data indicates that differences in the impacts from social conditions including the incidence of crime and anti-social behaviour between neighborhoods could be underlying factors affecting stress level variations. These take on a gendered dimension with significant differences in women's PSS between Kaptembwo (large to somewhat negative crime impacts (1.7); somewhat negative anti-social behaviour impacts (2.4) and Shabab (somewhat negative crime (2.26) and anti-social behaviour (2.46) impacts) (see **Figure 5**). Within neighborhoods, women's stress was significantly higher than men's in both the least affluent Kaptembwo but also the more affluent CBD (where crime and behaviour both affect women's wellbeing more strongly (see supplementary materials: two Assessment of socio-economic conditions) (see **Figures 3**, **4** below).

Wellbeing scores and stress were not significantly correlated with age.

Udon Thani Subjective Wellbeing

The SWEMWBS are lowest for the extreme's of high and low objective wellbeing neighborhoods indicating lower overall life satisfaction in these locations (see **Figure 5**) with the best and worst socio-economic conditions. The surveyed stress scores range from low-to-moderate stress levels and vary independently of affluence indicating other factors are affecting wellbeing beyond employment, tenancy and job type and does not show statistically significant variation by neighbourhood (see **Figure 6**). We identified that older people (61 + yrs) have lower wellbeing and higher stress levels. Additionally gendered differences emerged between men and women's stress levels in the affluent (as measured through objective wellbeing) Baan Muang neighbourhood.

Inter-city Comparison

Both cities perceived stress results can be characterized as 'moderate'. Perceived stress in Nakuru (mean score 18.25) was significantly higher than in Udon Thani (mean 14.24), however, subjective wellbeing scores were only marginally different. This highlights that even when urban conditions are a source of persistent stress, longer term personal life satisfaction can remain high.

Nakuru Urban Environments

The following sections results present findings relevant to our initial research question of "how is the relationship between subjective wellbeing mediated by the quality of urban environments?"

In our survey, two aspects of the quality of physical environments were considered; availability of greenspace (vegetated parks, sports grounds, temples and woods) and public realm spaces (town square, markets, shopping malls, sports and community centers) both of which can be used for recreation promoting both physical and mental health.

Nakuru Greenspaces

Our spatial analysis showed statistically significant differences in NDVI values between neighborhoods (see **Figure 7**). These were not correlated with affluence indicating some poorer neighborhoods had more greenspace than wealthier locations. Responses to the neighbourhood survey highlighted that availability of walking distance greenspace varied significantly. However, there was no significant relationship between availability and use. These findings indicate that availability of green infrastructure cannot infer usage or accessibility with other factors or preferences either enabling or inhibiting participants use of greenspaces. However, comparing the average greenness of neighborhoods (from the NDVI values) to the PS scores indicated significant stress level reductions. This indicated that more local greenery reduced stress regardless of usage of these environments for recreation.

For those participants who did utilize greenspace, spending greater than 2 h per week in natural environments led to significant improvements in subjective wellbeing (SWEMWBS) (from scores of 26.9 (±4.5) to 28.3 (±5.2)). This 2 h threshold links to recommended "doses" of greenspace use (White et al., 2019) found in other studies from the Global North. Spending longer quantities of time showed no greater improvements with the limited number of respondents exceeding 3 hours having no significant improvements in their subjective wellbeing and stress scores.

Nakuru Public Realm Spaces

The survey findings identified a weak but significant association between neighbourhood and walking distance access to public realm spaces. Shabab, the best planned neighbourhood, reported the greatest accessibility (with 76% of respondents reporting they lived within walking distance). The survey also indicated that increased availability of public realm space led to greater use by residents. These results highlight the unequal

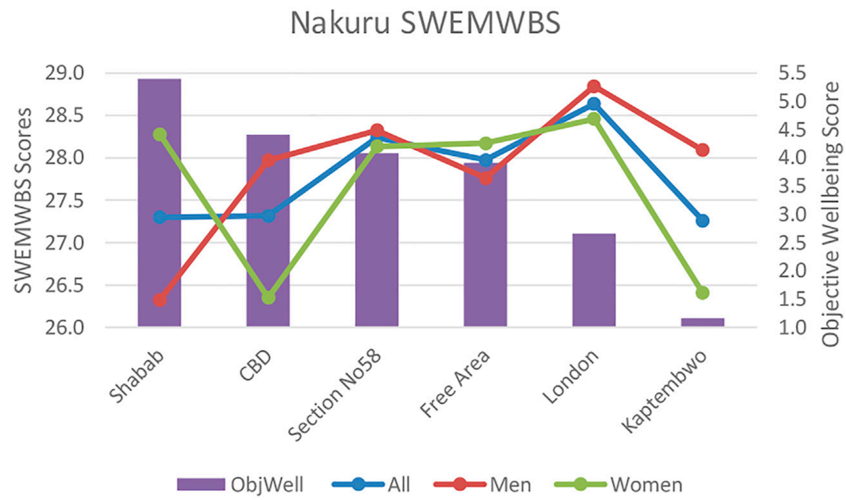

FIGURE 3 | Nakuru Short-Warwick subjective wellbeing scores (SWEMWBS) versus objective wellbeing scores by neighbourhood.

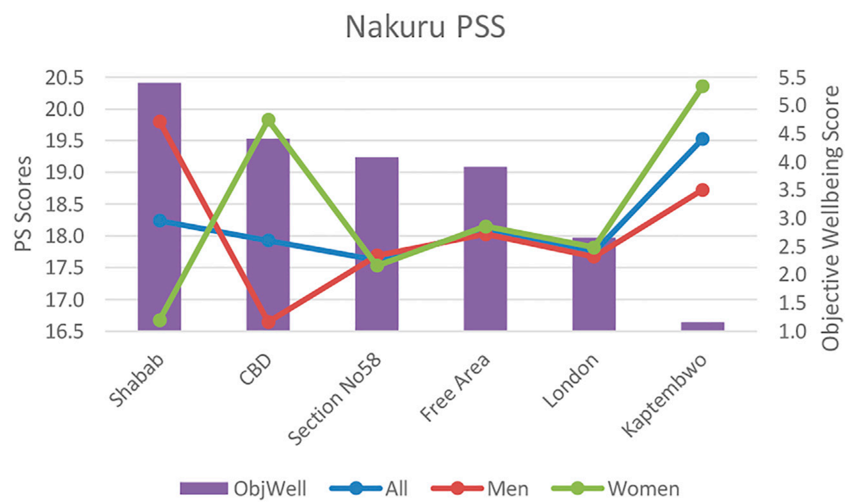

FIGURE 4 | Nakuru perceived stress scale (PSS) scores versus objective wellbeing scores by neighbourhood.

distribution of public realm assets leading to different opportunities for residents to access sociable community spaces.

Nakuru Environments Effects on Mood

The only significant effect of undertaking the transect walk was upon men's hedonic tone who ended their route in the park. Hedonic tone indicates feelings of happiness or sadness and this result suggests there maybe gendered effects to the benefits from public greenspaces.

Udon Thani Greenspaces

Our spatial analysis indicated differences between neighborhoods in terms of their extreme greenness (high NDVI pixel values (50–79) or extreme greyness (low values 10–29) (see **Figure 8**). These differences manifested in significant differences in residents perceptions of accessibility of walkable distance greenspace. However, the perception of access to greenspace did not always correlate with the measured differences in greenness (NDVI). For example, only 35.6% of Baan Muang one residents indicated that they lived within walking distance of greenspace despite relatively high levels of vegetation (mean NDVI value of 45.05 compared to the highest Non Than with 52.02). Low perceptions of walkable greenspace correlated with significant lower usage of greenspace for recreation.

Use of greenspace did not lead to any significant differences in subjective wellbeing measures (SWEMWBS and PSS). The majority (67.5%) of respondents were making some recreational use of greenspace indicating this behaviour was ubiquitous, however, 65.7% (n = 375) of respondents were spending less

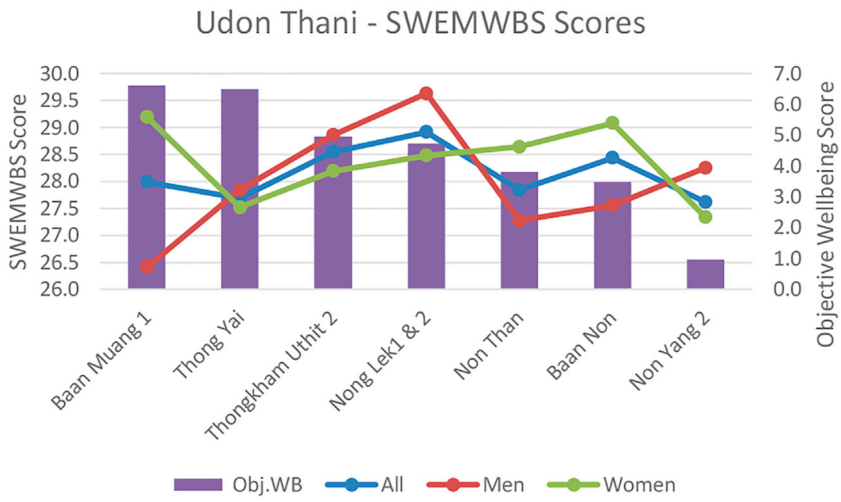

FIGURE 5 | Udon Thani Short-Warwick subjective wellbeing scores (SWEMWBS) versus objective wellbeing scores by neighbourhood.

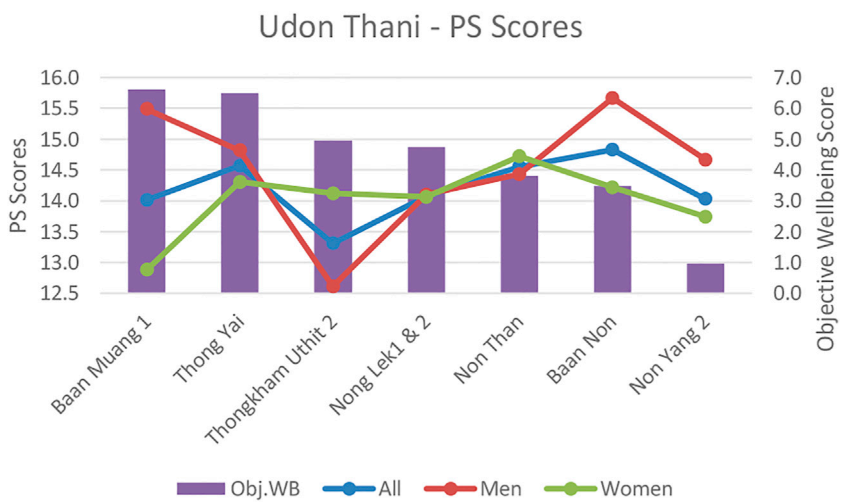

FIGURE 6 | Udon Thani perceived stress scores versus objective wellbeing scores by neighbourhood.

than the two 2-h per week threshold. For those who did spend time in greenspace there was no significant relationship or time related benefit on subjective wellbeing or stress scores.

Udon Thani Public Realm Spaces

There was a significant difference in the perceived accessibility of public realm spaces by neighbourhood. In general, those neighborhoods on the periphery had less access to public spaces than inner city locations. Approximately 70% of respondents are making use of public realm spaces for recreation, however, greater equality of provision could increase this usage. There was no significant impact on the use or length of time spent in public spaces for recreation on wellbeing or stress.

Udon Thani Environments Effects on Mood

Looking at the influence of route on the participants in the transect walk, those who began their walk in the park and ended in the market did not see a significant change in hedonic tone (happiness). However, participants who began their transect walk in the market and ended in the park saw a significant decrease (pre-mean = 23.74; post-mean = 22.32) in hedonic tone (indicating increased sadness). These decreases effected both men and women significantly. These results

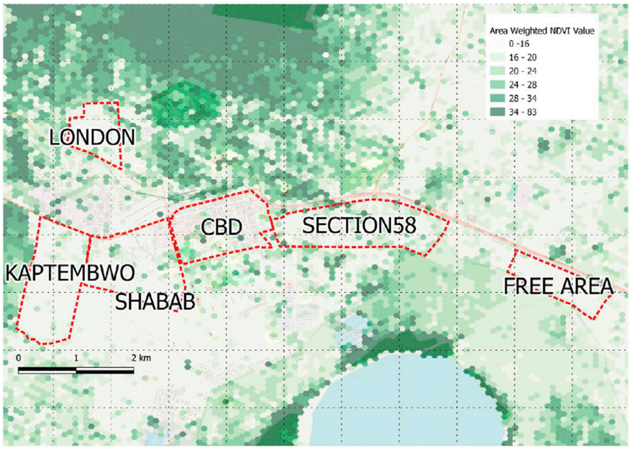

FIGURE 7 | Nakuru Mean NDVI Jan-Dec 2018 **(Landsat Imagery)** [**source:** http://climateengine.org/]. Values have been visualized as mean area weighted NDVI values by 100 m hex grid derived from original 29 m pixels.

contradict the findings from Nakuru where there were hedonic benefits attributed to greenspace for men.

DISCUSSION

Our results address how urban quality affects both objective and subjective wellbeing outcomes. The variation in our measured scores between neighborhoods across both cities confirms that our sample sites have a diversity of economic affluence allowing us to usefully compare how objective aspects of local environmental conditions interact with the subjective wellbeing of residents in LMIC settings.

How Are Objective Aspects of Wellbeing Are Related to Subjective Assessments of Life Satisfaction?

In Kenya, our data indicates how informal and poorly implemented infrastructure has resulted in unequal access to the provision of basic services. Further exploration of these results highlight that poor water access and quality, and solid waste pollution contribute to measurable differences in objective wellbeing impacts between planned and less affluent districts. However, across all neighborhood's people perceived the limited water access and crime incidence were undermining their wellbeing demonstrating that some challenges are ubiquitous.

In comparison, in Thailand, overall infrastructure and socio-economic conditions were largely un-problematic. However, despite the effects on objective wellbeing being marginal there were a greater number of differences between neighborhoods related to variations in air quality, noise pollution and traffic congestion. This indicates that as relative affluence increases, the significance of marginal inequalities between neighborhoods can become more pronounced. Our Udon Thani data also mirrors findings from other middle income countries (Colombia) (Scopelliti et al., 2016)that mid-affluent communitie's wellbeing can benefit most from infrastructure availability and environmental improvements (refer to **Figures 5**, **6**).

How Is the Relationship Between Subjective Wellbeing Mediated by the Quality of Urban Environments?

In Nakuru, subjective wellbeing predominantly lay in the 'good' range (scores of 26–28) (Ng Fat et al., 2017) whilst perceived stress fell within the moderate range (scores of 14–26) (State of New Hampshire Employee Assistance Program, 1983). In Udon, wellbeing was "good" to "excellent" (28+) but with between neighbourhoods differences becoming significant. Stress ranged from "moderate" to "low" but did not vary by neighbourhood indicating additional lifestyle factors beyond local environmental conditions were becoming influential on individual mental health. The absence of basic infrastructure (access to water; sanitation) causes significant stress alongside the obvious direct human-health effects (Ritchie and Roser, 2019). The granularity of our findings (at the neighbourhood scale) indicates that unequal access to basic services linked to affluence within LMIC cities significantly affects inequalities in resident's subjective wellbeing. This contradicts the findings of Kelley and Evans (Kelley and Evans, 2017) who concluded from national survey data that inequality in income distribution boosted wellbeing in low-income countries. Overall, our

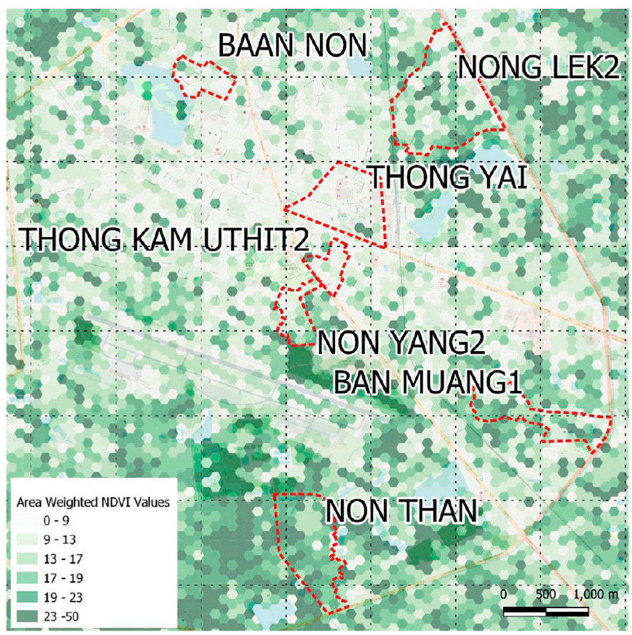

FIGURE 8 | Udon Thani Mean NDVI Jan-Dec 2018 **(Landsat Imagery)** [**source:** http://climateengine.org/] Values have been visualized as mean area weighted NDVI values by 100 m hex grid derived from original 29 m pixels.

findings show that city form and quality (both physical infrastructure provision and social interactions) in LMIC cities can have measurable impacts on the subjective wellbeing and stress of residents, potentially undermining their long-term mental health.

Significantly, our Kenya case results demonstrates that residents who make regular use of greenspace (greater than 2-h per week) show benefits to their subjective wellbeing independent of their neighborhoods conditions. This indicates the psychologically restorative benefits of greenspace can offset stress even for those living in informal settlements. In Thailand we did not find associations between wellbeing improvements and greenspace use. This could indicate that other factors influencing ability to spend time in greenspace (e.g. age, employment status) that also affect stress or wellbeing, are masking any benefits of time spent in natural surroundings. The Thai satellite data revealed, whilst there were variations in greenness between the city-centre and peri-urban fringe, most neighbourhoods had significant levels of vegetation. We hypothesize an alternative explanation for these findings that as urban vegetation is more equitably distributed across a city, routinely exposing people to nature, spending time specifically in greenspace has less discernable mental health benefits. The young cohort of Udon Thani transect walk participants indicated a subjective preference for a sociable retail space over a city park again indicating that perhaps greenspace may be less appreciated when it is widely available.

What Are the Implications for Urban Development to Achieve Equitable Wellbeing Improvements?

Our mixed-method approach highlights the complexity of these inter-relationship; however, they do identify a prioritization for urban planners when considering delivery of life satisfaction improvements. Our cross-city comparison highlights that delivering basic needs infrastructure or services universally must always be the primary city development priority. However, once these services are widely available urban form (distribution of public realm or greenspace) and management (socio-environmental conditions) require greater attention. Wellbeing effects associated with variations in these factors begin to take on a gendered and age dimension independent of neighbourhood affluence (employment and housing status). This conclusion is supported by other studies undertaken in higher income locations (Modai-Snir and van Ham 2018; Patel et al., 2017).

Our findings demonstrate that accessible public realm greenspace and neighbourhood greenery can offset some of the negative impacts on wellbeing of urban living even in challenging environments (socio-economic conditions) including informal settlements counteracting some income related health inequalities (Scopelliti et al., 2016). This supports findings on wellbeing impacts for low-income residents from park use in Indian and Colombian cities (Scopelliti et al., 2016; Ahirrao and Khan, 2021). When combined with the recognized physical health benefits (Siqueira Reis et al., 2013; Canterbury District Health Board, 2016; Adhikari et al., 2019), improved neighbourhood economic prosperity (Ahirrao and Khan, 2021) and co-benefits for active travel (Fluhrer et al., 2021) delivering these features more equitably across cities should be a key consideration for planners.

Our results also highlight that neighbourhood greening needs to be culturally appropriate and relevant for local communities including the urban poor (Ramaswami et al., 2016). This supports call for studies investigating distinct cultural and environmental conditions to make urban greenspace recommendations locally relevant in the Global South (Scopelliti et al., 2016) as the lived experience of residents from African, Asian or Latin American cities can vary distinctly due to factors including interactions of environment and infrastructure (Nagendra et al., 2018). For example, our Thai findings reveal local preferences for incorporating green infrastructure into retail and built public realm spaces to the maximize the distribution of wellbeing and ecosystem service benefits in this urban setting.

How do Our Findings Compare to Studies From Across Global South Cities?

Our results highlight that distributing greenery throughout cities enables a wider cross-section of residents to enjoy benefits to their underlying wellbeing without needing to spend dedicated time in specific parkland destinations (Cocks et al., 2016; Markevych et al., 2017). This implies cities should incorporate greenspaces through street trees, greened roadside verges, or small-scale pocket parks rather than prioritizing larger but scarcer public parks supporting the findings of (Siqueira Reis et al., 2013). This could begin to counteract the emerging crisis in the rise of non-communicable diseases linked to inactivity and stress identified across South Asia (Adhikari et al., 2019). We add support to the social and spatial justice arguments for widening the distribution of urban greening (Camargo et al., 2017; Rigolon et al., 2018; Zuniga-Teran and Gerlak, 2019; Ahirrao and Khan, 2021) by adding in quantitative evidence on the wellbeing and livability benefits such improvements could bring.

Such distributed greenspace would also ensure equity in other ecosystem service benefits such as urban cooling; shading; biodiversity increases; and surface water flood mitigation (Panagopoulos et al., 2015; Canterbury District Health Board, 2016). Unfortunately, urban greenspace is declining across Global South cities especially rapidly growing secondary cities (Nero, 2017; Adhikari et al., 2019; Fluhrer et al., 2021). As highlighted by (Bai et al., 2018) city planners need greater access to neighbourhood scale data to truly understand the distributional impacts of urban form on resident's health and city function. For example, internally displaced people residing in Nakuru county have been shown to have poor mental health, quality of life and life satisfaction (Getanda et al., 2015) contrasting with our Nakuru city participants who reported good overall life satisfaction and moderate stress. This demonstrates how high-resolution data is required to identify issues for specific places or population groups understanding local preferences to ensure city developments are appropriate and not merely transferred from different contexts (Nagendra, 2018; Cocks and Shackleton, 2021).

Cross-cutting development issues by their complex nature benefit from an integrated, multi-sector, consultative approach to problem solving if identified solutions are to be resilient (Mitra et al., 2017) in the context of diverse and dynamic city environments. New configurations of actors and collaborations are needed that include vulnerable groups and those typically excluded from city planning (Cinderby et al., 2021; Shackleton et al., 2021). This ambition to make improvements locally relevant (Patel et al., 2017) requires city authorities to plan using participatory co-design approaches that harnesses the collective creativity of people working together in a development process (Lam et al., 2017). Such approaches enable the development of improved shared understandings of complex problems allowing diverse stakeholder to collaborate and agree on locally relevant solutions (McArthur and Robin, 2019). This consensus building aids decision makers identification and delivery of more effective actions (Adelina et al., 2020). These approaches are particularly pertinent when addressing greenspace justice as public institutions typically fund these assets meaning all citizens should enjoy their benefits related to delivery of SDG 11.7 s (Daniel, 2014) ambition to "provide universal access to safe, inclusive and accessible, green and public spaces."

Limitations of Findings

The cross-sectional survey data used in this analysis represents a snapshot of conditions at a particular moment. Cities and communities are dynamic - investigating wellbeing's relationship to changing urban environments would therefore benefit from a long-term longitudinal approach, similar to cohort studies from health sciences. Including a wider range of quantitative data with which to compare subjective wellbeing results and environmental perceptions would also provide a more robust picture of the relationship between people and cities. This could include measuring environmental factors known to affect wellbeing such as air and noise pollution, temperature, and humidity, but also quantitative recording of locally relevant socio-economic conditions such as crime or fluctuations in employment levels. Also improving our understanding in a more nuanced way of the interactions of people and places beyond home neighborhoods would explore temporal and seasonal dimensions. Incorporating more qualitative data from participants would add significant richness and additional context to the findings. Results from a complementary survey undertaken by the paper authors in both case study cities on the cultural ecosystem services that different urban spaces provide addresses this shortfall to a certain extent (Cinderby et al., 2021). We also recognize that focusing on greenness in our analysis lacks inclusivity of other spaces that may be valued within different cultural contexts. We would advocate for a wider definition of beneficial urban infrastructure to include natural (brown-, green-, blue-, and barren spaces) alongside built PRS (indoor and outdoor spaces), and their combinations when looking at the interactions of urban form and wellbeing in the Global South. Finally, this study was only undertaken in two cities; collecting similar data from a wider range of locations would significantly improve the robustness and transferability of our findings allowing a generic set of recommendations for a healthy, liveable city to be identified.

CONCLUSION

This study contributes to filling data gaps from LMIC secondary cities on the impacts of urban living on resident's wellbeing. Our data highlights that delivering basic services to all neighborhoods should be the initial priority. Once these amenities are provided inequalities in the availability of other infrastructure and socio-cultural conditions begin to impact life satisfaction and stress. Our findings indicate that enabling residents to spend 2 hours per week in greenspace may generate similar wellbeing benefits to those identified in European studies. Improving equitable access across cities by dispersing green infrastructure should therefore be a key target for urban planners. Our Thai findings indicate that accessible greened spaces that support social interactions should be the preferred model for implementing these recommendations to support wellbeing for the widest cross-section of city residents.

Rapidly changing cities need to take greater account of the impacts urban form have upon human health and wellbeing. Ensuring equitable access to greenspace entails city authorities prioritize maintaining existing green infrastructure whilst protecting locations that will enable the inclusion of public realm spaces as the urban area expand. Adding improved neighbourhood scale data on human health and wellbeing benefits to the understanding of other ecosystem services provided by urban nature could justify such protection. Our findings indicate that such evidence could counterbalance significant densification pressures driven by cities ambitions to improve efficiency through the conversion of natural spaces into conventional economic assets. Expanding nature provision as cities evolve rather than expensively retrofitting greenspace into built infrastructure is a more cost-effective strategy for LMICs. Further evidence is needed of the financial costs of poor mental health or the economic gains resulting from access to green infrastructure from a wider cross-section of LMIC cities to strengthen these recommendations ensuring they become a key development issue and priority for urban authorities.

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 Department of Environment and Geography at the University of York, United Kingdom. The patients/participants provided their written informed consent to participate in this study.

AUTHOR CONTRIBUTIONS

SC - Conceptualization, Methodology, Formal analysis, Writing - Original draft preparation; DA–Data Collection, Methodology, Writing - Original draft preparation; VM - Conceptualization, Writing - Original draft preparation; CN - Conceptualization, Methodology, Formal analysis, Writing - Original draft preparation; RO - Data Collection, Methodology; RP - Data Collection, Methodology; CM - preparation; RO - Data Collection, Methodology; RP - Data Collection, Methodology; CA - Data Collection, Methodology; HT - Conceptualization, Methodology.

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The Impact of Haze on Healthcare Utilizations for Acute Respiratory Diseases: Evidence From Malaysia

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Haze imposes a substantial disease burden on the human population especially in the Southeast Asia region due to the high frequency of haze episodes. The reduction of air quality levels by haze has resulted in a substantial disease burden and an increase in healthcare utilization (HU). This study aims to determine the association between haze and HU of haze-related respiratory illnesses with a focus given on the acute exacerbation of bronchial asthma (AEBA) and chronic obstructive pulmonary disease (AECOPD). A cross-sectional study was conducted through secondary data collection of haze/non-haze episodes as the study exposures and HU related to the exacerbation of bronchial asthma and COPD as the study outcomes. Data on haze/non-haze episodes and HU for four consecutive years (2012–2015) were retrieved from the Department of Environment and Ministry of Health Malaysia, respectively. In the four consecutive years, the percentage of haze episodes recorded in all stations was higher (67%) as compared to non-haze (33%) episodes. Means (SD) of patients diagnosed with AEBA and AECOPD were also significantly higher ($p < 0.05$) for inpatient 74 (62.1) and outpatient 320 (650.1) cases during haze episodes as compared to inpatient 34 (16.5) and outpatient 146 (170.5) cases during non-haze episodes. Findings from this study indicated that haze episodes incurred a significant healthcare burden due to an increase in HU. The evidence from this study will help the policymakers to prepare and allocate resources to control future implications of haze-related illnesses.

Keywords: haze, air pollution, healthcare, utilization, respiratory

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INTRODUCTION

Haze is a condition associated with the disruption of visibility, clarity and transparency in an area due to the presence of fine suspended particles (Cheng et al., 2013; Othman et al., 2014). Specifically, it is defined as "an aggregation in the atmosphere of very fine, widely dispersed, solid, or liquid particles, or both, giving the air an opalescent appearance that subdues colors" (Hyslop, 2009, p. 182). Over the recent decades, Malaysia's transformation from an agricultural-based to a more industrial-based country has led to rapid urbanization and an increase in manufacturing and industrial processing activities (Awang et al., 2000; Abdullah et al., 2012). Migration to urban areas such as Klang Valley has resulted in a dramatic increase in population density and traffic congestion. Exhaust emissions from motor vehicles and pollutants from industrial activities, in that order, are the main domestic sources of haze in Malaysia (Afroz et al., 2003; Abdullah et al., 2012; Mohd Shahwahid, 2016). Transboundary air pollutants from neighboring countries also contribute significantly to the occurrence of haze episodes in Malaysia. Along with other countries in the Southeast Asian region, Malaysia has been affected several times by haze episodes due to open forest burning in Indonesia (Othman et al., 2014; Mohd Shahwahid, 2016).

Haze poses serious and recurring medical problems, especially among susceptible individuals such as those suffering from chronic diseases, children, the elderly and pregnant women. The burden of illness is directly proportionate to the intensity of haze due to higher healthcare utilization and related healthcare costs during the haze episode. The health impacts of air pollution are more pronounced on the respiratory system than on the cardiovascular system (CVS). Healthcare utilization related to CVS illnesses mainly involves inpatient rather than outpatient cases, as most of these conditions require patients to be admitted for further monitoring and assessment. In contrast, exposure to air pollution has an immediate impact on the respiratory system, which makes it easier for researchers to attribute the episode to air pollution (Adar et al., 2014; Sahani et al., 2014). The main reasons for respiratory-related outpatient visits and hospital admissions are acute exacerbation of bronchial asthma (AEBA), acute exacerbation of chronic obstructive pulmonary disease (AECOPD), acute bronchitis, pneumonia and bronchiolitis (in infants), with exacerbation of asthma and chronic obstructive pulmonary disease accounting for the majority of cases (Peacock et al., 2011; Anderson et al., 2012; Mehta et al., 2013; Laumbach and Kipen, 2014). Health complications due to haze-related illnesses are significantly associated with increased healthcare utilization and reduced productivity due to work absenteeism. Both transboundary and local sources that result in haze episodes have been found to play a significant role in the increased use of healthcare facilities (Brauer and Jamal, 1998; Othman et al., 2014) and, hence, increased healthcare cost and expenditure. Together with the loss of productivity due to complications from haze-related illnesses, the health impact of haze episodes produces a significant financial burden for both healthcare providers and patients (Kochi et al., 2010; Othman et al., 2014). The objective of this study is to determine the trends in haze incidence and healthcare utilization in Malaysia so that steps can be taken to combat haze and allocate appropriate resources to meet the healthcare demands associated with haze episodes.

MATERIALS AND METHODS

Study Background

The study was conducted for 4 years. Data for the period 2012–2015 were collected from the Department of Environment (DOE), the Ministry of Health (MOH), and University Malaya Medical Center (UMMC) and represented the latest data available from these bodies. A period of 4 years was considered sufficient to ensure adequate variation in the data distribution. This study was conducted in Selangor, which is one of the most developed state but with the worst air quality level compared to other states in Malaysia. The total population of Selangor in the year 2019 was 6.48 million people with a total land area of 7,930 km^2 (674 persons per km^2). Selangor has nine districts, each of which has its own health district office and at least one public hospital. The state has 12 public hospitals and 57 health clinics.

Study Population and Sampling Method

A universal sampling method was used in this study. First, districts with a Continuous Air Quality Monitoring (CAQM) station in Selangor were identified. These four districts are Petaling, Klang, Kuala Selangor, and Kuala Langat (**Figure 1**). All four districts were then included in the study. Districts without CAQM stations were excluded since it was difficult to identify a proxy measure of exposure levels to air pollutants. Data on the level of air pollutants and ecological factors recorded by the respective CAQM stations in each district were retrieved from DOE. To ensure the information on healthcare utilization corresponded to the data on the level of air pollutants and ecological factors in the identified area, patients who were diagnosed with AEBA or AECOPD and were admitted to public hospitals or visited emergency departments or health clinics in Petaling, Klang, Kuala Selangor, and Kuala Langat districts were identified and included as the study population. Their information was collected from cumulative monthly record data for every included hospital and health district office.

Exposure and Outcome

In this study, the definition of haze episode was based on World Health Organization (WHO) Air Quality Guideline (AQG), mean readings with 24 h PM_{10} value of $\geq$ 51 $\mu g/m^3$ were considered as a haze episode and readings with 24 h PM_{10} value of $\leq$ 50 $\mu g/m^3$ were considered as a non-haze episode (Krzyzanowski and Cohen, 2008). Haze and non-haze episodes were used as the exposure variable while the outcome was the utilization of healthcare facilities (treatment episodes of admissions and number of visits to the emergency department of selected public hospitals and health clinics) by patients diagnosed with AEBA and AECOPD during the defined events. CO, O_3, NO_2, SO_2, temperature, wind speed, and humidity were treated as the covariates, based on previous literature (Dominick et al., 2012; Othman et al., 2014).

Data Collection

All data used in this study were from secondary sources. The process of data collection had two components: a collection of data on air pollutants and ecological factors; and collection of data on admissions (inpatients) and visits to public healthcare facilities (outpatients) for patients diagnosed with AEBA or AECOPD. Data on air pollutants and ecological factors were taken from CAQM stations in Selangor. Each station monitors five air pollutants and three ecological parameters, namely, PM_{10}, CO, O_3, NO_2, SO_2, ambient temperature, wind speed, and relative humidity. These data were collected since they are the two major components of haze, which are the particles and

FIGURE 1 | Map of Selangor and its districts ("Selangor state," 2013).

gaseous pollutants. Particles are formed by suspended liquid or solid elements and are commonly referred to as particulate matter (PM) (Hyslop, 2009). The other component, gaseous pollutants are made up of carbon monoxide (CO), sulfur dioxide (SO_2), nitrogen dioxide (NO_2), and ozone (O_3). NO_2 is one of the compounds of the family of nitrogen oxides (NOx), along with nitric oxide (NO), nitrogen trioxide, nitrogen tetroxide, and di-nitrogen pentoxide. Fossil fuel combustion results in the formation of atmospheric nitrogen (N_2). The oxidation process of atmospheric N_2 produces NO and subsequently NO_2. CO is produced through incomplete combustion of carbonic materials and is a good indicator of the presence of pollution associated with combustion. SO_2 can combine with water to produce sulfurous acid, which causes considerable irritation, especially in the eyes, skin and mucous membrane. The main source of SO_2 is the combustion and roasting of sulfur-containing materials such as metal sulfide ores. Finally, O_3 is formed in the troposphere through the reaction between reactive hydrocarbon (from combustion), UV light and NO_2 (Brook et al., 2004). Data on the ambient temperature, wind speed, and relative humidity levels were also collected since these factors might also have a significant influence on the air pollutants level.

Except for PM_{10} (where the unit of measurement is in $\mu g/m^3$), parts per million (ppm) is used as the unit of measurement for other air pollutants. For the ecological parameters, the unit of measurement is in Celsius (C) for temperature, kilometers per hour (km/h) for wind speed and percentage (%) for humidity (Rahman et al., 2015). These parameters were collected in the form of daily data. It was then converted into monthly data since there was a possibility of lag effect (2–4 weeks) from the time of exposure with the onset of respiratory symptoms. This was done to match and determine the association of air pollution with healthcare utilization. Details on the trend and factors associated with haze were presented in another manuscript (Jaafar, 2019).

For healthcare utilization, the data were retrieved from the health information center (HIC) under MOH and the UMMC records unit. As described earlier, all public healthcare facilities located in districts with a CAQM station (Petaling, Klang, Kuala Selangor, and Banting) in Selangor were included. Data from the CAQM stations were matched with healthcare utilization data for the public healthcare facilities located in their respective districts. The public healthcare utilization data needed were the numbers of inpatient and outpatient cases diagnosed with AEBA and AECOPD at the respective healthcare facilities. The diagnosis used was based on the discharge diagnosis and followed the WHO International Classification of Diseases (ICD 10). These are the standard codes used by all healthcare facilities under MOH Malaysia and were also used in the HIC recording system. The ICD code for asthma is ICD-10:J45-J46 and ICD-10:J44 for COPD (WHO, 2016).

Data Analysis

Descriptive analyses were done to determine healthcare utilization in public healthcare facilities during haze and non-haze episodes. Continuous variables were presented as mean and standard deviation (median and inter-quartile range for data that were not normally distributed). These include monthly data for CO, O_3, NO_2, SO_2, temperature, wind speed, and humidity.

For the categorical variable (haze episode), the data were presented as the frequency with percentage. Healthcare utilization that was in the form of discrete data (also known as a Poisson distribution) was also presented as frequency and percentage.

Bivariate analyses were done to determine any association between haze episode (as the exposure) and healthcare utilization (as the outcome). Associations between exposure and covariates were also included in the analyses to identify any interaction between those variables. For bivariate analysis, the level of significance was pre-set at 0.05. For bivariate analyses, Independent T-test (Mann-Whitney for non-normally distributed data) was used to identify a significant association between exposure (haze episodes) and outcomes (healthcare utilization). For the covariates (CO, O_3, NO_2, SO_2, temperature, wind speed, and humidity), since all the data were continuous variables, Pearson's (Spearman's for non-normally distributed data) correlation test was used instead. Associations between exposure and covariates were also analyzed using Independent T-test (Mann-Whitney for non-normally distributed data) to identify any interaction that needed to be controlled in multivariate analysis. Results from bivariate analysis were presented with frequency, percentage and mean difference with p-value and 95% confidence interval (for Independent T-test/Mann-Whitney test) and correlation coefficient with p-value and 95% confidence interval (for Pearson's/Spearman correlation test). All variables with significant results in bivariate analysis were further analyzed in multivariate analyses. Multivariate analysis was done using Poisson regression under generalized linear models. Poisson regression was chosen since the outcomes variables were in the form of discrete data. Variables with significance value ($p < 0.25$) in the univariate regression analysis were included in multivariate analysis. Results for multivariate analysis were presented with p-value, 95% confidence interval and odds ratio. Variables with significant values ($p < 0.05$) were included in the final regression model.

RESULTS

Healthcare Utilization of Haze-Related Respiratory Illnesses

Overall, 192 monthly readings were collected for all air pollutants and ecological factors from four CAQM stations (located in Petaling, Klang, Kuala Selangor, and Banting). These data were matched with monthly healthcare utilization data from public hospitals and health clinics located in the same district from 2012 to 2015 (12 readings for monthly data in a year $\times$ 4 districts $\times$ 4 years). PM_{10} level was used as a reference to define a haze episode based on the World Health Organization Air Quality Guidelines (WHO AQG) for PM_{10}. PM_{10} levels of $\geq 51\ \mu g/m^3$ were categorized as a haze episode and PM_{10} levels of $\leq 50\ \mu g/m^3$ were categorized as a non-haze episode. Of the 192 readings, the highest daily PM_{10} level recorded was 595.1 in July 2013. 129 (67.19%) were classified as haze episodes and 63 (32.81%), as non-haze episodes. The total utilization of public healthcare facilities (for inpatient and outpatient cases of AEBA and AECOPD) in the studied districts was 62,051. Of these, 11,670 (18.81%) were inpatient cases and 50,381 (81.19%) were outpatient cases. The majority of inpatient cases were diagnosed with AEBA (n = 7,894, 12.72%), compared to 3,776 (6.09%) for AECOPD. The number of AEBA outpatient cases (n = 47,897, 77.19%) was also higher compared to AECOPD (n = 2,484, 4.00%).

Data from the MOH did not contain information on the age, gender and ethnicity of patients who sought treatment in its facilities. Therefore, a sub-analysis for socio-demographic characteristics was conducted using healthcare utilization data collected at UMMC. The total numbers of inpatient and outpatient cases that received treatment for AEBA and AECOPD in UMMC from 2012 to 2015 were 1,068 and 3,124, respectively. More than half of these patients were male and approximately two-thirds of them were Malay, followed by Indian and Chinese. Details of healthcare utilization rates and socio-demographic characteristics are summarized in **Tables 1**, **2**.

Association Between Healthcare Utilization and Haze Episodes

For AEBA, healthcare utilization for both inpatient and outpatient cases was higher during haze episodes. The average monthly admission for AEBA was 50 cases during a haze episode compared to 23 cases during non-haze periods. Similarly, the average monthly numbers of AEBA outpatient cases were 307 and 132 during haze and non-haze episodes, respectively. AECOPD also recorded a higher average monthly admission of 24 cases during haze compared to 11 during non-haze episodes. A different pattern, however, was observed for AECOPD outpatient cases, which recorded a higher average monthly visit rate during non-haze periods (14 cases) compared to that during haze episodes (13 cases). For total inpatient cases (for both AEBA and AECOPD), the average number of monthly admissions was

TABLE 1 | Healthcare utilization for AEBA and AECOPD in public healthcare facilities for all studied districts 2012–2015.

Outcome	*N* (%)
Total inpatient	**11,670 (18.81)**
(a) Inpatient AEBA	7,894 (12.72)
(b) Inpatient AECOPD	3,776 (6.09)
Total outpatient	**50,381 (81.19)**
(a) Outpatient AEBA	47,897 (77.19)
(b) Outpatient AECOPD	2,484 (4.00)
Total all cases	**62,051 (100)**

TABLE 2 | Number of haze episodes and socio-demographic characteristics of AEBA and AECOPD cases 2012–2015 (from UMMC data).

Variables	*N* (%)
Haze	129 (67.19)
Non-haze	63 (32.81)
Gender*	
Male	2,694 (64.27)
Female	1,498 (35.73)
Ethnicity*	
Malay	2,618 (62.45)
Chinese	581 (13.86)
Indian	845 (20.16)
Others	148 (3.53)

**Subanalysis using UMMC data.*

74 compared to 34 during non-haze episodes. Average monthly outpatient cases were also higher during haze episodes (320 cases) compared to non-haze episodes (146 cases). The difference in healthcare utilization between haze and non-haze episodes was statistically significant (p-value < 0.05) for all types of healthcare utilization except AECOPD outpatient cases. Results for the association between healthcare utilization and haze episodes are summarized in **Table 3**.

Association Between Healthcare Utilization and Other Pollutants and Ecological Factors

To identify possible confounders among the covariates (other air pollutants and ecological factors), a correlation test was conducted to determine whether there was any significant association between the covariates and healthcare utilization. Inpatient AEBA cases were not significantly associated with any of the air pollutants or ecological factors. For the inpatient AECOPD cases, significant negative correlations were observed with CO ($r = -0.24$, p-value = 0.01) and temperature ($r = -0.20$, p-value = 0.02). Outpatient AEBA cases were significantly correlated with all parameters except for SO_2 ($r = 0.05$, p-value = 0.65) and NO_2 ($r = 0.19$, p-value = 0.06). For outpatient AECOPD cases, significant correlations were also observed with CO ($r = 0.30$, p-value < 0.001), NO_2 ($r = 0.26$, p-value = 0.02), and humidity ($r = -0.30$, p-value < 0.001). For total inpatient cases, significant correlations were recorded with CO ($r = -0.22$, p-value = 0.02), NO_2 ($r = -0.25$, p-value = 0.03), and temperature ($r = -0.19$, p-value = 0.03). All parameters were significantly correlated with total outpatient cases except for SO_2 ($r = 0.05$, p-value = 0.66) and NO_2 ($r = 0.20$, p-value = 0.06). Although some of the air pollutants and ecological parameters were significantly associated with healthcare utilization (p-value < 0.05), none of the correlation coefficient (r) values was more than 0.5, indicating that there was no strong correlation between these variables. Results for correlations between healthcare utilization and other pollutants and ecological factors are summarized in **Table 4**.

TABLE 3 | Association between healthcare utilization and haze episodes.

Healthcare utilization	Episode		*N* (Mean)	*P*-Value
AEBA inpatient	Non-haze	63	1,431 (23)	<0.001
	Haze	129	6,463 (50)	
AECOPD inpatient	Non-haze	63	701 (11)	<0.001
	Haze	129	3,075 (24)	
AEBA outpatient	Non-haze	63	8,296 (132)	0.014
	Haze	129	39,601 (307)	
AECOPD outpatient	Non-haze	63	872 (14)	0.932
	Haze	129	1,612 (13)	
Total inpatient	Non-haze	63	2,132 (34)	<0.001
	Haze	129	9,538 (74)	
Total outpatient	Non-haze	63	9,168 (146)	0.013
	Haze	129	41,213 (320)	
Total all patients	**Non-haze**	**63**	**11,300(179)**	**0.001**
	Haze	**129**	**50,751 (393)**	

Association Between Healthcare Utilization and Sociodemographic Characteristics

A sub-analysis was conducted using UMMC data to determine the association between healthcare utilization and age, gender and ethnicity during haze and non-haze episodes. Overall, there were no significant gender differences in total inpatient [X^2 (1, N = 1,068) = 0.335, p-value = 0.56] and total outpatient [X^2 (1, N = 3,124) = 0.113, p-value = 0.74] cases during haze and non-haze episodes. Stratification based on ethnicity also revealed no significant difference in total inpatient [X^2 (1, N = 1,068) = 6.078, p-value = 0.108] and total outpatient [X^2 (1, N = 3,119) = 1.33, p-value = 0.72] cases during haze and non-haze episodes. There were also no significant age differences for both total inpatient [t (1,066) = 0.743, p-value = 0.17] and outpatient [t (3,122) = −0.489, p-value = 0.67] cases during haze and non-haze episodes.

Multivariate Analysis: Healthcare Utilization and Haze-Related Respiratory Illnesses

Univariate regression analyses were performed for each of the variables included to determine the crude odds ratio (OR). For inpatient cases, only haze episode (OR = 2.19, 95% CI: 1.611–2.964, p-value < 0.001) and NO_2 (OR = 0.02, 95% CI: 0.992–0.999, p-value = 0.02) showed a significant association with the outcome. Both CO (OR = 0.96, 95% CI: 0.709–1.288, p-value = 0.79) and temperature (OR = 0.93, 95% CI: 0.817–1.057, p-value = 0.29) had a p-value of > 0.25 and thus were not considered in multivariate analysis. For outpatient cases, all variables had p-value of < 0.25. The highest crude OR was recorded by haze episode (OR = 2.20, 95% CI: 1.628–2.980, p-value < 0.001), followed by CO (OR = 1.97, 95% CI: 1.377–2.807, p-value < 0.001), wind speed (OR = 1.28, 95% CI: 1.184–1.386, p-value < 0.001), and temperature (OR = 1.14, 95% CI: 1.004–1.284, p-value = 0.05). The lowest OR was recorded by humidity (OR = 0.94, 95% CI: 0.924–0.960, p-value < 0.001), followed by O_3 (OR = 0.96, 95% CI: 0.946–0.968, p-value < 0.001). All variables were then considered and included as in multivariate analysis.

Multivariate analyses were then conducted for both outcomes. For inpatient cases, both haze episode and NO_2 recorded a significant p-value of < 0.01. The goodness of fit test showed a deviance/df ratio of 0.523 and Pearson chi-square/df ratio of 0.526. Since both tests showed a ratio of less than 2, the final regression model was considered to have a good model fit. For outpatient cases, initial multivariate analysis showed significant results (p-value < 0.05) for haze episodes and O_3 only. The analysis was then rerun with the exclusion of non-significant variables. All variables included in the subsequent analysis were noted to have a p-value of < 0.001. Goodness of fit test showed a deviance/df ratio of 1.086 and Pearson chi-square/df ratio of 1.352. For inpatient cases, since both tests showed a ratio of

TABLE 4 | Correlations between healthcare utilization and other pollutants and ecological factors.

Healthcare utilization		O_3	CO	SO_2	NO_2	Wind speed	Temperature	Humidity
AEBA inpatient	*r*	0.02	−0.17	−0.03	−0.22	0.17	−0.13	−0.13
	P-value	0.80	0.07	0.71	0.06	0.07	0.11	0.09
AECOPD inpatient	*r*	0.13	−0.24	−0.04	−0.20	0.17	−0.20	−0.12
	P-value	0.09	0.01	0.63	0.07	0.17	0.02	0.11
AEBA outpatient	*r*	−0.43	0.36	0.05	0.19	0.31	0.23	−0.43
	P-value	<0.001	<0.001	0.65	0.06	0.01	0.01	<0.001
AECOPD outpatient	*r*	−0.14	0.30	−0.06	0.26	−0.07	0.14	−0.30
	P-value	0.07	<0.001	0.44	0.02	0.48	0.18	<0.001
Total inpatient	*r*	0.07	−0.22	−0.04	−0.25	0.17	−0.19	−0.11
	P-value	0.34	0.02	0.64	0.03	0.12	0.03	0.13
Total outpatient	*r*	−0.43	0.37	0.05	0.20	0.31	0.23	−0.44
	P-value	<0.001	<0.001	0.66	0.06	0.01	0.01	<0.001

less than 2, the final regression model was considered to have a good model fit. Based on the final regression model, the odds for both hospital admission and outpatient visit were 2.2 times higher during haze compared to non-haze episodes. Results of multivariate analysis are summarized in **Tables 5**, **6**.

DISCUSSION

Except in 2014, PM_{10} concentration reached its maximum level around May to September each year. This pattern was observed in all CAQM stations (**Figure 2**). The maximum level recorded during these periods coincided with biomass burning from forest and peat fires (as a result of agricultural land clearing in Indonesia) (Azmi et al., 2010; Abdullah et al., 2012; Norela et al., 2013) It was aggravated by the south-west monsoon season (June–September) (Azmi et al., 2010; Rahman et al., 2015). The southwest monsoon wind from Sumatra took only 48 h to reach all CAQM stations in Selangor (Azmi et al., 2010). The wind carried the particles and gaseous pollutants from the affected area to the western part of the Malaysian Peninsular, particularly Selangor. In addition, unlike the northeast monsoon, the southwest monsoon is commonly associated with the "dry season" (Azmi et al., 2010). PM_{10} is usually "washed out" of the atmosphere through a chemical reaction with water vapor (Norela et al., 2013). Low humidity levels together with less rainfall during this period resulted in an accumulation of PM_{10} in the atmosphere for a longer duration and caused deterioration in air quality (Azmi et al., 2010; Juneng et al., 2011; Norela et al., 2013; Rahman et al., 2015).

TABLE 5 | Crude and adjusted odds ratios in univariate and multivariate regression analyses (inpatient).

	Univariate		Multivariate	
Variable	**Crude OR (95% CI)**	***P*-value**	**Adjusted OR (95%CI)**	***P*-value**
Haze episode	2.19 (1.611–2.964)	<0.001	2.155 (1.587–2.926)	<0.001
CO	0.96 (0.709–1.288)	0.79		
NO_2	0.10 (0.992–0.999)	0.02	0.995 (0.991–0.999)	0.014
Temperature	0.93 (0.817–1.057)	0.29		

Overall results from this study showed that healthcare utilization for haze-related respiratory illnesses was 4.5 times higher during haze compared to non-haze episodes. These results were based on the total and the average number of hospital admissions and outpatient visits for both AEBA and AECOPD. The peak utilization of healthcare services was observed during the haze episode especially during the "dry season" where PM_{10} concentrations were consistently high almost every day and reached their maximum level. For total inpatient cases, the number of hospital admissions recorded was 4.5 times higher during haze episodes. Similarly, the number of outpatient visits for AEBA and AECOPD was also 4.5 times higher during haze episodes. Results from the multivariate analysis showed that the odds for both hospital admissions and outpatient visits were 2.2 times higher during haze episodes.

Although the effect sizes varied, the results from this study were consistent with findings from other literature and within the expected outcome. A local study showed significantly higher weekly mean (SD) hospital admissions during haze, with 27 (9.2) cases, compared to non-haze episodes with 15.7 (6.7) cases (Ming et al., 2018). Similar to our findings, the odds for hospital admission were 1.95 and 2.13 times higher during haze episodes for AEBA and AECOPD, respectively. A higher number of intensive care admissions and longer hospital stays were also recorded during haze episodes (Ming et al., 2018). The results from this study were also supported by another local study which showed a 31% increase in hospital admissions during haze episodes (Othman et al., 2014). Similarly, a systematic review of the health impacts of haze in Southeast Asian countries showed an increase in respiratory morbidity, hospital admissions, and clinic visits during haze episodes (Ramakreshnan et al., 2017).

Previous systematic reviews (SR) and meta-analyses (MA) reported similar findings for healthcare utilization during haze episodes (Cheong et al., 2019). Nevertheless, direct comparison of the effect size and associated risk were difficult as the methodologies and parameters used in those reviews differed from those used in this study. Instead of categorizing exposure

TABLE 6 | Crude and adjusted odds ratios in univariate and multivariate regression analyses (outpatient).

	Univariate		Multivariate (Model 1)		Multivariate (Final model)	
Variable	**Crude OR (95% CI)**	***P*-value**	**Adjusted OR (95%CI)**	***P*-value**	**Adjusted OR (95%CI)**	***P*-value**
Haze episode	2.20 (1.628–2.980)	<0.001	1.57 (1.119–2.203)	0.01	2.21 (1.635–2.999)	<0.001
O_3	0.96 (0.946–0.968)	<0.001	0.96 (0.946–0.972)	<0.001	0.95 (0.943–0.967)	<0.001
CO	1.97 (1.377–2.807)	0.01	1.58 (1.098–2.270)	0.14		
Wind speed	1.28 (1.184–1.386)	<0.001	1.22 (1.081–1.373)	0.085		
Temperature	1.14 (1.004–1.284)	0.05	0.83 (0.704–0.967)	0.091		
Humidity	0.94 (0.924–0.960)	<0.001	0.99 (0.965–1.018)	0.673		

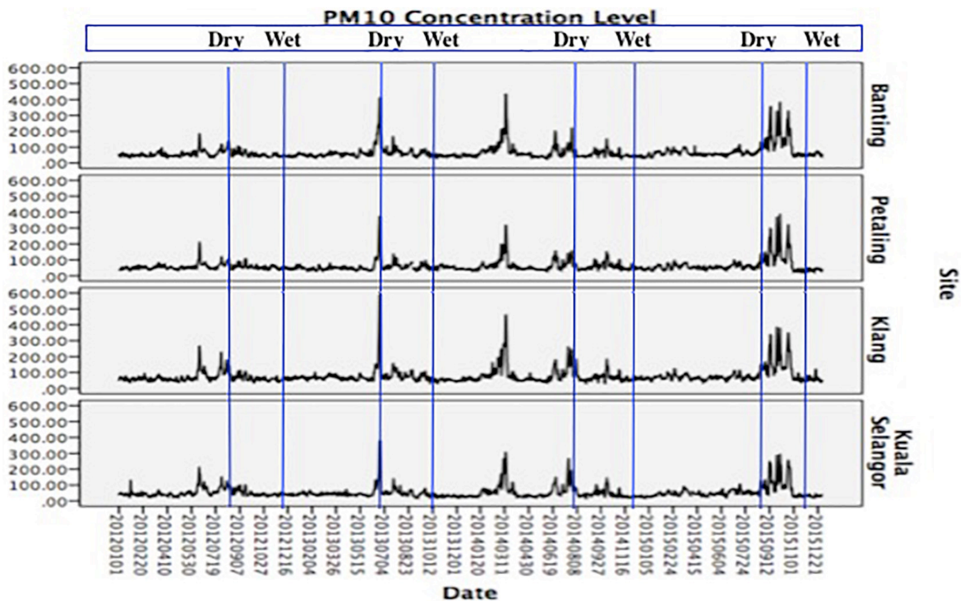

FIGURE 2 | Trend of PM_{10} ($\mu g/m^3$) level in Selangor 2012–2015.

into the haze and non-haze episodes, most of the SRs and MAs used particulate matter ($PM_{2.5}$ and/or PM_{10}) as a continuous variable to determine the linear relationship with healthcare utilization. In addition, the outcome parameters measured were in the form of daily data compared to monthly healthcare utilization data used in this study. The summary conclusion from the reviews was that for every 10 $\mu g/m^3$ increase in PM_{10}, there was a 0.51–3.7% increase in the hospital admission rate and a 0.3–3.7% increase in outpatient visits (Kochi et al., 2009; Adar et al., 2014; Atkinson et al., 2014; Lu et al., 2015). Recent research on the health impact of both PM_{10} and $PM_{2.5}$ also showed an increase of 0.43% in respiratory hospital admission cases with every 10 $\mu g/m^3$ increase in PM_{10} (Qiu et al., 2018) and 0.67–1.78% increase in emergency room visits with 10 $\mu g/m^3$ increase in $PM_{2.5}$ (Mo et al., 2018). Although the studies differed in the definition of haze episode, type of illnesses and parameters used, methodological approaches and effect size measurement, the outcomes were still consistent, supportive and comparable with the findings from this study.

During haze episodes, particulate matter was the most dominant air pollutant. Previous studies showed that particulate matter triggered inflammatory reactions, mainly affecting the respiratory system, and resulted in AEBA and AECOPD (Anderson et al., 2012; Athanazio, 2012). As the result of exposure to higher concentrations of particulate matter during haze episodes, the number of AEBA and AECOPD cases was also expected to rise and subsequently lead to an increase in healthcare utilization during haze episodes, as seen in this study.

Accordingly, two action plans—the National Haze Action Plan and the Haze Management Action Plan—were developed by DOE and MOH, respectively, to reduce the haze-related clinical burden in Malaysia and identify actions that need to be taken during haze episodes (Sulaiman, 2017). From a health perspective, the main focus was on increased awareness of and reduced exposure to haze among the public, to minimize the health impact of haze. The MOH played several roles, including public advisory, health education and promotion, disease surveillance, and disease management. Advice to reduce outdoor activities and wear protection (such as a face mask) was disseminated through social and electronic media, press releases and pamphlets at strategic locations to reach the target population. In addition, decisions to close schools and to cancel public events were also taken based on recommendations from the MOH (Sulaiman, 2017).

However, the strategies used to minimize the impact of haze on health were problematic because of the definition of haze episode used by the DOE. Interventions and advice were only deployed when the PM_{10} concentration level was more than 150 $\mu g/m^3$ (API > 100). In fact, during the 2015 haze episode, the decision to close schools was only made when the PM_{10} concentration was more than 250 $\mu g/m^3$ (API > 150) (Sulaiman, 2017). The daily PM_{10} concentration levels at that time were persistently high (250–350 $\mu g/m^3$) for 2 months (September to November 2015) and recorded the longest streak of haze episodes compared to the year 2012, 2013, and 2014 (Jaafar, 2019). It is preferable to use a lower threshold for the definition of haze based on the World Health Organization Air Quality Guidelines (WHO AQG) as the main reference. All interventions and advice to minimize the health impact of haze should be initiated once the PM_{10} concentration exceeds the target level set by WHO AQG. If the action plan were based on the WHO AQG target level, there is a higher probability that the healthcare utilization rate during haze episodes could be reduced.

Apart from the target level used to define a haze episode, the effectiveness of the interventions is also open to question. For example, one recommended strategy is to use surgical masks and N95 respirators. While the N95 respirator offers better protection than a mask, its cost and the breathing discomfort associated with wearing it reduce compliance. Although the effectiveness of the surgical mask is still debated, it is more convenient, readily available and cheap, and is more widely used as personal protective equipment (PPE) from haze compared to the N95 respirator (Sulaiman, 2017). In addition, advice to reduce outdoor activities and stay indoors during haze fails to consider the indoor air quality and infiltration rate (indoor-outdoor ratio) of pollutants for different types of buildings (naturally ventilated or air-conditioned). The use of air filters should be considered as an intervention that can improve indoor air quality, especially during haze episodes (Sulaiman, 2017).

Apart from haze episodes, other factors that could contribute to the increase in healthcare utilization due to AEBA and AECOPD in this study were assumed to have comparable triggering capacity during haze and non-haze episodes. For instance, work-related exposure and indoor air pollution (apart from pollutants that infiltrate from outdoor sources) that can trigger the exacerbation were anticipated to be similar and dispersed among the population throughout the study period. In addition, 4 years of cumulative data were considered to account for seasonal variability that might be associated with the healthcare utilization rate for AEBA and AECOPD. In this study, all ecological factors were included and controlled in the multivariate analysis.

Results in this study showed that seasonal variability and ecological factors such as wind speed, temperature and humidity were not significantly associated with the healthcare utilization rate. Similar findings were reported from a local study, which concluded that the exacerbation episodes and healthcare utilization associated with asthma and COPD during haze episodes in Malaysia were not directly influenced by any seasonal factors (Othman et al., 2014). In addition, sub-analysis in the present study of the socio-demographic factors that can contribute to exacerbation episodes revealed no significant differences in the healthcare utilization rate according to gender, ethnicity, and age between haze and non-haze episodes. These findings were supported by a local study that showed no significant differences for respiratory admissions between haze and non-haze episodes by age and gender (Ming et al., 2018). Another local study showed that the total inpatient cases were higher among children, followed by young adults, adults and infants, during haze episodes (Othman et al., 2014). However, the study did not compare total inpatient cases during non-haze episodes to identify differences in the age of patients admitted for respiratory illnesses during haze and non-haze episodes. In another study conducted in Taiwan, a comparison was made on the contribution of indoor and outdoor air pollution to the 1-year prevalence of asthma. The study found that both exposures to smoking/indoor air pollution and exposure to total suspended particles (outdoor air pollution) recorded similar OR (1.29) of developing asthma (Wang et al., 1999).

The present study identified several limitations in the healthcare utilization data. First, the rate of utilization of healthcare services, especially for AECOPD outpatient cases, might be underestimated. There were missing data due to the failure of several health clinics to document the specific diagnosis. Hence the recorded incidence of AECOPD outpatient cases might be lower than the number of actual cases.

Second, for inpatient cases, the selection of healthcare facilities was based on the nearest hospital to the location of CAQM stations in that particular district. However, there is a possibility that patients might obtain treatment from a hospital that was not included in the study area, especially when other public or private hospitals were located quite close to the selected hospital, such as in Petaling district.

For outpatient cases, the healthcare utilization data collected were more complete with less risk of sample contamination. The main issue here was the determination of exposure level. If the haze episode was from a transboundary source, exposure to air pollutants could be assumed to be similar in all health clinics. If it was due to local sources, however, the level of air pollutants measured at the CAQM station might not reflect the actual exposure level at selected health clinics that were far from the stations, even though they were in the same district. For example, if domestic waste burning occurs nearer to a health clinic than to the CAQM station, the exposure levels experienced by people living near the burning site might be higher than the level recorded by the CAQM station.

In addition, no patient-based data were available in this study. The influence of age, socio-demographic background, comorbidity, and other contributing factors for AEBA and AECOPD were assumed to be similar during haze and non-haze episodes. Assumptions were also made that the exposure levels among the population were similar to the readings recorded at the CAQM stations located in the same selected district. The incidence rate might also be higher if self-treating and private patients were included. However, since this study focused on the impact of haze from the healthcare provider's perspective, the exclusion of this information was justifiable.

DATA AVAILABILITY STATEMENT

The data analyzed in this study is subject to the following licenses/restrictions: confidentiality of data involving healthcare utilization information. Requests to access these datasets should be directed to eha@doe.gov.my, norazean@ummc.edu.my, and muzalawati@moh.gov.my.

AUTHOR CONTRIBUTIONS

HJ, AA, MI, and MD developed concepts, researched, and wrote the manuscript. HJ and AA performed analyses. All authors had approved the final manuscript for submission.

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Why Home Gardens Fail in Enhancing Food Security and Dietary Diversity

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Visions of sustainable cities mostly conjure up well tended home and community gardens, where owners and residents plant fruits and vegetables that supply some of their livelihood needs. Indeed, home gardens can contribute to household food security but often fail to do so. Moreover, gardens can provide several additional ecosystem services and impact entire communities. This paper seeks to answer why these gardens often do not provide adequate services to make a substantial contribution to food security and identifies possible solutions. We undertook a case study in South Africa in a low-income former township area. The area is characterized by poverty, high levels of unemployment and food insecurity. We interviewed 140 respondents with home gardens to determine what role their own garden plays in household food security. Only 10% of households were found to be completely food secure. Of the rest, 39% experienced hunger that affected everyone in the household and 51% were at risk of hunger. Despite the fact that 72% of the respondents planted vegetables or fruits, the gardens did not contribute substantially to food security. The respondents mostly bought their food, with subsequent food shortages when they did not have enough money. The dietary diversity and consumption of vitamin A-rich fruits and vegetables were very low. The most important constraints inhibiting urban agriculture in the study area were cultural practices, such as the presence of large, bare, open spaces, or "lebala," the focus of home gardeners on ornamental species and lawns; and a reliance on purchasing of foods.

Keywords: food security, dietary diversity, home gardens, cultural practices, urban agriculture

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INTRODUCTION

Food availability, accessibility and utilization are the three dimensions of food security (Jones et al., 2013). Food security is assured when "all people at all times, have physical, social and economic access to sufficient, safe and nutritious food that meets their dietary needs and food preferences for an active and healthy life" (World Food Summit [WFS], 1996). However, in 2020 it was estimated that between 720 and 811 million people worldwide faced hunger (FAO et al., 2021). Moreover, the report stated that "the world has not been generally progressing either toward Sustainable Development Goal (SDG) Target 2.1, of ensuring access to safe, nutritious and sufficient food for all people all year round, or toward SDG Target 2.2, of eradicating all forms of malnutrition"

(FAO et al., 2021). It has also been estimated that two billion people are deficient in micronutrients, 50 million children under the age of 5 years are dangerously thin and 790 million people have insufficient daily dietary energy intake (International Food Policy Research Institute [IFPRI], 2015). Undernutrition is associated with three million child deaths annually, which is almost half of child deaths globally (Myers et al., 2017). A diverse diet is important to ensure that the requirements for essential nutrients are met (Arimond and Ruel, 2004). As an essential nutrient, vitamin A deficiency, especially in children, is a worldwide problem but particularly so in sub-Saharan Africa, leading to malnutrition, stunted growth and even deaths as children with vitamin A deficiency are at greater risk to die from diarrhea, measles and malaria (Black et al., 2003; United Nations Children's Fund [UNICEF], 2008; World Health Organization [WHO], 2009). In their review on malnutrition and health, Müller and Krawinkel (2005) conclude that "diet-based strategies are probably the most promising approach for a sustainable control of micronutrient deficiencies."

With regards to the accessibility of foods, urban areas are often thought to have an advantage over rural areas, however, recent evidence indicate that the urban poor face distinct barriers that limit their access to healthy diets (Vilar-Compte et al., 2021). Financial barriers are a common feature of less healthy eating patterns throughout the developing world (Faber et al., 2017). A study assessing 76 low- and middle-income countries found that domestic food production, as well as inequality in income and consumption, and market conditions, plays a critical role in the food security of these countries (Thome et al., 2019). With the reality that the majority of people now reside in urban areas (United Nations [UN], 2019), the food security of urban residents needs urgent attention. The increase in severity and frequency of natural disasters coupled with the recent global scale COVID-19 pandemic exacerbated fragile food systems worldwide (O'Hara and Toussaint, 2021; Ruszczyk et al., 2021). The fragility of food systems, through dependence of cities on global resources, point strongly to the fact that cities are not as resilient as they ought to be (Gulyas and Edmondson, 2021). Many new urban residents end up living in peri-urban areas (Baud, 2000) often characterized by informal settlements or "slums" (Smit et al., 2017) which are described as spatial clusters of food insecure (Hunter-Adams et al., 2019) households without access to improved water, sanitation, sufficient living area, permanent dwellings or land tenure (UN-Habitat, 2010).

Thus, an approach that can mitigate food security issues, and potentially enhance livelihoods, is the pursuit and encouragement of urban agriculture (Orsini et al., 2013). Zezza and Tasciotti (2010) defined urban agriculture as "the production of crop and livestock goods within cities and towns." In their multi-country study, they indicated that in many countries greater household dietary diversity correlated with urban agricultural practices (Zezza and Tasciotti, 2010). Tontisirin et al. (2002) also argue that the promotion of home gardens and small livestock production can be used as strategies to address micronutrient malnutrition through increased dietary diversity. The most widespread form of urban agriculture is that based in private gardens (Lin and Egerer, 2018), which includes community, domestic and home gardens (Cilliers et al., 2018). Urban agriculture can provide many benefits or ecosystem services, such as habitat for biodiversity (Lin and Egerer, 2018), mitigation of food security (Aerts et al., 2016), contribution to human nutrition (Boeing et al., 2012), alleviation of poverty (Adeyemo et al., 2017) and improvement of human wellbeing (Othman et al., 2018). Its potential role in enhancing urban resilience is also acknowledged (Gulyas and Edmondson, 2021). Historically gardens have provided resilient food and nutrition security for garden owners during times of economic crisis and food shortages (Barthel et al., 2015; Warren et al., 2015).

However, not all agree that urban agriculture is a silver bullet to solve urban hunger. Warren et al. (2015) found that there is substantial debate about the association between urban agriculture and food security and dietary diversity. In assessing the potential of urban agriculture for improving city resilience in the Global North, Gulyas and Edmondson (2021) proposed five factors which determine the success of urban agriculture practices namely: "its scale, the extent to which it is integrated into the urban fabric, its inclusiveness, the efficiency of food production, and human and environmental safety of practices." They go further to state that "these factors in turn depend on the amount of institutional and public support for urban agriculture, the presence of a sufficient knowledge base to guide policy and practice, communication and collaboration among different actors, and resourcefulness in finding alternative ways to use space and other resources efficiently."

Householders establish and maintain gardens for a variety of reasons. Gardeners who rely monetarily on gardens may be more likely to plant fruit and vegetables (Lubbe et al., 2011; Swanepoel et al., 2021). In contrast, higher incomes and access to urban markets may cause a garden composition shift toward ornamentals which provide aesthetic and cultural ecosystem services (Davoren et al., 2016). More broadly, for gardens as a whole, socioeconomic and demographic factors, such as education level and wealth, have been widely shown to have a positive relation to vegetation cover in human dominated ecosystems (Wang et al., 2015; Lin et al., 2017). Local agricultural traditions and preferences may also influence the composition of vegetation providing specific ecosystem services (Barau et al., 2013; Clarke et al., 2014).

The potential of food production in home gardens for enhancing food security and improving dietary diversity, combined with the debate on its efficacy prompted this study. Specifically, the well documented crisis with undernutrition and the effect of vitamin A deficiencies on populations in low-income countries need urgent attention. Clear evidence exists for beneficial effects of eating fruits and vegetables for preventing chronic diseases (Wang et al., 2014). We apply an ecosystem service approach, with a view on human wellbeing, to home gardens in a former township of South Africa. South Africa is much more developed than the rest of Africa, and yet still has issues of poverty, food insecurity, and poor health (Shisana et al., 2013; Statistics South Africa [SSA], 2016). Our study aims to answer three distinct questions: (1) what is the effect, if any, of home gardens on food security and on dietary diversity? (2) What factors characterize households with higher plant diversity? (3)

Why home gardens often do not provide adequate services to significantly contribute to food security?

MATERIALS AND METHODS

Study Site

South Africa has a rich cultural and ethnic diversity, evidenced by its 11 official languages. It is a middle-income country with around half of the population living in poverty and a fifth in extreme poverty (Statistics South Africa [SSA], 2016). There are high levels of food insecurity, especially in urban informal settlements (32.4%) and for black Africans (30.3%) (Shisana et al., 2013). Formal settlements refer to permanent, local council-organized urban residential areas with water and electricity. Whilst informal settlements are on un-surveyed land, usually in the outskirts of towns and without basic service provision (Statistics South Africa [SSA], 2004). The North-West Province has been reported to have the highest proportion (21%) and the biggest increase of households living in informal settlements nationally (45.1%) (Statistics South Africa [SSA], 2016).

We focused on Ikageng, a peri-urban suburb and former township of the city of Potchefstroom (**Figure 1**) in the North-West Province. Peri-urban refers to an inhabited area on the fringe of a city characterized by relatively high-density housing, poor services, limited commercial opportunities, few recreational green spaces and widespread poverty (Shackleton et al., 2015). Ikageng has a population of 87,701 inhabitants and 26,245 households of which 71% are in formal settlements. However, only 37.6% of households have piped water inside the home, 88.6% have electricity and 80% have a flush toilet connected to sewerage (Statistics South Africa [SSA], 2018). The apparent discrepancy between piped water inside the house and flush toilets is due to the fact that many flush toilets are in separate buildings on the yard. The toilets are connected to municipal sewage infrastructure without further proper plumbing in the homes. In Ikageng, the study site was defined by the catchment area of the Steve Tswete Health Clinic (see Cilliers et al., 2018 for a discussion on the importance of health clinic gardens), which falls into the boundaries of the local election wards 20 and 26 (**Figure 1**). These wards contain both formal and informal settlements.

The Batswana ethnic group is predominant at the study site. In rural settings their home gardens or Tswana tshimo have been described by Molebatsi et al. (2010) as a "model of sustainable resource management." They are informally designed to contain six main micro gardens, namely food gardens (include vegetable gardens and orchards), medicinal gardens, ornamental gardens (flower beds, containers and lawns), structural species (windbreaks, fire screen, shade trees and hedges), bare open areas completely devoid of any vegetation ("lebala") and natural areas left to grow wild ("naga"). However, as the Batswana people become more westernized and urbanized their garden design becomes more "colonial" defined by a more formal garden design with more focus on ornamentals and the absence of some of the micro-gardens (Davoren et al., 2016). Thus, this study captures the full range of gardens from traditional to modern.

Data Collection

We developed a household questionnaire to capture the (**Supplementary Table 1**) socioeconomic data, food security, dietary diversity, home garden benefits (i.e., perceived ecosystem services and uses) and ecosystem disservices, reasons for and against gardening, plants grown in the last 12 months, and garden composition. The questionnaire was developed through a series of interviews and focus groups with local experts including academics, the Director of Hospital Services, health clinic gardeners and agriculture extension workers. The questionnaire was piloted with 25 households, and questions subsequently amended for clarity and relevance, and the inclusion of photos for fruit and vegetable identification in the dietary diversity section. The questionnaire was complemented by a visual survey of the home garden, where plants and trees were identified on site where possible. Unusual species were identified post-interview based on photos and specimens. The fieldwork was conducted in winter, so self-seeding annuals and herbaceous perennials were not visible.

The initial questionnaire was developed in English, surveys were administered in the local language, Batswana. Households were systematically sampled within formal and informal settlements at regular intervals, beginning with streets closest to the health clinic. If no one was home or refused to participate at the selected household, the neighboring house was visited and so on until a participant was found. The systematic sampling approach recommenced from a successfully sampled house. To be included in the sample, households had to own, or have direct access to, a home garden from their house. For the purpose of this study a home garden was defined as a land-use form on private or communal land surrounding an individual house with a defined border (although not always physical), in which several useful plant species are cultivated together (Molebatsi et al., 2010).

Wealth indices are an effective way of measuring socioeconomic status, often outperforming expenditure data or income in explaining variations (Filmer and Scott, 2012; Smits and Steendijk, 2015). When developing our questionnaire, we used an index that has been externally validated in relevant low-income contexts (Filmer and Pritchett, 2001; Nundy et al., 2011). The wealth index is based on data on asset ownership (e.g., private or shared toilet, water tap) and household characteristics (e.g., number of rooms per person). Information on household assets was analyzed using a Principal Components Analysis (PCA), which combines the original questionnaire responses weighted by an asset's contribution to explaining the variance and then used to calculate each observation's score (Filmer and Pritchett, 2001; Nundy et al., 2011). **Table 1** lists all the variables used to calculate the PCA and their calculated weights. As recommended by the Department of Demographic and Health Surveys (DHS) of the United States Agency for International Development (USAID), the individual households were then divided into five wealth quintiles, i.e., from poorest to richest households (Rutstein and Johnson, 2004).

The general categories of the Common International Classification of Ecosystem Services (CICES) of the European Environment Agency were applied to classify the plant uses reported by respondents. Respondent reported uses including food, medicine, firewood (provisioning services), shade, fencing,

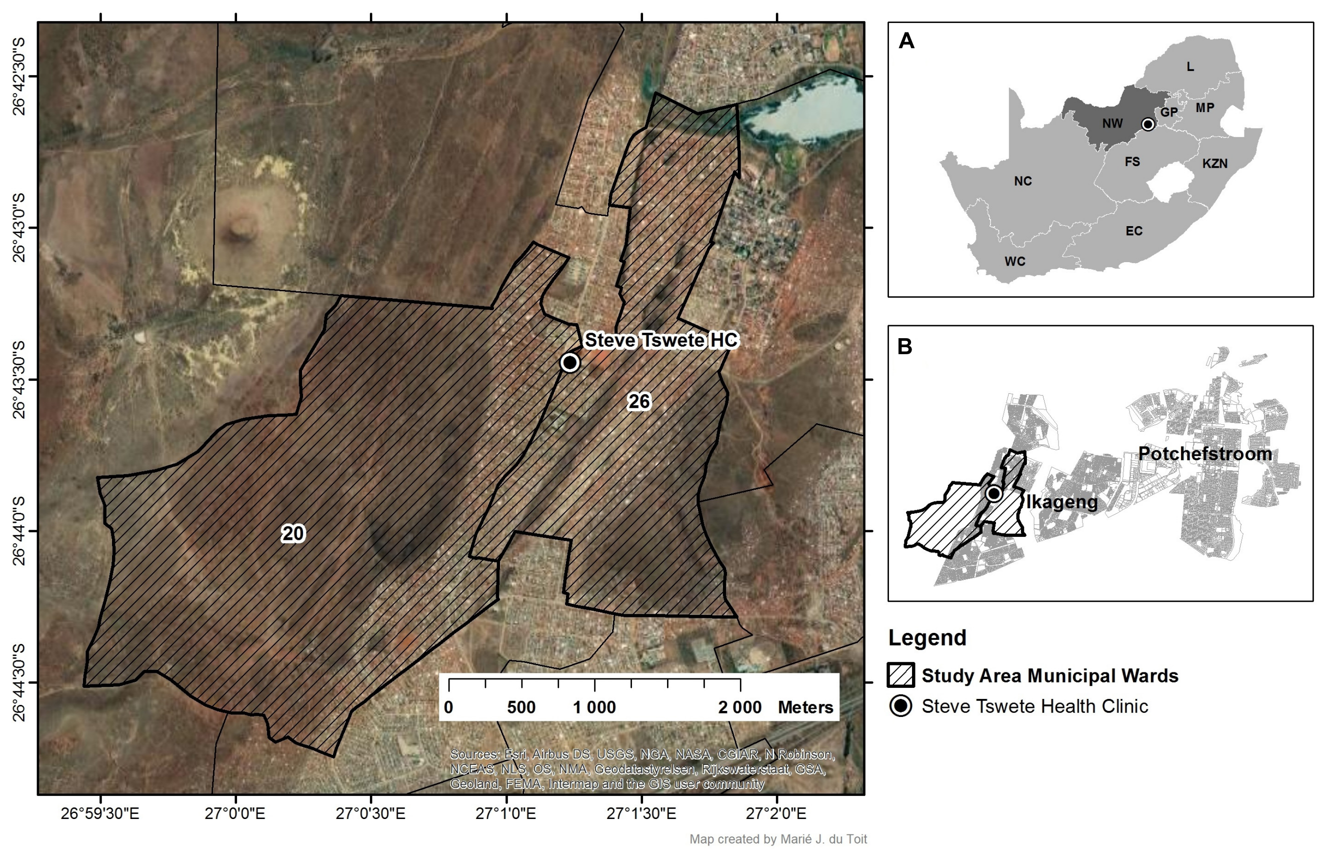

FIGURE 1 | Map of the study area indicating the two Ikageng wards in which the surveys were undertaken. The inset maps show the location of Potchefstroom in South Africa **(A)** and the location of the study area within Potchefstroom **(B)**.

windbreak (regulating and maintenance services), spiritual and ornamental (cultural services). Many species had multiple uses and were noted once for each use. The extent and composition of home gardens was also characterized according to the presence of the six tshimo micro gardens (Molebatsi et al., 2010).

Food security data were obtained using the internationally used and validated Community Childhood Hunger Identification Project (CCHIP) index (Wehler et al., 1992). This index is used by the South African National Health and Nutrition Examination Survey (SANHANES-1) (Shisana et al., 2013). The CCHIP index is based on eight occurrence questions that represent a generally increasing level of severity of food insecurity. They are related to whether the household, adults and/or children are affected by food shortages, perceived food insufficiency or altered food intake due to constrained economic resources within the household (Coates et al., 2007). For each of the questions, participants were asked about occurrences in the 30 days prior to the questionnaire (Shisana et al., 2013). A score of five or more affirmative responses out of eight indicates the presence of food shortage or "hunger" in the household. A score of one to four indicates that members of the household are at risk of hunger. A score of zero indicates that the household is food secure (Shisana et al., 2013; Walsh and van Rooyen, 2015). The frequency of food insecurity occurrence in the last 30 days was coded with the Household Food Insecurity Access Scale Score (HFIAS; Coates et al., 2007). The HFIAS score was estimated by assigning each frequency response to one of four categories: 0 = Never, 1 = Rarely (once or twice in the past 4 weeks), 2 = Sometimes (three to ten times in the past 4 weeks) and 3 = Often (more than ten times in the past 4 weeks). The HFIAS score could range between 0 and 24; the higher the score, the more food insecurity the household experienced.

Dietary diversity was calculated by a recall technique following the Household Dietary Diversity Score (HDDS) as suggested by Kennedy et al. (2010). The study was only interested in the contribution from the number of vitamin A-rich fruits and vegetables consumed (range 0–13), i.e., richness, to nutritional security. A two-level score was estimated from the data: a score of the variety or richness of vitamin A consumption (i.e., number of items eaten); and a score of the abundance of vitamin A consumption (i.e., times eaten and portion size). Vitamin A-rich fruit and vegetables at the study site included: Yellow-orange vegetables (i.e., carrot, butternut squash, Hubbard squash, pumpkin, orange-fleshed sweet potato), yellow/orange (non-citrus) fruit (i.e., mango, pawpaw/papaya, melon, apricot, peach) and dark-green leaves (i.e., spinach and wild and

TABLE 1 | The calculated wealth indexes of 15 household assets indicators sorted by category (house, main construction material, toilet facilities, and access to water) using Principal Component Analysis.

Asset category variable	Wealth index weight
House	
Rooms per person	0.42
Electricity	0.91
Main construction material	
Redbrick	0.25
Concrete	0.55
Aluminum	–0.93
Toilet facilities	
Bucket	–0.06
Pit latrine	–0.20
Working flush toilet	0.70
Not working flush toilet	0.10
Communal toilet or latrine	–0.59
Using neighbor's toilet or latrine	–0.65
Access to water	
No tap	–0.30
Tap in yard	0.37
Tap in house	0.34
Communal tap	–0.87

traditional leaves such as "morogo"). Respondents were first asked if they ever ate the listed plants and then if they had eaten them in the last 24 h, including how many times and the portion size. Therefore, the nutrition and dietary diversity results of this study only refer to the consumption of vitamin A-rich plant sources.

Data Analysis

Four Quasi-Poisson regressions were carried out to determine (a) what characterized households with more or less total number of plant species grown in their garden, (b) what characterized households with food insecurity or a high CCHIP index, (c) what characterized households with different frequencies of food insecurity (HFIAS), and (d) what characterized respondents with different Dietary Diversity Score (DDS) i.e., richness of vitamin A-rich plants consumed. The variables used in the different regressions are presented in **Table 2** and were selected based on existing literature on community and home gardens.

This study is not without limitations. We acknowledge that studies like the current study which does not include vitamin A rich foods from other sources (West and Darnton-Hill, 2008) cannot truly assess the real dietary diversity of interviewed respondents. The findings only focus on home gardens as a source of vitamin A-rich plant consumption and as such the contribution of animal foods for respondents who prefer it instead of eating fruits and vegetables was not recorded. Furthermore, although the study covers 1 year of garden cultivation, seasonal variation in garden produce was not considered in the study design.

RESULTS

Respondent and Household Characteristics

The overall sample consisted of 140 households with respondents who self-identified as the home gardener. Results were included in **Table 3**. The sample was close to an equal balance between men and women, which was an interesting finding as the literature often reports a high proportion of women as urban small-scale farmers (Orsini et al., 2013). Just over half of respondents were Batswana. The next most common ethnic groups were Sotho and Xhosa. Levels of education were low, with two thirds only completing high school. Half of respondents were in a relationship, where cohabiting was predominant. There was a high level of unemployed respondents, with only 11% working full time. There was a high level of households with one or more unemployed persons and just over a third had one or two people employed full time. Households reported a median of 4 family members. Households were distributed quite evenly across the wealth index quintiles.

Just over half of houses were made of concrete block, followed by red brick or aluminum. Houses had a median of five rooms including add-on structures in backyards. Most houses had a water tap, in either their yard or house, an electricity connection and a functional flush toilet. Although most households had access to basic services, there were some houses without electricity, having to borrow a neighbor's toilet or without access to water (**Table 4**).

Home Garden Composition and Ecosystem Services

Most households (91%) had a garden with vegetation, while the rest (mainly in informal settlements) only had bare open space (i.e., lebala). More specifically, 93% of households had bare open space, 61% had ornamental beds, 60% had fruit, 58% had lawn, 48% had shade trees, 41% vegetables, 31% had medicinal plants, 25% had hedges and 4% had natural areas. The largest overall micro garden cover was bare open space in 81% of households and the largest vegetated micro garden cover was lawn (38%). Average vegetable area per household was 11 m^2, ranging from 0.01 to 98 m^2. A total of 87 plant species were recorded (**Supplementary Table 2**), of which 30 provided shade, 26 were vegetables, 23 fruits, 21 medicinal, 12 for firewood, and 8 for spiritual use. Only vegetables and fruit grown in the last 12 months were recorded.

The vegetables and fruit grown was used for consumption in all households but some was at times sold (a mean 9% of vegetables, range 4–17%; and 6% of fruit, only one instance) or gifted to friends and neighbors (mean 23% of vegetables, range 5–60%; 37% of fruit, range 8–67%). All vegetable and fruit growers recognized the food benefits (provisioning services) from their garden but only 11% of respondents mentioned any other benefits from their garden vegetation: relaxation and aesthetics (cultural services) and shade, wind protection and dust protection (regulating services). Just 4% of respondents

TABLE 2 | The explanatory variables used in the four Quasi-Poisson regressions to determine the characteristics of the respondents and their households, with their expected sign and justification.

Variable name	Description	Expected sign	Justification
Female	Gender of respondent: 1 = female, 0 = male	Positive	Women have been found to be more involved in urban agricultural practices (Magidimisha et al., 2013)
Age	Age of respondent	Positive	Younger adults are less involved in gardening than older people (Dunnett and Qasim, 2000)
Low education	Education level of respondent: 1 = primary school completed or less, 0 = high school or higher	Negative	Respondents with at least some high school education (≥ 8 years) were more likely to be food secure (Faber et al., 2017)
Employed per household	Proportion of people employed per household	Negative	Reflects less time available to garden (Kornrich and Roberts, 2018)
Wealth	Wealth quintile of household: 1 = lowest, 5 = highest	Positive	Assets, such as type of dwelling, have been found to be significant for food security (Walsh and van Rooyen, 2015). Individuals with access to resources, labor or financial means are capable of effecting change in their urban environment, e.g., increase species richness (Davoren et al., 2016).
Children present	1 = Children under 18 years old are present in the household; 0 = no children present	Positive	Households with children are more likely to want to provide for their children and sacrifice for them (Walsh and van Rooyen, 2015)
Batswana language	1 = Batswana home language; 0 = other home language	Positive	Individuals' food intake is influenced by the beliefs and behaviors of the different ethnic and cultural groups (Wentzel-Viljoen et al., 2011). Batswana people are known for the Tswana tshimo or micro gardens (Molebatsi et al., 2010).
Informal settlement	0 = Formal settlement; 1 = Informal settlement	Negative	Informal settlement households have no garden area or resources to grow vegetables and/or fruit (Cilliers et al., 2013)
Know health clinic garden	Know about the existence of their local health clinic garden	Positive	Health clinic gardens have a positive influence on home gardening (Cilliers et al., 2018)
Grew vegetables and fruit	1 = Has grown vegetables and/or fruit in last 12 months; 0 = has not	Positive	Households growing vegetables and/or fruit have higher food security (Drechsel and Dongus, 2010; Zezza and Tasciotti, 2010)
Dietary diversity	Dietary Diversity Score (DDS)	Positive	Households growing vegetables and/or fruit have higher dietary diversity (Johns and Eyzaguirre, 2006)
Total vitamin A	Consumed Vitamin A (RE mcg per 100 g) from fruit and vegetables in last 24 h	Positive	Vitamin A–rich food sources were significantly higher for food secure households, pointing toward a more frequent dietary intake of vitamin A–rich foods in the food secure households for both plant and animal sources of vitamin A (Faber et al., 2017).

mentioned negative effects, e.g., allergy to a plant. However, when queried directly about plant usage, respondents reported growing plants for medicine (34%), firewood (29%) and spiritual uses (19%; e.g., wild garlic to repel snakes, indigenous *Aloe* species for funeral cleansing).

A quasi-Poisson regression to determine what characterized the number of plant species grown at each household explained 30% of variance (**Table 5A**). Older gardeners grew more plant species, female gardeners grew more plant species than men, and the number of species grown increased with wealth. Specifically, wealth quintiles 2, 3, 4, and 5 were all significantly higher than wealth quintile 1. Wealth quintile 3 was significantly higher than 2, but there was no difference between 3, 4, and 5. So there is a threshold around wealth quintile 3 above which respondents with higher wealth did not increase the number of species grown.

A total of 72% households had grown vegetables and/or fruit in their garden and 12% had vegetables planted by the municipality (i.e., with or without their consent). The most frequently grown vegetable was spinach (35%) and fruit was peach (54%). The main reasons for growing vegetables and fruit were to use available land (53%), lack of money to buy vegetables (25%), to gift to neighbors (9%), and other minor reasons (13%). Contrastingly, the main reasons for not growing (more) vegetables and fruit in their garden were bad, rocky or muddy soil (40%), limited or no land (28%), lack of money (19%), lack of gardening skills/knowledge (6%), and other minor reasons (7%).

Food Insecurity and Home Gardens

Only 10% of households were found to be completely food secure. Of the rest, 39% experienced hunger that affected everyone in the household (38% experienced it in the last 30 days) and 51% were at risk of hunger (37% experienced it in the last 30 days). Thus, over a third of households are food insecure and over half of households are at risk of becoming food insecure. The CCHIP Index ranged from 0, being food secure, to 8, being food insecure, with a median of 3. The average number of days in the last 30 days in which a food insecure condition was experienced was 6.5 days for adults and 2.3 days for children. The survey also indicated parents going without food to feed children as a coping strategy to deal with food insecurity. The HFIAS score for frequency of food insecurity occurring in the past 30 days had a median of 4 with a range between 0 and 22 (out of 24 maximum).

A quasi-Poisson regression to determine what characterized households with food insecurity found that respondents with

TABLE 3 | The socioeconomic characteristics of the respondents of 140 households who self-identified as the home gardener.

Variable	Median or percentage	*SD*	Range
Female	55%		
Age	44	15	20–96
Education level	2% Post-school 60% High school 27% Primary school 11% No education		
Home language	55% Batswana 24% Sotho 19% Xhosa 2% Others		
Family members	4	2.48	1–18
Marital status	31% Single 31% cohabiting 21% married 17% divorced, widowed, separated		
Employment status	44% Unemployed 21% retired 16% work part-time/temporarily 11% work full-time 8% housewives/unable to work		
At least one family member unemployed	68%		
At least one family member employed (respondents could select more than one option as needed)	38% full-time 40% part-time		
Wealth index quintiles (Q)	14% Q1 29% Q2 16% Q3 21% Q4 20% Q5		

TABLE 4 | House characteristics of the respondents of 140 households who self-identified as the home gardener.

Variable	Median or percentage	*SD*	Range
House material	55% Concrete 24% red brick 21% aluminum		
Number of house rooms	5	1.93	1–10
Water access	48% Water tap in yard 36% water tap in house 13% communal tap 3% no tap		
Electricity	86%		
Toilet access (1% missing data)	64% Functional flush toilet 16% broken flush toilet 8% communal toilet 7% neighbor's toilet 4% pit latrine		

low education had a higher CCHIP index, indicating they were more food insecure (**Table 5B**). While households with a higher proportion of employed persons and with the highest wealth quintile were more food secure. More specifically, wealth quintiles 4 and 5 were significantly different from quintiles 1, 2, and 3 but were the same to each other. Growing vegetables and/or fruit showed a negative relationship to food insecurity although it was only significant at 90% level, indicating a suggestion of a correlation between growing food and vegetables and increased food security.

A Quasi-Poisson regression was also estimated to determine what characterizes the frequency of household food insecurity (HFIAS score). Food insecurity is more frequent for respondents with low education and is less frequent for households with a higher proportion of employed persons per household. These results are similar to those in **Table 5B** for presence of food insecurity and are thus included as **Supplementary Table 3**.

Dietary Diversity and Home Gardens

Respondents reported consuming on average eight (range 4–11) vitamin A-rich vegetables and fruits. Carrot, spinach, peach and apricot were the most frequently consumed in their lifetime. When asked about consumption in the last 24 h, on average 1 plant was consumed (range 0–4) and 51% of households had not eaten any of the vitamin A-rich vegetables and fruits. The most frequently consumed vegetables and fruit in the last 24 h were spinach, Hubbard squash and carrots. These findings evidence a very low consumption and dietary diversity of vitamin A-rich vegetables and fruit.

A Quasi-Poisson regression did not find evidence of explanatory factors for the Dietary Diversity Score (DDS) i.e., richness of vitamin A-rich plants consumed (**Supplementary Table 4**). It is likely that wealth, and other variables, are not significant as this study does not cover all possible food sources of vitamin A, i.e., animal products.

TABLE 5 | Quasi-Poisson regressions for: **(A)** Total number of plant species grown per household (1 missing value) and **(B)** household food insecurity (CCHIP) (2 missing values).

	A) Number of species grown	B) Household food insecurity
Variable	**Estimate (S.E.)**	
(Intercept)	–0.222 (0.405)	1.864 (0.213)***
Low education	0.301 (0.181)	0.242 (0.114)*
Employed per household	–0.003 (0.003)	–0.010 (0.003)***
Wealth 2	0.748 (0.327)*	–0.198 (0.181)
Wealth 3	1.345 (0.327)***	–0.139 (0.213)
Wealth 4	1.028 (0.340)**	–0.533 (0.224)*
Wealth 5	0.988 (0.335)**	–0.529 (0.211)*
Female	0.326 (0.145) *	
Age	0.012 (0.006)*	
Children present	–0.242 (0.165)	
Know health clinic garden	0.166 (0.169)	
Grown vegetables and fruit		–0.256 (0.152).
Informal settlement		–0.097 (0.203)
Batswana language		–0.001 (0.115)
Dietary Diversity Score		0.013 (0.095)
Total vitamin A		–0.000 (0.000)

*Significance codes: 0.0001 = "***"; 0.001 = "**"; 0.01 = "*"*

DISCUSSION

Potential of Gardens to Reduce Food Insecurity and Increase Dietary Diversity

Two of the main causes for food insecurity in South Africa are weak support networks and inadequate and unstable household food production (Shisanya and Hendriks, 2011). Food insecurity has been found to be significantly more prevalent in urban than in rural areas of South Africa, as well as in informal settlements vs. formal settlements (Walsh and van Rooyen, 2015). Thus, there is a belief that home gardens have the potential to contribute substantially to household food security (Cilliers et al., 2018). However, the presence of a home garden does not guarantee food security of the adults or children of a household. As our results indicated that despite 72% of the respondents growing fruit or vegetables only 10% were completely food secure. The garden composition results indicated that planting of fruits and vegetables were not the highest priority in the home gardens. Bare open spaces "lebala" and then a lawn occupied most of the space in the gardens. Even the maximum vegetable area was much lower than the 230 m^2 recommended by Trainer (1995) as the minimum area required to feed one individual. This provides indirect evidence of the shortfall of home gardens toward household food security. Faber et al. (2017) reported that the frequency of household consumption of vegetables and fruit was lowest in food insecure households, mostly due to financial constraints.

The questions asked in the survey on food security all revolved around money to buy food. Together with the results from the garden survey, findings indicated that respondents mostly rely on money to buy food, rather than producing their own food. Moreover, the main reasons for growing vegetables and fruit did not include food production. Respondents stated that they mainly did it to use the available land (53%) or due to a lack of money to buy vegetables (25%).

At the study site, food insecurity figures are similar to those reported by Walsh and van Rooyen (2015) for the neighboring Free State Province in South Africa. However, our findings show a much higher incidence of food insecurity than the national averages, which are 68.5% for informal and 44.6% for formal settlements (Shisana et al., 2013). The results on parents going without food to feed their children as a coping strategy is similar to findings in other studies (e.g., Walsh and van Rooyen, 2015).

Home gardens in our study were not found to be making a substantial contribution to vitamin A consumption of households or food security. Contrastingly, Faber et al. (2017), have found a more frequent dietary intake of vitamin A–rich foods in food secure households for both plant and animal sources of vitamin A. Other studies in Sub-Saharan Africa have shown the importance of gardens to increase nutrient uptake and diversification (Johns and Eyzaguirre, 2006) and traditional leafy vegetables can considerably increase iron and vitamin A in the diet (Van Jaarsveld et al., 2014). It seems that these benefits are more commonly observed in rural areas growing more vegetables and fruit (Faber et al., 2010). It is important to note that the Dietary Diversity Score is based on a single day's food intake, which usually has a day-to-day variation, particularly for non-staple foods, and availability across seasons has an effect (Faber et al., 2017).

Garden Composition and Perceptions

The study could not identify characteristics that fully explained the number of plant species grown in the gardens. Of the variance explained, wealth, gender and age were the most important. The study site gardens can be described as transitional gardens (high lawn and bare open space areas) which are between traditional (mainly useful plants with large areas of bare open space and natural areas with wild vegetation) and European colonial gardens (mainly ornamental and lawn with practically no bare open space) as described by Davoren et al. (2016). There was a predominance of large areas of bare open space, lawn and ornamentals, with smaller areas of utilitarian plants such as those with a spiritual use. Both bare open space, a traditional garden cover, and lawn, a more colonial garden cover, are in conflict with growing more utilitarian plants. Several authors have mentioned that crops and plants are strongly related to local culture and traditions (e.g., Orsini et al., 2013). Bare open spaces have important cultural significance as it indicates the tidiness of the household (Cilliers et al., 2009) and are immaculately swept daily (personal communication). It also offers another key service as it is used for safety, however, it means that many households are trading off the opportunity to grow food items by leaving all or a large area of their yard as an open space. These transition gardens are also retaining traditional spiritual use plants (such as wild garlic) and giving up traditional food plants. Other studies also reported that households in urban or peri-urban areas of South Africa have a predominance of ornamental plants over food and medicinal plants (Mosina et al., 2014). The transitional gardens evidences how socioeconomic status overrides cultural preferences as peri-urban residents gain access to resources needed to effect change in their gardens (Davoren et al., 2016).

Garden composition at the small scale was also similar to other studies. For instance, van Vuuren et al. (2020) and Nemudzudzanyi et al. (2010) also reported the presence of spiritual-relevant plants. Likewise, the most predominant plants found coincided with those reported by Walsh and van Rooyen (2015) in Free State Province, South Africa. Home gardens can contribute to human wellbeing beyond just health (via food). When the reported ecosystem services are linked to human wellbeing (Smith et al., 2013) we find that home gardens also contribute to spiritual and cultural fulfillment, connection to nature (via relaxation and aesthetics), social cohesion (via gifted vegetables and fruit), and living standards (via sold vegetables and fruit). It is also important to note the wellbeing contribution of having bare open space to safety and security, as it was associated with not providing vegetation for snakes or criminals to hide in near the house (Molebatsi et al., 2010).

Authors, such as Clarke et al. (2014), have hypothesized that the higher number of ornamental species and decreased edible cover in suburban and peri-urban gardens is attributed to luxury investments. These are due to the desire to imitate European colonial gardens that provide a certain status. However, the findings point to these luxury or "status" investments happening in low wealth households at risk of or food insecure, and with low dietary diversity, especially regarding vitamin A consumption. The way in which gardens are transitioning from Tshimo to European colonial gardens is influencing the ecosystem services of food production (and its associated wellbeing) that households could derive from their home gardens.

Acknowledgment of the value of an ecosystem service may vary with the role and perception of the stakeholder and the ecosystem service considered (de Groot et al., 2010). There is a clear mismatch between the participants' perceived ecosystem services from their garden (and thus the wellbeing they derive from it) and those observed by researchers. This has been reported by other authors regarding the relationship between biodiversity of green urban spaces and psychological wellbeing (Dallimer et al., 2012; Gaston et al., 2013). A key benefit from trees that users of urban green spaces highlight in sub-Saharan Africa is their role in providing shade (Guenat et al., 2019). However, few participants perceived such benefits from their garden trees and hedges, perhaps partly explaining why there were so few of these beneficiary plants. Similarly, many participants perceived that they had insufficient space as part of their properties to allow them to maintain a garden, and thus grow food or other beneficial plants. This is probably linked to a lack of gardening knowledge regarding required space for crops. This perception is a crucial barrier to the generation of ecosystem services from home gardens as participants decide not to plant at all. It is vital to overcome these awareness and knowledge deficiencies as a first step to get households' "buy-in" into urban gardening.

Why Home Gardens Fail

Several studies indicate that despite unemployment or low incomes with related food insecurity, the main source of food consumed is food that is purchased with a low reliance on self-production (e.g., Acquah et al., 2014; Crush and Caesar, 2014; Ngema et al., 2018; Garekae and Shackleton, 2020; Lowe et al., 2021). Moreover, in Windhoek, Namibia, 51% of the respondents indicated that buying food is much easier than growing food, with 46% indicating that they are not interested at all in growing food (Crush et al., 2018). In this study, gardens consisted mainly of large bare open spaces, lawns, and ornamentals with very small vegetable gardens and areas with fruit trees. The main function of the garden was not for food production. Moreover, the main reasons given for growing fruit and vegetables were to use the land available (53%) and due to a lack of money to buy vegetables (25%). When asked why they did not grow (more) vegetables and fruits the reasons included bad, rocky or muddy soil (40%), limited or no land (28%) and a lack of money (19%). The reliance on home gardens when respondents do not have money to buy food and the explanation that a lack of money constrained them from not planting more vegetables is a major conundrum for expanding urban agriculture. As evidenced by the studies mentioned above, if more money were provided, householders would be more likely to use this for buying food than to expand their vegetable gardens.

Gardens are often used to improve the status of the householder. For instance, in a Chinese study on suburban and peri-urban gardens, Clarke et al. (2014) suggested that gardens with high ornamental species richness and decreased edible cover can be attributed to luxury investments. Moreover, a South African study indicated that especially young people have negative feelings toward urban agriculture stating that it is "not modern" and that they were "not interested" (Thornton, 2008). Fruit and vegetables are also not the only source of food for urban dwellers with many preferring and supplementing their diets with animal products (Reynolds et al., 2015). This preference for animal products can also impact the success of gardening endeavors.

Home and community gardens also often cannot produce all year round or supply enough for household needs. A survey on urban farmers working in community gardens in Emfuleni, South Africa showed that 86% of the participants agreed that they could supply fresh vegetables to their own families from the garden. However, 41% indicated that it was not enough to feed their families (Modibedi et al., 2020). Moreover, 54% of the respondents stated that they could not rely on daily supplies of vegetables due to unreliable production. Further challenges affecting successful urban agriculture include land tenure insecurities, land use conflicts, water accessibility, weak regulatory frameworks to support urban agriculture (Puppim de Oliveira and Ahmed, 2021), limited space in urban areas (Lowe et al., 2021), and insufficient resources to maintain or start a garden (Bannor et al., 2021). Moreover, the safety of food produced using urban wastewater, planting crops or rearing livestock or poultry on polluted soils and indiscriminate use of pesticides, are crucial issues that impact acceptance and consumption of urban agricultural products (Bannor et al., 2021; Gulyas and Edmondson, 2021). These health concerns cause people to prefer to buy food from supermarkets (Wertheim-Heck et al., 2019).

Strategies for Successful Adoption of Urban Agriculture

Social grants have been endorsed as a way to ensure that food security problems are addressed, and for many it is their sole source of income. On its own, grants are inadequate to eliminate food insecurity and to date, in South Africa, has not significantly reduced malnutrition (Chakona and Shackleton, 2019; Waidler and Devereux, 2019). The main reasons for this are the fact that food prices continually increase but the grant amount does not, moreover grants are not solely used to buy food (Chakona and Shackleton, 2019; Waidler and Devereux, 2019). Therefore, urban agriculture has a crucial role to play as "safety net" and secondary food supply (Zimmerer et al., 2021) and ensuring local food supplies in times of crises such as the COVID-19 pandemic where global supply chains were severely affected (Gulyas and Edmondson, 2021). To promote urban agriculture, cooperation is needed with government entities on multiple levels as well as stakeholder participation in decision-making (Obosu-Mensah, 2002; Puppim de Oliveira and Ahmed, 2021). Moreover, viable agricultural land should be protected by acknowledging its role as a citizen-led urban green strategy and through relevant policies and the inclusion of urban agriculture in mainstream urban planning and development (Cilliers et al., 2020; Bannor et al., 2021; Steenkamp et al., 2021). Strategies should also focus on planting food with high nutritional value, combining it with poultry and livestock, and striving for gardens with high diversity specifically focusing on the richness and abundance of species to ensure year-round production (Lowe et al., 2021). Educating householders on optimal gardening practices and providing adequate support can greatly enhance the success of urban agriculture toward reducing food insecurities. However, in areas such as South Africa where cultural practices have a major influence on garden design, it is imperative to understand and find possible solutions for how cultural practices and optimal food production can coexist without residents losing their cultural identity. The excellent review of Gulyas and Edmondson (2021) on urban agriculture in the Global North proposed factors that we feel are universal for low- and middle-income countries as well and addressing these will greatly improve the value and contribute on urban agriculture to urban food security. These factors are: "the scale, the extent to which it is integrated into the urban fabric, its inclusiveness, the efficiency of food production, and human and environmental safety of practices."

CONCLUSION

The problem of reduced dietary diversity and food security is, of course, influenced by more than just home gardens (e.g., access to vegetables and fruits at shops, tastes) and needs to be dealt with a holistic approach that includes home gardens. Households need to be provided with the skills, resources and knowledge to be able to grow fruit and vegetables (e.g., soil improvement, space management) for food availability. The conflict of maintaining certain traditional cultural practices (e.g., bare open space, spiritual plants) and the loss of other traditional garden attributes (e.g., vegetable and fruit growing) with regards to the imitation of European colonial gardens (e.g., large lawns) that limit garden benefits needs to be further studied and managed. Further research could explore which ecosystem services and wellbeing benefits are more important to households and why. This could inform how to best manage culturally significant bare open space areas and also growing vegetables and fruits for food, income and social cohesion.

Low- and middle-income countries, like South Africa, have the potential to enhance peoples' food security and dietary diversity by encouraging the growing of fruit and vegetables in urban home gardens. However, there are many limitations to this, ranging from a lack of awareness of other garden benefits, lack of gardening skills and knowledge, cultural loss of utilitarian gardens and imitation of ornamental European garden styles, and an over reliance on purchasing of foods. The current protracted COVID-19 pandemic has shown that global supply chains can be vulnerable to disruption (Xu et al., 2020) and that low levels of self-sufficiency can have negative consequences for food security when food systems fail (Garnett et al., 2020; Reardon et al., 2020). The threat of global climate change further impels scientists and decision-makers to find more sustainable and resilient local food system pathways (Ghadge et al., 2019). Despite our findings of a weak contribution of home gardens to underpinning food security and dietary diversity, it is still worth trying to find ways to overcome the challenges constraining effective urban agricultural practices.

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.

ETHICS STATEMENT

The studies involving human participants were reviewed and approved by the Ethics Committee of the Faculty of Health Sciences, North-West University in Potchefstroom (NWU-00064-16-S1). The patients/participants provided their written informed consent to participate in this study.

AUTHOR CONTRIBUTIONS

MD, SC, and OR conceived and designed the study. OR performed the data collection. OR and VC analyzed the data, with help from MD. MJD created the map. MJD, OR, and VC wrote the manuscript. All authors contributed to the final draft and gave final approval for publication.

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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.2022.804523/full#supplementary-material

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Changes in Green Space Use During a COVID-19 Lockdown Are Associated With Both Individual and Green Space Characteristics

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Mobility restrictions imposed during the COVID-19 pandemic present a useful study system for understanding the temporal and spatial patterns of green space use. Here, we examine green space characteristics and sociodemographic factors associated with change in frequency of green space use before and during a COVID-19 lockdown in Brisbane, Australia drawing on a survey of 372 individuals. Applying regression analysis, we found that individuals who visited a different green space during lockdown than before tended to decrease their frequency of visits. In contrast, individuals who continued visiting their usual green space during lockdown were more inclined to increase their number of visits. Changes in frequency of green space use were also associated with particular characteristics of their usually visited green space. The presence of blue spaces and accessibility (carparks/public transport) were associated with increased frequency of use while foliage height diversity was associated with reduced frequency of use. We found that females were more likely to change their green space visitation frequency during COVID-19 compared to men and they also reported greater importance of green spaces for social and family interactions and spiritual reasons during COVID-19 compared to before. Males showed greater increases than females in the importance of green space for nature interactions and mental health benefits during the COVID-19 lockdown compared to before. Our results provide key insights for future resilient urban planning and policy that can fulfil a wide range of physical and psychological needs during a time of crisis and beyond.

Keywords: parks, wellbeing, human health, self-perceived benefit, ecosystem services, human-nature interaction, urban nature, pandemic

INTRODUCTION

On March 11 2020, the World Health Organisation declared COVID-19 an international health emergency. Following the announcement, countries across the world took various measures to slow the spread of the virus that causes COVID-19, including stay at home orders, closures of schools and workplaces, and significant limits on travel. These restrictions inevitably disrupted people's daily activity patterns, social interactions, and use of leisure facilities such as gyms, cafes, and places of worship. Such disruptions caused uncertainty and instability, taking a toll on individual health and community wellbeing (Godinić et al., 2020; Wang and Boros, 2021). The effects were exacerbated in urban areas where dense living arrangements, longer and more severe restrictions, higher infection risk, and unequal access to services and nature coalesce (Maher, 1994; Sharifi and Khavarian-Garmsir, 2020; Mouratidis, 2021; Spotswood et al., 2021). Outdoor green spaces were one of few recreational places that remained accessible during periods of lockdown.

The importance of urban green spaces for ecosystem services—the benefits humans derive from nature—is well established in the literature (Bolund and Hunhammar, 1999; Breuste et al., 2013; Haase et al., 2014). Green space use is positively associated with general physical health and mental health and wellbeing (Grilli et al., 2020). For instance, Olszewska-Guizzo et al. (2021) suggests that COVID-19 restriction periods may have contributed to a heightened risk of mental health disorders, such as depression and/or reduced cognitive functioning and that green spaces are a way to offset the neuropsychological effects of such periods. In view of this, multiple studies have highlighted increased frequency of green space use during periods of COVID-19 restrictions (Berdejo-Espinola et al., 2021; da Schio et al., 2021; Lu et al., 2021) and suggested this is due to their multifunctionality and their capacity to mitigate some of the negative effects of the COVID-19 pandemic on human health and wellbeing. Changes in frequency of urban green space use during COVID-19 lockdowns show that urban green spaces were discovered by some during the pandemic (new park users; Berdejo-Espinola et al., 2021) and rediscovered by other individuals (previous users, re-engaged and increased their use; Venter et al., 2020). Yet, there is also evidence that some regular users decreased their use of green spaces during the pandemic (Khalilnezhad et al., 2021). Travel restrictions imposed during the pandemic present a useful study system for understanding the temporal and spatial patterns of green space use. These conditions present an opportunity to gain deeper insights into features of green spaces associated with increased use during times of crisis and the self-perceived benefits of visiting green spaces during these times.

Green spaces serve a variety of functions for individuals in the community and perceptions of green spaces can be diverse. Individuals may visit green spaces to access specific features such as walking paths, playgrounds, or ecosystem services such as shade and clean air, offered by that particular space. Yet, not all urban green spaces offer the same types, qualities, and quantities of ecosystem services and not every individual has the same demands for such services. Studies show that biophysical features of green spaces, such as tree and grass cover, vegetation complexity, blue spaces, biodiversity, size, and shape play a role on people's decision to visit green spaces. After conducting a survey of individuals in Portugal, Madureira et al. (2018) found that richness and diversity of plant species was a highly attractive feature of green spaces. However, Shanahan et al. (2017), found that people surveyed in Brisbane and the United Kingdom did not preferentially visit green spaces with higher plant species richness or greater tree cover despite evidence that these features offer the potential for improved nature-based experiences and greater wellbeing benefits. Further, Zhang X. et al. (2020) found that irregularly shaped green spaces were associated with pronounced increases in people using the space for walking. Rey Gozalo et al. (2019) discovered that the size of green spaces was positively correlated with the frequency of walking, exercising, and relaxing in Spain. Location and the facilities within green spaces also play a role in how attractive the green space will be to potential visitors. Anecdotal evidence suggests that frequent green space use for physical activity was greater when the time taken to reach a green space was shorter (Dallimer et al., 2014). Voigt et al. (2014) found that facilities in green spaces that promote relaxation and leisure activities were preferred by residents in Berlin, Germany.

Although previous quantitative research has explored patterns of green space use and features of green spaces associated with frequency of use (Dallimer et al., 2014; Madureira et al., 2018; Dade et al., 2020), we know relatively little about green space characteristics that encourage people to repeatedly visit the *same* green spaces, and we have limited empirical knowledge of the extent to which patterns of green space use and features associated with use change when routine activities abruptly shift; as was the case during the COVID-19 pandemic. In this study we examine green space characteristics and sociodemographic factors associated with changes in frequency of green space use during a period of COVID-19 restrictions in Brisbane, Australia, and explore changes in self-perceived benefits associated with visiting green spaces. We (i) quantify how the frequency of green space use changed during the COVID-19 lockdown, (ii) determine the extent to which these changes were associated with sociodemographic and green space characteristics, (iii) identify changes in self-perceived benefits of using green spaces during the restrictions period, and (iv) analyse the association between changes in self-perceived benefits of using green spaces and biophysical green space characteristics.

MATERIALS AND METHODS

We conducted a survey (n = 1,002) in the Brisbane Local Government Area, Australia, to capture people's urban green space visitation patterns, including change in frequency of, and self-perceived benefits associated with, using urban green spaces before and during a stay-at-home restriction (hereafter "lockdown"). We only considered survey participants that provided information about the specific green spaces they visited *before and during* the pandemic (n = 372). We developed statistical models to identify which sociodemographic characteristics and urban green space biophysical variables were associated with changes in frequency of green space use and/or self-perceived benefits of green space use during COVID-19 compared to before the pandemic.

Study Area and Data Collection

The Brisbane Local Government Area (hereafter "Brisbane") has an estimated human population of 1.27 million residents, approximately 4.9% of Australia's population with a population density of 947 individuals per km^2 (Brisbane City Council, 2021). Brisbane's green space network comprises more than 2,100 urban parks, picnic grounds, pocket parks, riverside spaces, botanic gardens, nature reserves, and beaches. These green spaces are widespread across the city containing both native and non-remnant vegetation cover that provides habitat and connectivity to over 80 different vegetation communities and over 2,300 species of wildlife and native plants (Shanahan et al., 2017; Brisbane City Council, 2020a,b).

Relative to other jurisdictions, Brisbane experienced few COVID-19 infections and related lockdowns during the first year of the pandemic. In Brisbane, the most significant lockdown restrictions were introduced on 23 March 2020 and continued until 2 May 2020 when measures were partially relaxed. During lockdown residents were instructed to work from home where possible; practice social distancing and good hygiene and leave home only for essential trips. The restrictions involved the closure of schools and universities, indoor fitness and sports facilities, and all food, drink, and cultural venues. One of the few reasons people were expressly permitted to leave home was for recreation or physical activity in a public green space. However, these trips were limited to the immediate residential neighbourhood and could be conducted with no more than two people from different households (Australian Government Department of Health, 2020).

Our survey was administered immediately following this lockdown period (survey distributed in June 2020) using a market research company (Q&A).[1] Participants were invited to complete the survey according to four nested stratification criteria that ensured the sample reflected a range of Brisbane's demographic groups, broad socioeconomic spread, and an even spatial distribution across the city. The stratification rules were as follows: (a) an equal number of males and females, (b) an equal number of participants above and below 45 years of age, (c) income quartiles reflecting those of the whole Brisbane population and (d) an even distribution of participants' location of residence across Brisbane (North/South/East/West side of Brisbane). The survey was delivered online by a market research company in accordance with the University of Queensland Human Research Ethics Approval, approval number 2020001073. All participants were at least 18 years old and provided written consent to participate in the survey. The survey asked participants to report whether they had visited an urban green space (a) before and (b) during the COVID-19 lockdown. If they answered yes, they were asked to list up to seven of the green spaces they visited, specifying the name and the frequency of use during both periods (*never, once every 2 weeks, once a week, 2–3 days a week, 4–5 days a week, and 6–7 days a week*). For this study, we used the first-listed green space in each time period (hereafter "nominated green space") since 60.05% of survey respondents listed only one green space, and to simplify the interpretation of the analysis. We then geolocated each nominated green space with reference to a Brisbane City Council spatial dataset of public green spaces (Brisbane City Council, 2020c). We also asked survey participants to provide either their exact address, the address to the nearest 10 houses or the location of the street corner closest to their home, depending on what they felt most comfortable revealing. We used this information to geolocate residences using either the exact address when provided by participants, or the mid-point of a street in other cases.

Dependent Variables

Change in Frequency of Green Space Use During COVID-19 Lockdown

This variable was computed as the difference between frequency of green space use during the COVID-19 lockdown [*never (1), once every 2 weeks (2), once a week (3), 2–3 days a week (4), 4–5 days a week (5) and 6–7 days a week (6)*] and frequency of green space use before lockdown. Given that the frequency of use variable is not continuous we coded change in frequency into three categories, where 0 is *no change* in frequency of green space use during lockdown compared to before the pandemic; 1 is *increased* use of green spaces during the pandemic compared to before lockdown; and 2 is *decreased* use of green spaces during lockdown compared to before the pandemic.

Change in Self-Perceived Benefits of Urban Green Spaces

Participants were asked to report on the extent to which ten common benefits associated using green spaces had increased or decreased in importance during the pandemic related lockdown using a 5-point Likert scale (*1 = much more important, 5 = much less important*). For the analysis, we constructed two new variables named "psychological benefits" and "nature interactions." For "psychological benefits" we constructed mean scale scores taking each participant's mean response across three reasons in the survey questionnaire: reduction of stress, reduction of anxiety, and reduction in depression (*Cronbach's alpha* = 0.94). We followed the same process to construct a "nature interactions" variable using three reasons: connection to nature, appreciation of the environment, and provision of clean air (*Cronbach's alpha* = 0.89). We use the individual Likert variables as outcome variables for physical health benefits, spiritual connection, social interactions, and family interactions.

Independent Variables

Biophysical Factors

Vegetation characteristics (tree cover, grass cover, and foliage high diversity -FHD-) were derived from LiDAR data and other high-resolution imagery at a resolution of 30 m (see Caynes et al., 2016 for more details). We calculated the proportion of *grass* and *tree cover* within each green space using the *raster* and *fasterize* R packages package in R (Ross, 2018; Hijmans, 2021). FHD is a measure of vegetation vertical complexity that accounts for how evenly vegetation is distributed among vertical strata (Caynes et al., 2016). Vegetation (or forest) strata can be composed by three classes of vertical layers. The lowest layer is an herbaceous/shrub story with grasses, herbs, and shrubs. The understory has trees above the shrub layer and below the canopy;

[1] https://qandapanel.com.au/

and the overstory comprises the highest layer of vegetation in a forest, including canopy trees (Berger and Puettmann, 2000). Caynes et al. (2016) separated the FHD data into five discrete height intervals, including very low ($\geq$ 0.15– $<$ 1 m), low ($\geq$ 1– $<$ 2 m), medium ($\geq$ 2– $<$ 5 m), high ($\geq$ 5– $<$ 10 m) and very high vegetation ($\geq$ 10 m). As such, FHD values are high where vegetation is more evenly distributed across the vertical strata and low where vegetation is less evenly distributed. We calculated mean FHD within each green space using the raster and *fasterize* packages in R (Ross, 2018; Hijmans, 2021). Data on *blue spaces* present within a green space was obtained from the Brisbane City Council (2018). This is an indicator variable where a 1 indicates the presence of blue spaces in a green space and a 0 indicates no blue space. The *shape* of a green space was characterised using the shape index defined by McGarigal et al. (2012), calculated by dividing the perimeter of a green space by the minimum perimeter of that green space if the perimeter was rearranged to a maximally compact shape (circle). The shape index is minimum for maximally compact green spaces and increases as green space increases in complexity. For this study, we calculated each urban green space *size* and *shape* using the *landscapemetrics* R package (Hesselbarth et al., 2019).

Distance

We measure the distance from each participant's residence to the nominated urban green space visited before the COVID-19 lockdown by calculating the Euclidean distance between the two points using the *gDistance* function from the *regos* R package (Bivand and Rundel, 2020). In some cases, this will overestimate or underestimate the distance that needs to be travelled to reach the green space, but a detailed integrated transport network dataset was not available.

Facilities

Data on facilities present within each urban green space were obtained from Brisbane City Council (2018). During the COVID-19 lockdown, use of several green space facilities, including playgrounds, dog parks, water fountains, barbeque, and picnic areas were restricted; therefore, we excluded them for our analysis. We included only presence of a carpark and access by public transport. *Presence of a carpark* is a dichotomous variable where 0 indicates no carpark is present at the individual's nominated green space and 1 indicates that the green space does have an attached carpark. *Public transport* is also a dichotomous variable where 0 indicates no public transport access at the individual's nominated green space and 1 indicates public transport is available at the green space.

Change of Urban Green Space Visited During Lockdown

We operationalised this variable by comparing the first nominated green space visited before and during lockdown. We then computed it by creating a dichotomous variable where 0 indicates no change and 1 indicates that the individual changed the green space visited during COVID-19 compared to before the pandemic related lockdown.

Statistical Analysis

We first conducted a series of descriptive analyses to evaluate the differences between urban green spaces visited before and during lockdown. **Tables 1, 2** summarize descriptive statistics for each of the measures included in the regression analyses (**Supplementary Data Sheets 1, 2** respectively).

Second, we estimated a series of regression models to address our research questions. The first of these models is a multinomial logistic regression analysis to determine how the frequency of green space use changed during the COVID-19 lockdown, and the extent to which this was associated with sociodemographic and green space characteristics. The dependent variable in this model is change in frequency of use of green space during the COVID-19 lockdown compared to before, coded into three categories, where 0 = *no change*, 1 = *increased* use of green space during lockdown, and 2 = *decreased* use of green space during lockdown. The category "*no change* in frequency of green space use" is used as the reference category in the multinomial regression. Our selection of *independent* variables is informed by the results of previous research on people's perceptions and use of green spaces (Ode Sang et al., 2016; Braçe et al., 2021), and also scholarship on the frequency of green space use during COVID-19 (Uchiyama and Kohsaka, 2020; da Schio et al., 2021; Lu et al., 2021). Studies consistently identify *gender, age,* and *income* as associated with their perception of greens spaces and frequency of use of green spaces not only in "non-pandemic" times, but also during periods of COVID-19 restrictions (Braçe et al., 2021). The influence of green space characteristics on changes in the frequency of green space use during the COVID-19 pandemic is yet to be examined in depth. However, a study by Lu et al. (2021) examining use of urban green space in Asian cities during the pandemic found that during COVID-19 restrictions, urban residents preferred large nature parks to smaller urban parks and also visited green spaces closer to the city centre. Drawing on this limited research and previous research examining green space characteristics that

TABLE 1 | Characteristics of the 372 study participants that responded to items regarding green space visited and reasons for visiting green spaces during the COVID-19 lockdown.

Variables	*n*	Median or % (SD)	Min-Max
Gender			
Males	194	51.6	
Females	178	48.3	
Age	372	43 (17.7)	19–84
Income (AUD $/year)	372	102,000 (14,000)	0–104,000
Change of green space visited during lockdown			
Yes	124	33.3	
No	248	66.6	
Change in frequency of green space use			
No change	143	38.4	
Increased	150	40.3	
Decreased	79	21	

TABLE 2 | Characteristics of urban green spaces visited before and during the COVID-19 lockdown (*n* = 300).

Variable	Before			During	
	n	Median/n (*SD*)	Min-max	Median/n (*SD*)	Min-max
Distance to green space (metres)	300	1362.05 (4605.78)	81.86–25368.94	1269.91 (4651.94)	81.82–27190.37
Size of green space (hectares)	300	9.79 (222)	0.11–1413.93	9.55 (251.2)	0.05–1413.93
Shape of green space	300	1.72 (1.29)	1.06–11.79	1.65 (1.02)	1.08–11.79
Grass cover	300	22.29 (20.5)	0–100	24.88 (20.5)	0–100
FHD	300	0.80 (0.11)	0.53–1.13	0.80 (0.11)	0.58–1.13
Blue spaces	300	154		144	

influence use during "non-pandemic" times (for example see Abkar et al., 2010; Dade et al., 2020) we selected the following green space characteristics as independent variables: shape of green space, size of green space, presence of blue spaces, tree and grass cover, FHD, presence of car park, and presence of public transport node. We conducted Spearman's correlations and VIF statistics. All VIFs were below 3.6 suggesting multicollinearity would not be problematic, and Spearman's correlations were below 0.43 except in the case of tree and grass cover which returned a value of −0.66 (**Supplementary Table 1**). Tree cover was therefore excluded from the model, while *grass cover* of the green space visited before lockdown, *FHD* of the green space visited before lockdown, the presence of *blue spaces*, *size* of the green space visited before lockdown, and *shape* of the green space visited before lockdown were retained to represent biophysical features of the green space. To ensure model parsimony [lowest Akaike Information Criterion (AIC) (Burnham and Anderson, 2002)] and the selection of the best fitting model we added variables in a stepwise (hierarchical) fashion. Refer to **Supplementary Table 2** for details on the stepwise models not presented in the main text. Model 1 contains only the sociodemographic characteristics; Model 2 is our final model including independent variables that are both informed by the literature and ensure model parsimony. We used the *nnet* R package (Venables and Ripley, 2002) to run multinomial logit regressions. The analytic sample included survey participants with responses for items regarding urban green space engagement both before and during COVID-19 lockdowns (*n* = 300, **Supplementary Data Sheet 2**).

The second stage of our empirical analyses comprises a suite of six regression models (Models 3–8) that examine the association between changes in the importance of self-perceived benefits of green spaces during the pandemic and particular biophysical green space characteristics. Models 3–8 investigate changes in the importance of specific benefits associated with visiting green spaces including: *nature interactions; psychological health benefits; physical activity; spiritual connection; and social and family interactions*. Each model includes independent variables representing *gender, age, income, size* of green space visited during lockdown, *shape* of green space visited during lockdown, *grass cover* of green space visited during lockdown, presence of *blue spaces* in the green space visited during lockdown, and *FHD* of green space visited during lockdown. Variables were selected based on previous research showing that size, shape, FHD, and grass cover can influence reasons for visiting green spaces (Dade et al., 2020). Although Dade et al. (2020) also found that facilities influenced reasons for attending green spaces, we did not include facilities in these models given that all facilities at public green spaces were closed during the COVID-19 lockdown. Models 3 and 4 are generalised linear regression models (GLM) assuming a gamma distribution because both the "nature interactions" (Model 3) and "psychological health benefits" (Model 4) dependent variables include positive-only values, and the error distributions are right skewed. The dependent variables "physical activity" (Model 5), "spiritual connection" (Model 6), "social interactions" (Model 7), and "family interactions" (Model 8) are ordinal variables containing five categories; thus, we estimate ordered logistic regression models for these data. We used the *MASS* R package to run the ordered logistic regressions (Venables and Ripley, 2002). The analytic sample for Models 3–8 include all survey participants who responded to items regarding green space visited and reasons for visiting green spaces *during* COVID-19 lockdown (*n* = 372, **Supplementary Data Sheet 1**).

All statistical models were evaluated using goodness of fit tests. For multinomial logistic regression models, we used a Hosmer–Lemeshow Test from the *generalhoslem* R package (Jay, 2019), for generalised linear models we used a Likelihood Ratio Test from the *lmtest* R package (Zeileis and Hothorn, 2002), and for the ordered logistic regression we used a Lipsitz Test from the *generalhoslem* R package. We calculated models' R squared using the *rsq* R package (Zhang, 2021). All models were assessed for and met normality of residuals. All data analyses were carried out with R Studio V1.2 (RStudio Team, 2020).

RESULTS

Descriptive Analysis

The sample comprised 372 individuals of which 48.3% were females, and the average age was 43 (**Table 1**). Of the 372 individuals, 78.8% either increased or did not change their frequency of green space use, while 21.2% decreased their visits to green spaces. There was a great deal of flux in which green spaces were visited, with 33.3% of individuals visiting a different green space during the lockdown than before.

Although in aggregate there appears to be little change between the characteristics of green spaces used before and during the pandemic (**Table 2**), examining individual cases suggests several important differences in the characteristics of green spaces used during COVID-19 compared to before at the individual level. For example, before COVID-19 lockdown, a large section of the population visited green spaces two to 11 times farther away from their residences than the green space visited during lockdown. Conversely, during the COVID-19 lockdown, fewer individuals visited green spaces two to five times farther away from their residences than the green space visited before lockdown. Although a large section of the sample population (66%) visited green spaces of approximately the same size before and during the COVID-19 lockdown. During lockdown, a fraction of the sample population (9 and 9.6%) visited green spaces 10 or more times smaller and larger, (respectively), than the green space visited before lockdown. Urban green space shape patterns did not appreciably change during the lockdown (**Table 2**). There is a slight upward shift in grass cover from before [Interquartile Range (IQR) = 22.29–50.6] to during (IQR = 24.8–52.0) the lockdown. Although some individuals (13.6%) visited green spaces with less grass cover during lockdown (from 25 to 50% cover to 0%) many individuals (20.3%) visited green spaces with larger amounts of grass cover when compared to the green space visited before lockdown. Foliage height diversity followed a similar trend, with 16 and 18% of the sample population visiting green spaces with lower and higher FHD indexes (**Table 2**).

Change in Frequency of Green Space Use During COVID-19 Lockdown

A summary of the associations between dependent and independent variables of Models 1 and 2 are shown in **Table 3**. Detailed results of the multinomial logistic regressions are displayed in **Table 4** and are presented as relative risk ratios (RRR). RRR are the exponentiated regression coefficients and can be interpreted as the relative likelihood of an event occurring between two groups (e.g., change in frequency of use of green spaces during lockdown) given exposure to the variable of interest (e.g., shape of a green space) (Simon, 2001). The category "*no change* in frequency of green space use" is used as the reference category in the multinomial regression. Hosmer-Lemeshow goodness of fit tests indicates a significant improvement in model fit between the base demographic model (Model 1 $p < 0.0361$) and our final model (Model 2 $p < 0.6949$). The test score for Model 2 is well above the 0.05 threshold indicating the model is correct. It is also approaching one; therefore, demonstrating a better fit.

Results of Model 2 demonstrate that individuals were more likely to *increase* their use of green space during lockdown compared to before if they were female compared to male (RRR = 2.413, $p < 0.01$) and younger compared to older (RRR = 0.970, $p < 0.01$). Individuals who visited the same green space during lockdown compared to the green space visited before the lockdown tended to *increase* their frequency of use (RRR = 0.570, $p < 0.01$). Urban green space characteristics also played a role. Individuals were more likely to *increase* their frequency of green space use during the lockdown compared to before the lockdown if the green space they visited had a carpark (RRR = 1.217, $p < 0.01$), was accessible by public transport (RRR = 1.289, $p < 0.01$), contained a blue space (RRR = 1.307, $p < 0.01$), and/or had lower FHD –vegetation is less evenly distributed across vertical layers– compared to green spaces with higher FHD (RRR = 0.211, $p < 0.01$). Individuals were more likely to *decrease* their frequency of green space use during the lockdown compared to before if they were female (RRR = 1.179, $p < 0.01$), or changed their nominated green space during the lockdown compared to before (RRR = 1.437, $p < 0.01$), individuals were also more likely to reduce their frequency of green space use during the lockdown, compared to before, if the green space they visited did not have a carpark (RRR = 0.913, $p < 0.01$), was not accessible by public transport (RRR = 0.760, $p < 0.01$), and had higher FHD –vegetation is more evenly distributed across the vertical structure– compared to green spaces with lower FHD (RRR = 2.933, $p < 0.01$).

Changes in Self-Perceived Benefits of Urban Green Space Use

In Models 3–8 we examine changes in self-perceived benefits associated with urban green space use during the pandemic compared to before. A summary of the associations between dependent and independent variables of Models 3–8 are shown on **Table 3**. Model 3 shows that being male ($b = -0.01$, $p < 0.05$) and older ($b \leq 0.001$, $p < 0.001$) was significantly associated with reporting an increase in the importance of urban green spaces for nature interactions during the lockdown. Characteristics of the green space visited during the lockdown did not influence change in the importance of nature interactions. Model 3 accounts for 11.2% of variance in the outcome variable and was significant (Likelihood ratio test, $p < 0.001$). Similarly, Model 4 indicates that being male ($b = -0.001$, $p < 0.01$) and older than 43 ($b = 0.001$, $p < 0.001$) were significantly associated with reporting an increase in the importance of urban green spaces for psychological benefits during the lockdown. Model 4 accounts for 16.3% of the variance in the outcome variable, which was significant (Likelihood ratio test $p < 0.001$). Models 5–8 are ordered logistic regression models. Lipsitz goodness of fit tests for all models returned a p-value above the 0.05 threshold indicating that all models satisfy the proportional odds assumption and are correctly specified. Results of Model 5 reveal that younger individuals are more likely than older people to report increases in the importance of green spaces for physical activity during the lockdown than before (RRR = 0.983, $p < 0.01$). Model 6 indicates that the odds of reporting increased importance of green spaces for spiritual connection is higher for females compared to males (RRR = 1.516, $p < 0.1$), younger individuals compared to older individuals (RRR = 0.974, $p < 0.01$), and higher income earners (RRR = 1.071, $p < 0.05$). Using a smaller green space with greater shape complexity is also associated with higher odds of reporting increases in the importance of green spaces for spiritual connection during the lockdown (RRR = 0.999, $p < 0.1$; RRR = 1.211, $p < 0.1$). Results of Model 7 show the odds of reporting an increase in the importance of green spaces for social interactions during the COVID-19 pandemic was significantly higher for female compared to males (RRR = 1.461, $p < 0.1$), younger individuals (RRR = 0.983, $p < 0.01$), and higher income earners (RRR = 1.068, $p < 0.05$). Model 8 reveals that the odds

TABLE 3 | Significant associations between dependent and independent variables of eight regression models including sociodemographic and green space characteristics of green spaces used before (Model 1–2) and during (Model 3–8) a COVID-19 lockdown.

Independent variables	Dependent variables							
	Model 1	**Model 2**	**Model 3**	**Model 4**	**Model 5**	**Model 6**	**Model 7**	**Model 8**
	Change in frequency of use	Change in frequency of use	Nature interactions	Psychological health benefits	Physical activity	Spiritual connection	Social interaction	Family interactions
Gender—females	+	±	–	–		+	+	+
Age	–	–	+	+	–	–	–	–
Income						+	+	+
Change the green space visited		±						
Distance to green space								
Carpark		±						
Public transport		±						
Size of green space						–		
Shape of green space						+	+	
Grass cover								
FHD		±						
Blue spaces		+						

For Models 1-2: "+" denotes an increase and "–" denotes a decrease; and for Models 3–8: "+" denotes a positive association and "–" denotes a negative association. Gray cells represent variables that were not included in the regression.

TABLE 4 | Change in frequency of use of urban green spaces during the COVID-19 lockdown explained by sociodemographic and green space characteristics variables.

Independent variable	RRR		SE		95%CI			
	Increased use	Decreased use	Increased use	Decreased use	Increased use		Decreased use	
Model 1								
Intercept	2.243	0.527	0.568	0.694	−0.305	1.921	−2.000	0.717
Gender—females	2.430**	1.238	0.275	0.315	0.349	1.426	−0.403	0.830
Age	0.971**	0.990	0.008	0.009	−0.045	−0.013	−0.027	0.007
Income	1.003	1.035	0.037	0.045	−0.070	0.076	−0.053	0.122
Hosmer-Lemeshow Test[¥]		*p-value = 0.0361*						
AIC		*623.295*						
Model 2								
Intercept	7.345**	0.224**	0.006	0.010	1.982	2.005	−1.516	−1.478
Gender—females	2.413**	1.179**	0.007	0.008	−0.086	0.089	0.149	0.017
Age	0.970**	0.990	0.006	0.007	−0.004	−0.001	−0.024	<0.001
Income	1.000	1.039	0.033	0.038	−0.064	0.006	−0.034	0.011
Change the green space visited	0.570**	1.437**	0.003	0.011	−0.056	−<0.001	0.340	0.038
Distance to green space[§]	1.000	1.000	<0.001	<0.001	−<0.001	< 0.001	−<0.001	<0.001
Carpark[§]	1.217**	0.913**	0.003	0.007	0.019	0.020	−0.105	−0.007
Public transport [§]	1.289**	0.760**	0.002	0.001	0.025	0.025	−0.275	−0.027
Shape of green space[§]	1.047	0.928	0.102	0.134	−0.015	0.024	−0.337	−0.018
Size of green space	1.000	0.999	0.001	0.001	−<0.001	<0.001	−0.003	<0.001
FHD[§]	0.211**	2.993**	0.006	0.011	−1.568	−1.543	1.075	1.117
Grass cover[§]	1.004	1.005	0.006	0.007	−<0.001	0.001	−0.008	0.001
Blue spaces[§]	1.307**	0.988	0.012	0.021	0.024	0.029	0.052	0.002
Hosmer-Lemeshow Test[¥]		*p-value = 0.6949*						
AIC		*640.16*						

[§]*Urban green space characteristics are from before the COVID-19 lockdown (n = 300).* [¥]*Hosmer-Lemeshow Test with higher p-values indicate better model fit (Fagerland and Hosmer, 2012). ^p < 0.10, *p < 0.05, **p < 0.01. RRR = relative risk ratios. SE = standard error.*

of reporting increases in the importance of green spaces for family interactions were higher for females compared to males (RRR = 1.411, $p < 0.1$), younger people compared to older individuals (RRR = 0.982, $p < 0.01$), higher income earners

(RRR = 1.062, $p < 0.05$), and those who used green spaces with more complex shapes during the lockdown compared to those who visited more compact green spaces (RRR = 1.265, $p < 0.05$).

DISCUSSION

In this study we discovered that changes in the frequency of green space use and the importance of green spaces for self-perceived benefits use during a COVID-19 lockdown were associated with a number of individual characteristics and green space characteristics. In sum, our findings suggest four key takeaway points.

Our first key finding is that frequency of green space use increased during COVID-19 for some individuals but decreased for others. This pattern of change in frequency of green space use was particularly evident among females in the sample and suggests that the impact of COVID-19 restrictions on changes in frequency of green space use among females may have been moderated by green space characteristics. Our results provide support for those reported by Braçe et al. (2021) who found gender differences in perceptions of green space characteristics, with females attributing greater importance to characteristics, such as pleasant views, playgrounds, lightning, and safety than their male counterparts.

Secondly, individuals who did not change the park visited during the lockdown period were more likely to increase their frequency of visits. Increasing frequency of use may be associated with individuals having more free time and using green spaces for a greater variety of activities during COVID-19 when other facilities such as gyms and cafes were unavailable. Also, being familiar with the available features and other people who use the green space has the potential to increase opportunities for social, physical, and psychological benefits. Increased frequency of use was also associated with specific features of green spaces visited pre-lockdown; in particular the presence of a blue space. Recent research suggests that more frequent visits to blue spaces are associated with positive wellbeing and lower rates of mental distress Abkar et al. (2010) and White et al. (2021) suggests that water in green spaces is one of the most important biophysical factors that contribute to individuals' positive mood change. Thus, the presence of blue spaces might enhance the odds of individuals increasing their frequency of green space use especially during times of stress.

Individuals who went to a different green space during lockdown were more likely to report a decrease in visitation compared to before the lockdown. This could be explained by the routine activity theory (Cohen and Felson, 1979), which posits that changes in the structure of the patterns of daily activity could explain other events. During COVID-19 lockdowns, people's daily life was forced to drastically change because important urban nodes or places where daily activities are usually carried out were closed. Therefore, routine daily life during lockdown was characterised by reduced mobility, prolonged stays at home, telework or job losses, disruptions in social relationships, and declines in physical activity levels; all with implications for mental wellbeing (Ogden, 2020; Salari et al., 2020; Biroli et al., 2021; Mckeown et al., 2021; Mouratidis, 2021). One potential explanation for why individuals visited a different green space during the lockdown is that green spaces took on some of the functions of other nodes during the lockdown (for example gyms, cafes). That is, individuals visited green spaces with different characteristics, to satisfy unmet demands resulting from the restrictions. However, it is also plausible that the new green space visited during lockdown did not fulfil individuals' needs and may have caused them to decrease their frequency of use. For example, we found that green spaces with a higher proportion of canopy cover among forest strata (FHD), which tend to be located in the outskirts of the city, were associated with a remarkable decrease in visits during lockdown. In contrast, green spaces with fewer vegetation vertical layers (which are more characteristic of neighbourhood parks) were associated with increases in use. Our results may suggest that local green spaces and in closer proximity to people's homes can act as nature-based solution by fulfiling needs that are not necessarily related to nature interactions during times of crisis.

Our third finding is that people travelled to green spaces two to 11 times closer to their home during the lockdown compared to before, suggesting that the mobility restrictions imposed to stop the spread of COVID-19 may have changed individuals' daily activity patterns and further supporting a role for the routine activity theory in explaining the changes. Therefore, visiting urban green spaces with different characteristics may have served as a pathway to accommodate people's new daily activity patterns and needs resulting from the COVID-19 restrictions. However, the opportunity to access the services provided by either the same or a different green space may also depend on the configuration of the urban landscape and individuals' access to green spaces. Proximity was not the only indicator of accessibility that was associated with increased frequency of green space use during the lockdown. Individuals who visited green spaces with *carparks* and/or *public transport* nodes were also more likely to increase their frequency of green space use during lockdown. Although Brisbane residents have excellent accessibility to green spaces, relative to many other capital city residents (Berdejo-Espinola et al., 2021) this finding illustrates that ensuring equitable access across all sociodemographic groups and neighbourhood settings is pivotal for facilitating green space use to satisfy unmet needs during times of crisis.

Finally, our results show that the importance of self-perceived benefits associated with using green spaces changed markedly during the COVID-19 lockdown, and that these changes were associated with sociodemographic variables and to a lesser extent with green space characteristics. Despite the closure of facilities within green spaces, such a playgrounds, benches, and sporting features, on average, individuals reported increased importance of green spaces for their cultural ecosystem services. This reinforces the importance of biophysical features of green spaces in everyday urban life. We found that males and older individuals were more likely to express greater increases than females and younger people in the importance of green space for nature interactions and mental health benefits during the COVID-19 lockdown compared to before the pandemic. This might be related to changes in routine activity patterns for males, including teleworking, looking after children, and reallocation of household responsibilities that posed new stressors in their daily life (Douglas et al., 2020; Biroli et al., 2021). While the importance of urban green spaces for mental wellbeing is well established in the literature (Keniger et al., 2013; Zhang J. et al., 2020), these benefits have been articulated more strongly by females in times of crisis (Burnett et al., 2021; Gastelum-Strozzi et al., 2021; Lopez et al., 2021). In fact, Gastelum-Strozzi et al. (2021) have shown that males in Mexico City indicated feeling remarkably fewer effects of the pandemic on mental health than females. It may be that for males living in Brisbane, utilising green spaces for mental wellbeing has emerged as a condition of the COVID-19 pandemic after making other places for social interactions and physical health unavailable. In contrast, we found that females were more likely than males to report increased importance of green spaces for spiritual, social, and family interactions. This reflects research conducted by Burnett et al. (2021) in the United Kingdom, who found that females who reduced their use of green spaces during the pandemic were more likely to agree that they missed interacting with others in green spaces than males. Future research could explore the idiosyncrasies across sexes.

The shape of green spaces also emerged as an important feature associated with changes in benefits associated with green space use. Individuals using irregularly shaped green spaces were more likely to report increased importance of the green space for social interactions during the lockdown. Scholarship on social cognition suggests that social gazing—visual behavior between two or more individuals (Kleinke, 1986)—is one form to communicate and an important component of social interaction (Emery, 2000). Thus, spending time in less compact green spaces might help increase the number of everyday visual encounters where people can at least see others while socially distancing and stimulating a sense of social connection even when physical distancing is enforced. Further, individuals using irregularly shaped green spaces were more likely to report increased importance in green spaces for spiritual connection during the COVID-19 pandemic. Across Brisbane, the shape of urban green spaces tend to be relatively compact, with a mean shape index of 1.45 [compact shape = 1 according to McGarigal et al. (2012)], meaning that irregularly shaped green spaces are less common. Thus, the uniqueness of compact green spaces may facilitate the development of a sense of place attachment—the emotional bond between person and place (Scannell and Gifford, 2010). Stedman (2003) suggests that individuals do not become directly attached to the physical features of a green space, but rather to the meaning that those features represent. Thus, attachment to a green space could be related to the attachment to the experiences individuals can have in a green space, perhaps nature experiences (for example, birdwatching), or to the attachment to the social interactions that the green space affords them. Unique identifiers of the green space territory, such as a unique shape are important to facilitate opportunities for social connection, particularly during stressful times. In this context, future pandemic-resilient urban planning and policy might consider the size and shape of green spaces to ensure safe access to green spaces with appropriate social distancing.

While this study extended current knowledge on urban green spaces and their use during the COVID-19 pandemic it is not without limitations. We note that the sample is relatively small, and caution should be used when generalising the findings to other settings. Further, the study is conducted in Brisbane, Australia which had been relatively protected from COVID-19 compared to other regions during the early phase of the pandemic. Another limitation is that we asked individuals to report about the green spaces visited before and during a COVID-19 lockdown and did not account for the influence of household dynamics and how other members of the household may affect green space visitation. We encourage new research to explore this avenue. Lastly, our data are retrospective, given that participants reported their frequency of use and perceptions of the benefits of green spaces before and during the restrictions period 30–90 days after their use. This means that there may be some inaccuracy in people's reports due to the memory recall errors, however, we expect recall bias to be limited due to the relatively short reference period (Ayhan and Isiksal, 2004).

Our findings suggest a number of areas for future research and considerations for policy and practice. First, they highlight the need to ensure accessibility to urban green spaces with appropriate ecosystem services for all sociodemographic groups and all geographic areas, before, during, and after disasters/collective crises as they are important multifunctional sites that can serve as a nature-based coping mechanism for communities and individuals (Berdejo-Espinola et al., 2021). Accessibility to green spaces must consider not only proximity to, and availability of green spaces, but also modes of access, such as car parking, and public, and active transport (the latter not considered in this study). Yet, making green spaces available is only the first step; ensuring that they are matched to the needs of the population is also vital. Our findings also suggest that features found to be associated with mental health and wellbeing benefits such as blue spaces may facilitate higher usage during periods of crises. Thus, the presence of blue spaces may have the potential to benefit communities. Further, the complexity of green space shape may be important for facilitating place attachment and supporting the social function of green spaces. Planning and co-designing crisis-resilient urban areas while considering the key individual and community factors might lead to more accessible, functional urban green spaces.

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.

ETHICS STATEMENT

The studies involving human participants were reviewed and approved by The University of Queensland Human Research

Ethics Committee. The patients/participants provided their written informed consent to participate in this study.

AUTHOR CONTRIBUTIONS

VB-E and RZ: conceptualisation, formal analysis, and writing original draft. VB-E, RZ, RF, and AS-C: methodology. VB-E, RZ, AS-C, JR, and RF: writing-review and editing. All authors contributed to the article and approved the submitted version.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fevo.2022.804443/full#supplementary-material

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Signs of Urban Evolution? Morpho-Functional Traits Co-variation Along a Nature-Urban Gradient in a Chagas Disease Vector

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Environmental change (i.e., urbanization) impacts species in contrasting ways, with some species experiencing benefits given their way of life (i.e., blood-sucking insects). How these species respond to such change is not well understood and for species involved in human diseases, this "how" question is particularly important. Most Triatominae bug species inhabit tropical and subtropical forests where their vertebrate hosts' temporal abundance depends on climate seasonality. However, in human encroached landscapes, triatomines can benefit from resource stability which may lead to adaptive phenotypic change to track novel hosts. We tested for an association between different landscapes and morpho-functional traits linked to sensory, motion, and feeding functions in *Triatoma dimidiata* and compared fecundity (i.e., number of eggs) in each landscape as a proxy of fitness. Using geometric and traditional morphometric tools, we predicted a morphological simplification in bugs inhabiting urbanized areas. While wing morphology or proboscis were not influenced by landscape class, the opposite occurred for thorax morphology and number of sensilla. Wing and thorax morphology did not covary under modified landscape scenarios, yet we detected a morpho-functional convergence for thorax size and antennal phenotype in both sexes, with a simplification trend, from nature to urban settings. Given no fecundity differences across landscapes, there is no potential reproductive costs. Moreover, the convergence of thorax size and antennal phenotype suggests differences in flight/locomotion performance and host/environment perception, as a possible adaptive response to relaxed selective pressures of the bug's native habitat. These results imply that *T. dimidiata* could be adapting to urbanized areas.

Keywords: ***Triatoma dimidiata*, adaptation, urbanization, domiciliation, phenotypic variation, traditional morphometric, geometric morphometric**

INTRODUCTION

Urbanization is an important selective driver that has given rise to phenotypes finely co-adapted to novel environments (Miles et al., 2021). Urbanized species have taken advantage of human-provided resources and some examples include arthropod vectors of diseases which have successfully "tracked" human evolution by either making use of their phenotypic plasticity (i.e., the ability of a single genotype to give rise to different phenotypes after being exposed to distinct environments) and/or evolving new traits (Beaty et al., 2016; Fouet et al., 2018; Suesdek, 2019). An example of such plasticity is the overwintering ability of *Aedes albopictus* which has allowed it to colonize northern (and, thus, colder) European areas in less than three decades (Wilke et al., 2020). However, we are unaware of how much phenotypic plasticity (i.e., the capacity of species to respond to environmental variation by producing different phenotypes) and evolution (i.e., change in the genetic composition of a population over successive generations) of new traits have contributed to explain such arthropod vector success in urban contexts. In any case, since most of this knowledge comes from work with mosquitoes (e.g., Beaty et al., 2016; Fouet et al., 2018), it is unclear whether some other vector taxa have gone through similar evolutionary or microevolutionary (i.e., evolution within and among populations) responses. Gathering vector-wide information is key as zoonotic diseases are on the rise (Allen et al., 2017; Magouras et al., 2020), and tracking the evolution of underlying vectors may be just one more angle to think about the spread of such diseases.

Most species of triatomine (Hemiptera: Reduviidae: Triatominae) bugs are specialized in hematophagy which involves multiple adaptations associated with host localization and blood suction (Weirauch, 2021). These "kissing bugs" are vectors of the parasite *Trypanosoma cruzi,* the etiological agent of Chagas disease (CD) (Noireau et al., 2009). Tropical and subtropical forest are the primary vectors' habitat, yet the continued change of these landscapes by anthropogenic activities (Ellis et al., 2017) have promoted triatomine transient and seasonal invasion to human rural dwellings (Dumonteil et al., 2002, 2007; Nouvellet et al., 2011). Rural infestation by triatomines is the leading cause of CD epidemiology, which causes more than 15,000 human deaths and 40,000 new infections each year (Rassi et al., 2010). However, in the last two decades, more information is emerging that shows an "urbanization" scenario for CD for several Latin American countries (Moncayo and Silveira, 2009; Schmunis and Yadon, 2010; Sosa, 2010). Related to this, several studies have shown that at least 14 species of kissing bugs have invaded and colonized urban and peri-urban dwellings, increasing the frequency of vectorial infections in humans (Levy et al., 2006; Briceño-León, 2009; Provecho et al., 2014). Furthermore, records of urban infestation date back to 50 years for several of these countries, suggesting that it is not an emerging problem but an underestimated one (Gaspe et al., 2020).

Triatomines' sources of blood have changed along with landscape modifications: a reduction of many blood-source vertebrates' species from original forests (Cavada et al., 2019) and an increase of domestic hosts all-year-round from anthropic landscapes, including rural and urban settings (Ordóñez-Krasnowski et al., 2020). A shift in hosts would select for distinct functional and performance traits related to dispersal, foraging, and colonization in triatomines. Despite the shortage of studies on the effect of land-use change and urbanization on triatomine function and performance, these anthropogenic phenomena are repeatedly claimed as selective forces propelling the evolution of intraspecific morphological variation in triatomines (Bustamante et al., 2004; Dorn et al., 2007; Aldana et al., 2011).

The ability to detect and feed on vertebrate hosts by triatomines is mediated by their morphological traits (Dujardin et al., 2009). One first set of traits related to host detection is flight as this is largely involved in the infestation or reinfestation patterns in Triatominae (Ceballos et al., 2005). Some bionomic markers of this function are wing size, shape, and thorax (Dujardin et al., 2007; Gaspe et al., 2012; Hernández et al., 2015). Antennal sensilla are one second set of traits also under current selection (Dujardin et al., 1999; Schofield et al., 1999; Carbajal de la Fuente and Catalá, 2002), as they are related to dispersion and invasion ability to new habitats, as a signature of biotope-related selection (Catalá et al., 2005; Arroyo et al., 2007; Hernández et al., 2011). Finally, proboscis length is another functional trait that seems to have evolved in response to a wild-urban environmental gradient (Eggenberger et al., 2019). These flight- and feeding-related traits have led triatomine researchers to speculate about their covariation with habitat types (Dujardin et al., 1999; Schofield et al., 1999). Furthermore, wing morphology, antennal phenotypes, or the combination of both trait groups have been used to provide evidence related to "domiciliation" in triatomine bugs (Borges et al., 2005; Schachter-Broide et al., 2009; Hernández et al., 2011; Gaspe et al., 2012; Villacís et al., 2014). However, no studies have been carried out to understand how these and other functional traits have responded simultaneously to different landscapes, which has remained as one major question in the study of evolution of triatomine domiciliation (Flores-Ferrer et al., 2018). Given this, we have asked whether there is an association between the morpho-functional traits that underlie flight- and feeding-related traits and landscape type in triatomines. Having *Triatoma dimidiata s. l* as a study species and using geometric and traditional morphometric approaches, we hypothesized that morpho-functional traits associated with dispersal capacity and foraging behavior would reflect evolutionary responses to different landscapes in a nature-urban gradient. Furthermore, if such phenotypic change has an adaptative basis, we predicted similar fitness outcomes in different landscapes. According to this, we predict that: (a) the morphology related to flight and feeding will show patterns of simplification in urban habitats compared to non-urban habitats; and, (b) equal fitness payoffs across different nature-urban settings.

MATERIALS AND METHODS

Study Area

The study was conducted in the Yucatan state, located at the north of the Yucatan Peninsula (**Figure 1**). *T. dimidiata* were collected from eight sites corresponding to the northeast (Mérida

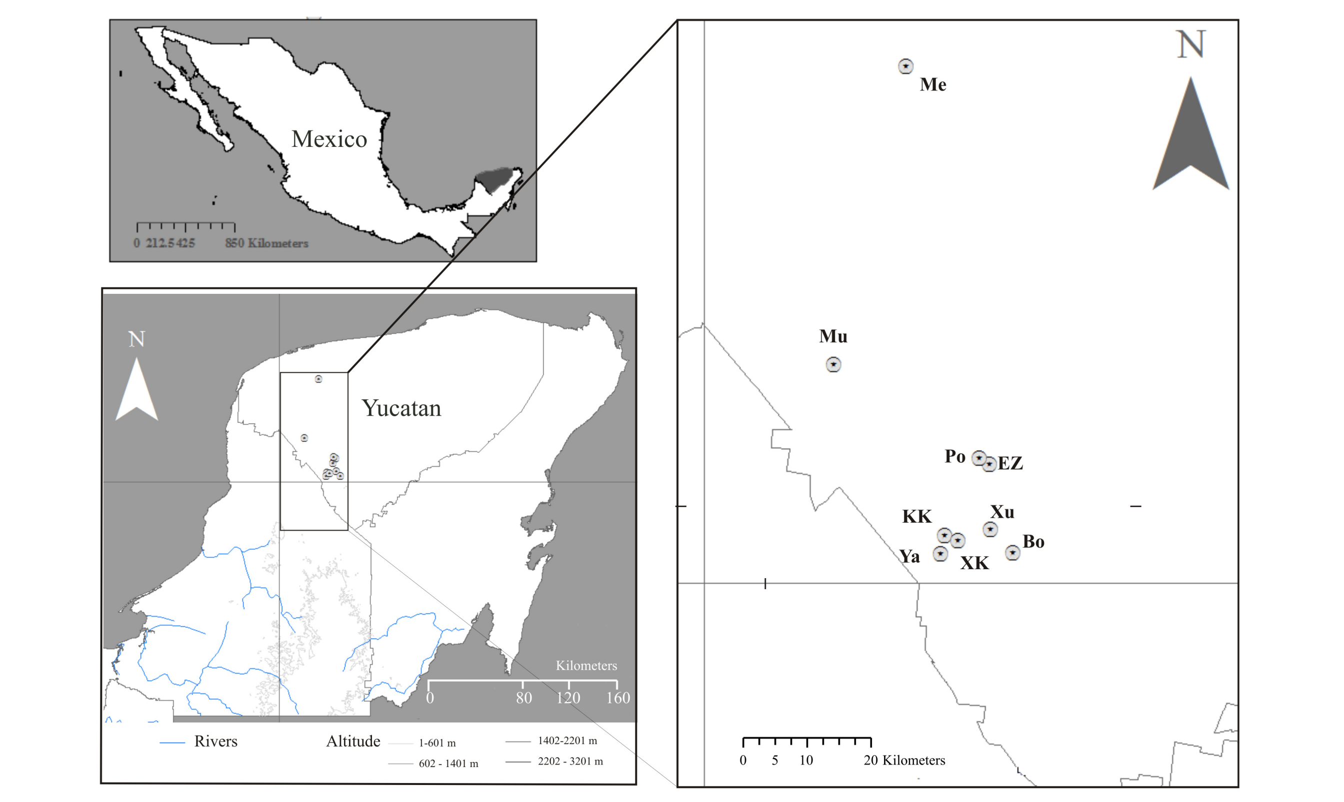

FIGURE 1 | Study area in Mexico and Yucatan State Collection sites are indicated: Urban, Mérida City (Me); Homegarden ("Solares Mayas": Emiliano Zapata (EZ) and Xul (Xu); Median subdeciduous tropical forest: Kaxil Kiuic (KK) and Yaxhachén (Ya); Secondary vegetation: X'kobenhaltun (XK), Bombahaltún (Bo), and Potoit (Po).

city) and southwest of Yucatan State (**Figure 1** and **Table 1**). All sites are located at an elevation of <96 m. The collection sites were grouped into four landscape types: Urban, Homegarden in rural communities ("Solares Mayas"), continuous tropical forest, and secondary forest. The Urban (U) class is represented by Merida City, a large commercial hub for southern and southeastern Mexico (Biles and Lemberg, 2020). Until 2010, the city had grown to occupy an area from 15,944 to 27,027 ha (Secretaría de Desarrollo Urbano y Medio Ambiente [SEDUMA], 2018) and currently concentrates 43% of the Yucatan state's population (García-Gil et al., 2010). It is estimated that between 1980 and 2015, population increased from 400.142 to 832.651 (Biles and Lemberg, 2020). *T. dimidiata* is the only Triatominae species in this region, with values of city infestation (percentage of houses with vector presence) and infection with *T. cruzi* of 38 and 48%, respectively (Guzmán-Tapia et al., 2007). According to data from the Secretaria de Salud de Yucatan (SSY, period 2012–2105), the presence of *T. dimidiata* nymphs has been documented in human housing all over the state, including Mérida city (González-Martínez, 2018). Mérida is organized in eight districts, and our collections were carried out in four of them: west, northwest, east, and south. The last one is a zone with active and significant circulation of *T. cruzi*, with evidence of seroprevalence in people, domestic dogs (Jiménez-Coello et al., 2010, 2015), marsupials and synanthropic rodents (Panti-May et al., 2017; Ucan-Euan et al., 2019).

All collections of Triatomine from southwest of Yucatan state were carried out in a region with a long history of ancient Mayan settlements (McAnany, 2016). This region is the central agricultural zone of the state after transformation from tropical dry forests (Klepeis and Vance, 2003; Manson and Evans, 2007; Hernández-Stefanoni et al., 2014).

The "Solares" (S) are essential in the cultural traditions and family economy of contemporary Mayan populations. They are environmental units that includes the house-room and spaces destined for agricultural production to subsistence. Blood sources in the Solares are humans, domestic vertebrates (pigs, cow, turkey, chicken, and dogs), and opportunistic vertebrates such as mice, marsupials, and birds that benefit from the presence of fruit trees [e.g., *Mangifera indica*, *Manilkara zapota* (L.) P. Royen, Guabana, Anona, Saramuyo], trees for local use [*Brosimum alicastrum* Swartz, *Bursera simaruba* (L.) Sarg, *Ceiba pentandra* (L.) Gaertn.] *Hylocereus undatus* [(Haw.) Britton and Rose] and other plants (Ordóñez-Díaz, 2018). Therefore, these systems offer good conditions for the infestation by *T. dimidiata*. We included two sites within this category: Emiliano Zapata and Xul, which are separated by 13.55 km (**Figure 1** and **Table 1**).

Kaxil Kiuic biocultural reserve (KKBR) and forest fragments around Yaxhachén (separated by 4.03 km) are the two localities with continuous forest and fragments of Median sub deciduous tropical forest (Msb), respectively (**Figure 1** and **Table 1**). KKBR has an area of 1,642 ha of Msd that has existed for

TABLE 1 | Number of specimens of *Triatoma dimidiata* according to measured traits per landscape class.

Landscape class	Altitude	Longitude	Latitude	Thorax		Wing		Sensilla		Proboscis	
				♀	♂	♀	♂	♀	♂	♀	♂
Urban (U)											
Mérida	34 ± 3	−89.62305	20.96583	31	31	59	31	24	18	32	28
Homegarden "Solares Mayas" (Hg)											
Emiliano Zapata, Oxkutzcab	58	−89.46663	20.22391	16	28	23	29	7	6	16	20
Xul, Oxkutzcab	50	−89.46426	2010191	2	1	1	2	2	–	2	1
Median subdeciduous tropical forest (Msd)											
Kaxil Kiuic, Oxkutzcab	96	−89.55111	20.08958	26	27	99	50	34	27	24	27
Yaxhachen Oxkutzcab	93	−89.52037	20.06158	14	13	2	12	2	1	16	15
Secondary vegetation (Sf)											
Xkobehaltun, Oxkutzcab	91	−89.52462	20.08072	26	31	49	26	7	11	26	30
Potoi, Oxkutzcab	68	−89.4864	20.23498	1	1	–	1	6	4	1	1
Bombahaltun, Oxkutzcab	87	−89.42347	20.05857	5	3	4	5	1	–	4	3
			Total	121	135	237	156	83	67	121	126

more than 100 years (Essens and Hernández-Stefanoni, 2013). Trees reach a height of 13–20 m (Dupuy et al., 2012) in both localities, dominating the arboreal and shrubby elements, with scarce climbing plants and epiphytes. Vertebrate communities of potential vector hosts at KKBR are composed by birds (151 species), mammals (40 species), and reptiles (36 species) (Callaghan and Pasos, 2010). Several mammal species present in this region have been implied in the in enzootic cycle of *T. cruzi* (Dumonteil et al., 2018; Moo-Millan et al., 2019; Torres-Castro et al., 2021).

Finally, secondary vegetation (Sv) is represented by three sites (X'kobehaltun, Potoi, and Bombahaltún) with different ages of abandonment after use of traditional agriculture with the method of roza-tumba-quema (**Table 1**). Among the most abundant tree species are *Neomillspaughia emarginata* (Grooss) Blake, *Gymnopodium floribundum* Rolfe, *Bursera simaruba* (L.) Sarg, and *Piscidia piscipula* (L.) Sarg (Rico-Gray and García-Franco, 1992). Vertebrate community and potential reservoirs of *T. cruzi* at these sites are species that tolerate agricultural land-use such as *Peromyscus yucatanicus*, *Oryzomys couesi*, *Sigmodon hispidus* rodents, *Artibeus jamaicensis*, *Desmodus rotundus* bats, and *Dasypus* spp., *Didelphis virginiana,* and *Nasua narica* mammals (Panti-May et al., 2021).

Insect Collections

Triatomine bug adults were collected from April 2014 to June 2019. We conducted sampling for 3 h (20:00–23:00 h) using 3 × 3 m, white blankets and led lamps (Light traps) with standardized light intensity (2000 lumens). We also searched for insects (e.g., under fallen leaves, rocks, trunks, houses) at a radius of five meters from the light traps. Collected bugs were preserved in ethanol (90%). Insects from the city were reported and delivered alive at inhabitants' homes the same or following day of report.

Measurement of Fitness Proxy and Feeding Information

Both males and females were dissected to extract the intestine, visually register the feeding status (i.e., blood presence), and calculate the proportion of feed insects. We quantified egg production as a fitness proxy registering the number of eggs per female in each landscape and year. For dissections, bugs were placed in a ventral view in Petri dishes sterilized with 96% alcohol. For each specimen, we used sterilized scalpels and made two lateral cuts in the ventral region of the abdomen from the anterior part to the anal section. Both dissections made it possible to remove the ventral area of the abdomen as a lid. The digestive tract was removed using sterilized entomological forceps. Dissections and egg number counting were performed by a single person to minimize sources of error.

Flight Morphology Data

The right wing and thorax for females (wings n = 248 and thorax n = 131) and males (wings n = 162 and thorax n = 134) were used to characterize flight morphology among the four landscape classes (**Table 1**). We obtained wing and thorax images with an integrated photo-macroscope model Leica Z16APOA (Leica Microsystems AG, Wetzlar, Germany). Images were processed in MakeFan8 (Sheets, 2003) to draw templates that allowed us to register the contours. For wings, we outlined three templates by registering 37 semi-landmarks. For thorax images, we removed four templates by registering 19 semi-landmarks (black dots, **Figures 2A,B**).

Landmarks type I were also recorded for wing venation origins and intersections (white dots, **Figure 2**), while for the thorax, we selected four landmarks type II (Bookstein, 1991). We employed TPSDig (Rohlf, 2007) to register the Cartesian coordinates of the 48 and 23 points for wings and thorax, respectively. Landmark configurations were adjusted with CoordGen8 (Sheets, 2005) using a Generalized Procrustes analysis based on the sum of the least-squares at each point. This adjustment allows to minimize the variation associated with the effects of positioning, orientation, and scale (Bookstein, 1991; Zelditch et al., 2004). A second adjustment was applied to eliminate variation by sliding the semi-landmarks based on the alignment distance criteria of the Semiland8 (Sheets, 2002). The superimposed and aligned Procrustes coordinates were used to calculate shape variables (Partial Warps, PW)

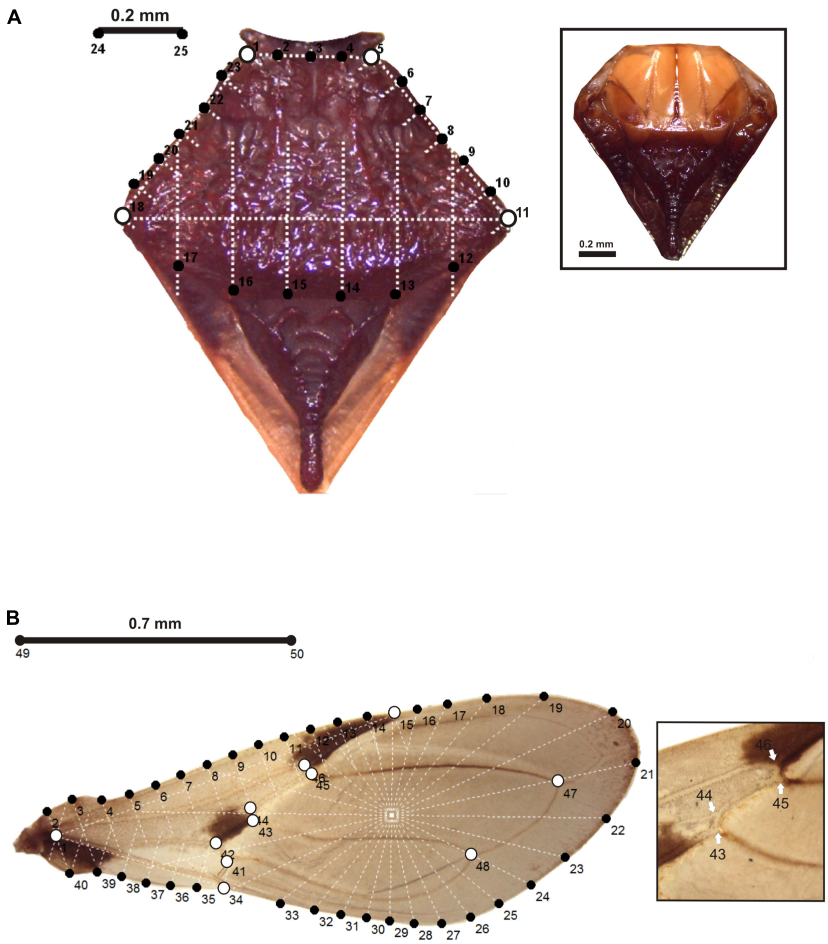

FIGURE 2 | Landmarks used to capture the configuration of *T. dimidiata* shape: **(A)** thorax (Landmarks, LM = 4 and Semilandmarks, SML = 19; right, muscle mass in thorax); and, **(B)** wing (Landmarks, LM = 11 and Semilandmarks, SML = 37). White dots are LM, and black dots are SLM.

and centroid size (Cs, as a measure of size) for each anatomical structure.

The wing aspect ratio (WAR) was calculated for each individual as wing span^2/wing area (Females n = 248 and Males n = 162). Measurements were carried out in ImageJ v. 1.52p (image processing and analysis in Java, free access[1], Rasband, 1997–2019).

Antennal Phenotype Data

We examined the antennal phenotype (defined by the number of sensilla) of 150 adult bugs (n = 83 females and n = 67 males) (**Table 1**). Previously, the antenna was dissected and processed with sodium hydroxide 4% for 6 h at 60°C, and then neutralized with glacial acetic acid (5%) for 2 min. Subsequently, each antenna was preserved and mounted temporally with glycerin for identification and counting of sensilla under a Leica DM300 microscope (Leica Microsystems AG, Wetzlar, Germany) at 400x. We followed the nomenclature proposed by Catalá and Schofield (1994) for olfactory sensillum identification. We identified and counted the followed sensilla for the ventral region of Pedicel (P), Flagellomere I (FI), and Flagellomere II (FII): Basiconic (Ba), bristles (Br), thin-walled trichoid (TH), and thick-walled trichoid (TK).

Feeding Morphology Data

We measured the proboscis length of 246 adult bugs (n = 121 females and n = 126 males, **Table 1**) from its basis to distal region. Measurement (in mm) was taken with a squared sheet (1 × 1 mm) under a Leica Z4 stereoscope (Leica Microsystems AG, Wetzlar, Germany).

The person who took all measurements was unaware of the bug's collection site to avoid any observer's bias.

STATISTICAL ANALYSES

Feeding Status and Fitness Proxy Differences

We carried out a generalized linear model setting a negative binomial distribution and Tukey *post hoc* tests to evaluate

[1]http://rsbweb.nih.gov/ij/

differences in feeding status among landscapes. Also, to evaluate differences between the number of eggs by females among landscapes we performed a Kruskal–Wallis test and *post hoc* test with Holm correction. The analysis was performed in R v. 3.1.0; MASS and emmeans packages were used for construction of generalized linear model and *post hoc* tests, respectively.

Geometric Morphometric Analysis

Given size-related gender differences in our study species (Lehmann et al., 2005), we first analyzed shape and size differences between sexes (dimorphism) in thorax and wings. Sexual dimorphism in size was tested through two-way analysis of variance (ANOVA), where shape dimorphism was determined through a permutation test using Goodall's F test implemented in TwoGroup8 with 3,600 bootstrap permutations (Sheets and Zeldich, 2001). Given the significant differences in both structures (see section "Results"), the subsequent analyses were performed by sex (**Supplementary Table S1**).

Because the two principal structures related to flight dispersion in Triatominae bugs are the thorax and wings (Hernández et al., 2015), we analyzed the possible covariation between these two structures, and also whether such covariation was affected by landscape type, thus producing different selective pressures and promote patterns, for example, in favor of individuals or populations with increased flight performance or dispersion. Therefore, we performed a partial least square analysis (PLS) to assess the covariance between the aligned morpho-anatomical structures (wing and thorax). Covariation analysis was performed using the function "two.b. pls" as implemented in the geomorph package v. 3.1.0 in R (Adams et al., 2020).

For each structure, we explored patterns of generalized shape variance among all individuals using the function "*gm.prcomp*" on Procrustes shape as implemented in geomorph package v. 3.1.0 (Adams et al., 2020). We used the function "*mshape*" to generate the consensus shape and the function "*shape.predictor*" to estimate the predicted shapes along the principal component axes. These inferred configurations were employed to visualize the shape differences among individuals along the principal component axes. Both functions are implemented in the "geomorph" package.

We conducted a linear mixed model (LMM) to evaluate the landscape effect on WAR, shape variables, and centroid size of both structures (wing and thorax). The year of the collection (period 2014–2018) was considered a random effect while body size and the interaction among variables were considered in the model. Additionally, for LMM where wing and thorax shape were entered as a response variable, we tested the relation with the WAR to detected patterns of covariation among these variables related to flight/dispersion. Because the detection of an allometric effect could influence shape inferences, we also tested if a significant association between shape and size (as Log transformed) existed. The LMM was performed according to a residual randomization permutation procedures (RRPP) approach with the "*rpp.lm*" function (with 2,500 permutations) as implemented in the RRPP package in R v.3.5.0 (Collyer and Adams, 2020). To assume that landscape type effects are independent of geographic location, we included a geographical covariance matrix in the "*rpp.lm*" function with the argument "*cov*". This allowed driving the non-independence of the error by geographic location in the estimation of the coefficients of the landscape effect via GLS (Generalized least square). We computed geographic covariance matrix with the "*cov.sp*" function in SpatialTool package v.1.0.4 (French, 2008).

Antennal Data Analysis

We carried out univariate and multivariate analyses to assess differences in the number of antennal sensilla among the four landscape classes. The univariate analysis allowed us to describe and identify the type of sensilla by segment that exhibited differences between the triatomines. Sensilla that did not show significant differences among landscapes classes were discarded in subsequent analyses. As recommend in antennal phenotypic analysis of Triatominae bugs (May-Concha et al., 2016), we performed the univariate analysis at three levels: sensillum (Ba, Br, TH, TK), segment (P, FI, FII expressed as Total), and antenna (defined as TOTAL). All univariate analyses were performed by sex. At the sensillum level, the mean and standard deviation for each type of sensillum was calculated by each segment, and differences among landscape type were evaluated. The mean and standard deviation of the total abundance by segment were estimated at the segment level, and differences between landscape classes were calculated. Finally, we assessed differences in the total abundance (number) of sensilla considering all segments (antenna level). Significant differences were evaluated following generalized linear mixed models (GLMM), considering collection year as a random effect (period 2015–2019). GLMM was performed using a Poisson distribution model, and multiple comparisons were estimated using Tukey's contrasts. The analysis was performed in R v3.5.0. "*lmer4*" and "*multcomp*" packages which were used to construct all GLMMs and multiple comparisons, respectively.

We employed the number of sensilla by segment (by each individual) to perform a principal component analysis (PCA) to explore patterns of variance among all individuals. We used the results of corresponding PC's to reduce the number of independent variables in the multivariate analyses. We discarded the PC axes whose contribution to the cumulative variance was not significant. Then, we conducted a multivariate analysis of variance with a canonical variate analysis (CVA) using Wilk's lambda test. CVA's was performed with all specimens for the four landscape class groups. The scores of each canonical variate were used for the univariate ANOVA according to the Scheffé test to determine the landscape class discriminated by CVA axes. The correct classification and CVA performance were evaluated considering the posterior probabilities of assignation. The multivariate analysis was implemented in the "*MASS*" and "*Candisc*" package in R v3.5.0.

Feeding Morphology Analyses

We carried out an LMM to evaluate differences in the proboscis length among landscape classes. The model was constructed considering sex and body size as independent variables, and entering collection year (period 2014–2018) as a random effect.

LMM was performed with 4,000 permutations in the RRPP package (Collyer and Adams, 2020) in R v3.5.0.

Morphological Disparity Analysis

The general concept of disparity refers to the relative quantity of variability scaled in a sample of measures. The disparity is measured from the quantitative description of the morphological variation and the empirical distribution of samples in some morpho-space (i.e., PCA, Foote, 1991). We considered that relative low values of disparities are equivalent to decreased variance in the phenotype. Our interest was to investigate whether the levels of morphological disparity in the four morphological traits were different between landscapes.

For proboscis length we calculated the coefficient of variation, which is the metric of variability in the univariate case (Van Valen, 2005). We evaluated differences in coefficient of variation with the "*asymptotic test*" function implemented in the "*cvequality*" package in R (Marwick and Krishnamoorthy, 2019). In the multivariate case, disparities in wing and thorax were evaluated using the Procrustes pairwise distances between specimens and the sample mean as metric of the morpho-space occupied by all specimens in a landscape class. We estimated the values of disparities of wing and thorax with the "*morphological.disparities*" function implemented in geomorph package in R. We used a permutation test (2,500 iterations) to assess statistical significance differences (Adams et al., 2020). Morphological disparities in the antennal phenotypes were estimated through principal components scores using the "dispRity" package in R (Guillerme, 2018). We used a bootstrapping procedure (2,500 iterations) to assess 95% confidence intervals and a *t*-test to evaluate significant differences between values of morphological disparity after Bonferroni correction.

Pattern of Similarities in Morphometric and Morphological Data

We calculated a Mahalanobis distance matrix for each morphometric and morphological data by sex. These distance matrices were employed to construct a pattern of hierarchical similarities among landscape classes that were visualized in a cluster analysis based on the UPGMA (Unweighted Pair Group Method using Arithmetic averages) algorithm performed in R v3.5.0. We compared the topology and correlation between clusters to detect patterns of similarity/concordance between dendrograms generated by each morphological and morphometric data. Due to the uneven number of samples between the morphometric and morphological data set, dendrogram comparisons were carried out with the dendextend package in R v.3.5.0 (Galili, 2015). We calculated the quality of the alignment among clusters (topology) with the "entanglement" function, where values range from 0 to 1 where a lower entanglement coefficient corresponds to a good alignment (values close to 0). Also, we calculated a correlation coefficient through de Goodman and Kruskal's gamma index (γ). The values of this index vary from 0 (absence of agreement) to 1 (100% positive association or perfect agreement).

RESULTS

Abundance, Fitness Proxy and Feeding Information

Eight hundred and seven bugs were collected during 2014–2019 (55% Msd, 18% Sv, and 15% U; **Supplementary Table S2**). Overall, 67% of the triatomines showed blood ingestion, of which 74% were females and 57% were males. Seventy nine percent and 70% of triatomines that were captured in Secondary vegetation (Sv) and forest (Msd) respectively were fed, while 65% of the urban bugs were fed. These frequencies were significantly different (Chi-squared = 18.938, df = 3, p-value = 0.00028; **Table 2**). A *post hoc* comparison showed further differences between forest (Msd) and Urban (U) (p = 0.018) and Secondary vegetation (Sv) and Urban landscape class (p = 0.015, **Table 2**). Only 18% of females were gravid (n = 84). Most gravid females were observed in 2015, 2016, and 2018. Two hundred and forty-two females were recorded in Msd, and 26% of them were gravid, while of the 100 females in Sv, 38% were gravid. In Homegardens (n = 52) and Urban (n = 79), only 21 and 19% of the females were gravid, respectively. The number of eggs per female between landscapes was barely significantly different (Kruskal–Wallis chi-squared = 8.4297, df = 3, p-value = 0.04; **Supplementary Table S2**), However, *post hoc* comparisons showed only one marginally significant difference between urban (U) and forest (Msd) landscape class (p = 0.049), with the former having fewer eggs.

Flight Morphology Variation

Thorax

The LMM detected significant differences in thorax size among landscape classes (**Table 2**). The same variation pattern was detected in both sexes. A statistically significant smaller thorax size was found in Urban (U) individuals, while conserved and rural landscapes had a significantly larger thorax size (**Figure 3** and **Table 3**).

A large thorax size was observed in individuals from Hg and Sv (**Figure 3**). For both sexes, the PCA based on thorax shape indicates a small proportion of variance explained by the first two components (females: PC1 35%, PC2 18%; males: PC1 38%, PC2: 15%) (**Figures 4A,B**). According to the LMM, no significant differences in thorax shape among landscapes were detected (**Supplementary Tables S3, S4**).

Wing

The LMM revealed no significant differences in wing size and WAR among sexes between landscapes (**Table 2** and **Supplementary Table S3**). For males and females, the PCA based on wing shape indicated that the first two components explained a small proportion of variance (female, PC1 30%, PC214%; male, PC1 35%, PC2 15%) (**Figures 5A,B**). Also, the LMM did not detect significant differences in wing shape among landscapes (**Supplementary Table S4**).

The PLS showed that wing and thorax morphology did not covariate (female, r = 0.359, P = 0.094; male, r = 0.302, P = 0.061).

TABLE 2 | Results of ANOVA for a LMM for wing and thorax size (as Cs) of *T. dimidiata*.

Component	Female							Male						
	DF	SS	M Sq	R^2	*F*	*Z*	*P*-value	DF	SS	M Sq	R^2	*F*	*Z*	*P*-value
Wing														
Body size	1	0.0137	0.0137	**0.0208**	5.3554	1.3192	***0.019***	1	0.0112	0.0112	0.0216	3.4544	1.1365	0.070
WAR	1	0.0011	0.0011	0.0016	0.4359	0.2256	0.495	1	0.0022	0.0022	0.0043	0.6904	0.4006	0.411
Landscape	3	0.0315	0.0078	0.0476	2.4012	1.2985	0.074	3	0.0170	0.0042	0.0329	4.1287	1.5726	0.205
Landscape: Year	8	0.0262	0.0032	0.0396	1.2753	0.6763	0.260	7	0.0072	0.0010	0.0139	0.3182	−1.7314	0.945
Residuals	223	0.5996	0.0025	0.9057				140	0.4814	0.0032	0.9271			
Total	236	0.6620						152	0.5192					
Thorax														
Body size	1	0.0016	0.0015	0.0004	0.1607	−0.260	0.687	1	0.0199	0.0199	0.0011	0.2671	−0.0305	0.601
Landscape	3	2.0473	0.5118	**0.6048**	2.3694	2.432	***0.005***	3	7.4537	1.8634	**0.4421**	2.49725	4.0428	***0.001***
Landscape: Year	9	0.1944	0.0216	0.05743	2.1945	1.605	0.500	7	0.4285	0.0535	0.0254	0.7178	−0.1243	0.532
Residuals	108	1.1418	0.0098	0.33730				121	8.9543	0.0746	0.5312			
Total	121	3.3851						133						

Significant P-values are in bold and italics.

Antennal Phenotype Variation

The univariate analysis rendered similar patterns in sensillum abundance in both sexes. Our results indicated significant differences between landscapes at the sensillum level, determined by the number of sensillum types TK and TH. Among landscapes, differences in TK and TH were observed in the three segments (**Supplementary Figures S1, S2** and **Supplementary Table S5**). In all segments, differences in the number of sensilla type Ba among landscapes were identified only in males.

At the segment level, in both sexes, the number of sensilla from Pedicel and Flagellomere I (FI) were greater in Msd and Sv than in Urban landscape (**Supplementary Figures S3, S4** and **Supplementary Table S5**). In Flagellomere II, differences in sensillum total abundance were only observed in males but not in females. Also, in FII, we found fewer sensory receptors for Hg and U, yet a high abundance for Msd and Sv (**Supplementary Table S5**).

The CVA detected significant differences in sensory phenotypes among landscapes in both sexes (**Supplementary Table S6**). For each sex, only one canonical ax allowed discrimination among landscapes (female, Wilks' $\lambda 1 = 0.5157$, $\chi^2 = 2.5957$, d.f = 21, $p < 0.001$; male, Wilks' $\lambda 1 = 0.4422$, $\chi^2 = 2.2066$, d.f = 24, $P = 0.0019$). In both sexes, the most different groups were the anthropized landscapes (U) and conserved ones (Msd) (**Figures 6A–D**). The proportion of correct assignment of specimens based on posterior probabilities obtained from the CVA scores supported the differences among U and Msd landscapes. In females, the main difference between the landscapes was observed in TH of the FII (0.088), while in males, it was observed in Ba of the FI (0.066) (**Supplementary Tables S6, S7**).

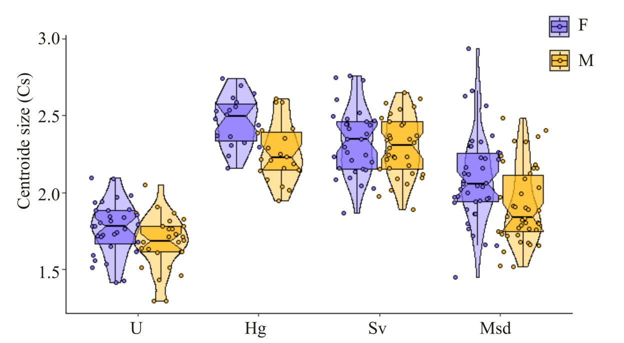

FIGURE 3 | Thorax size variation among landscapes for *T. dimidiata* female (F) and male (M). Landscape: Urban (U), Homegarden (Hg), Median subdeciduous tropical forest (Msd), and Secondary vegetation (Sv).

Feeding Morphology Variation

No significant differences were detected among landscape types for the length of the proboscis ($F_{3,245} = 2521$, $P = 0.069$, $R^2 = 0.027$, **Supplementary Table S8**). Also, the results showed that the length of the proboscis was not influenced by sex or body size (**Supplementary Table S8**).

Morphological Disparities

The disparity values of the morpho-functional traits revealed differences between landscapes, but not all the disparity values follow the urban and non-urban gradient (**Table 4**). For example, for females the results showed that the Procrustes variance in thorax size and AP decreases in urban landscapes, while the disparity values of the thorax shape, wing shape and proboscis length decrease in Msd landacape. The highest values of disparity in AP and proboscis length were observed in Hg and Sv, which exhibit about three times greater variance than Urban or Msd (**Table 4**).

In males, the pairwise disparity comparisons of the wing morphology (shape and size) and thorax shape did not reveal significant differences among landscapes, while the Procrustes variance of thorax size and proboscis length was significantly lower in Urban and Msd classes, respectively. On the contrary, disparity values in AP showed significant differences between

TABLE 3 | Results of pairwise comparison of thorax size of *T. dimidiata* among landscapes Z and P are based on 2,500 random permutations.

Landscape class	Female				Male			
	d	**UCL (95%)**	***Z***	**Pr > d**	***d***	**UCL (95%)**	***Z***	**Pr > d**
U: Hg	0.6984	0.1898	9.9027	***0.001***	1.3968	0.8620	6.8574	***0.001***
U: Sv	0.5628	0.1724	8.9711	***0.001***	1.1257	0.7062	6.5337	***0.001***
U: Msd	0.3259	0.153	5.4259	***0.002***	0.6518	0.4585	4.0255	***0.001***
Hg: Msd	0.3724	0.1824	5.2903	***0.001***	0.7449	0.5277	3.9390	***0.001***
Hg: Sv	0.1355	0.1901	0.9285	0.171	0.2710	0.2964	1.4669	0.092
Msd: Sv	0.2369	0.1536	3.6506	***0.002***	0.4739	0.3669	2.9436	***0.001***

Landscapes: Urban (U), Homegarden (Hg), Secondary vegetation (Sv) and Median subdeciduous tropical forest (Msd). Significant P-values are in bold and italics.

Urban and Msd, but the lower value was observed in Sv class (**Table 4**).

Patterns of Hierarchical Similarities

Thorax size and antennal phenotype similarities among landscapes were visualized in a cluster analysis based on the UPGMA algorithm. Thorax size and phenotype dendrograms showed entanglement values of 0.14 and 0.16 for females and males, respectively. For both sexes, the correlation coefficients for both dendrogram pairs were 0.85, suggesting high association and agreement. All dendrograms showed two groups, being the Urban the most distinct class. The second group is integrated by heterogeneous but conserved sites (Msd, Sv, and Hg) (**Figure 7**).

Differences in all dendrograms were related to the position of Hg and Sv. For females, the thorax size of Msd was most similar to Sv, but not so when considering the antennal phenotype. On the contrary, for males, the thorax size in Msd was similar to Hg but with an antennal phenotype similar to Sv (**Figure 7**).

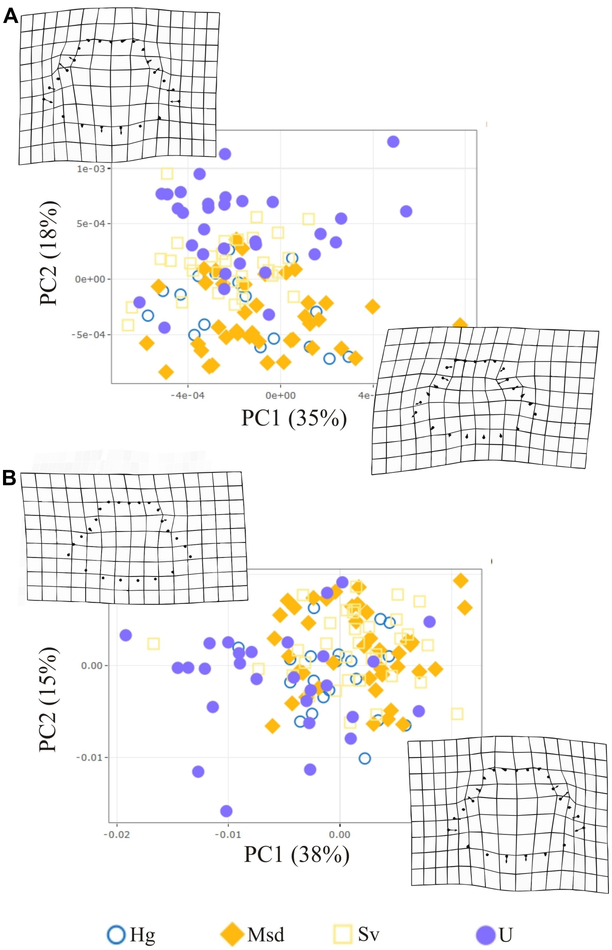

FIGURE 4 | PCA of thorax shape variation for *T. dimidiata* females **(A)** and males **(B)**. Landscape: Urban (U), Homegarden (Hg), Secondary vegetation (Sv), and Median subdeciduous tropical forest (Msd).

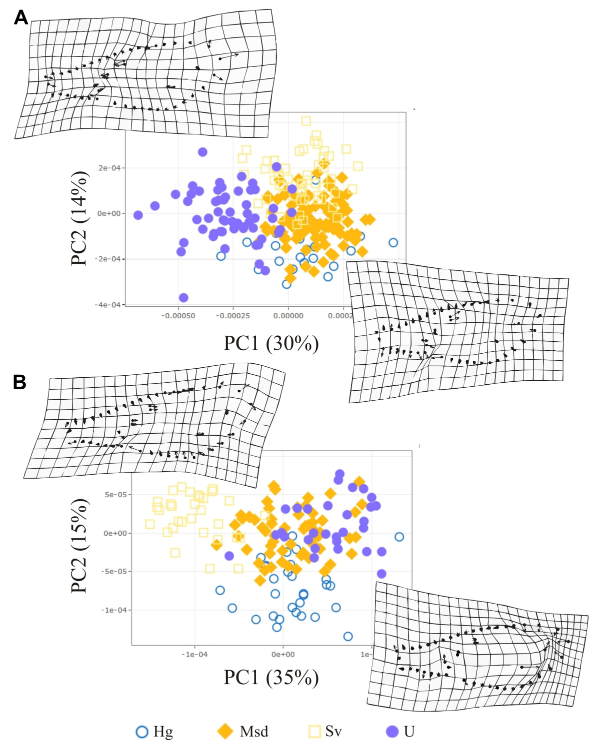

FIGURE 5 | PCA of wing shape variation for *T. dimidiata* females **(A)** and males **(B)**. Urban (U), Homegarden (Hg), Secondary vegetation (Sv), and Median subdeciduous tropical forest (Msd).

DISCUSSION

Our general assessment of how morpho-functional traits of *T. dimidiata* have responded to a non-urban and urban gradient provides new insights of likely drivers underlying adaptive processes for triatomines. In what follows, we discuss our results in terms of how microevolution is presumably acting in our study species at the population level. First, there were similar fitness payoffs among landscapes. Although only urbanized animals showed some reduction in egg number, gravid females were similarly common in urbanized and non-urbanized areas. Second, our results suggest that changes in wing morphology are not associated with landscape modification (e.g., Urban, Rural) in contrast to thorax size. Third, thorax size and antennal phenotype reveal a morpho-functional convergence as both show a similar trend in phenotypic simplification. For both sexes, the phenotypic pattern follows a simplification in the non-urban and urban gradient. And fourth, convergence of thorax size and antennal phenotype suggests that *T. dimidiata s. l.* exhibits differences in flight/locomotion performance and dispersion as an adaptive response to host availability. We discuss these results at length below.

There was a decrease in thorax (pronotum) size and sensillum antennal density in urban individuals relative to their sylvatic and rural counterparts. Previous studies had documented intra and interspecific variation in flight capacity in triatomine bugs associated to a lack of development of flight muscle or thorax size (e.g., *T. sherlocki*) (Almeida et al., 2012; Hernández et al., 2015; Nattero et al., 2017). Given that flight initiation and dispersion in triatomine bugs is modulated by starvation or inanition (Lehane and Schofield, 1982; Lehane et al., 1992; McEwen and Lehane, 1993; Galvão et al., 2001), a large thoracic area would grant bugs an enhanced ability to fly and, thus, disperse to track diverse and scattered host burrows. This could explain why in sylvatic and regenerating landscapes (i.e., abandoned fields) and rural ecotopes, triatomines exhibit a large thoracic size and disparity values. However, it is also possible that the similarities in thorax size between Msd, Sv, and Hp are explained by the fact that the great majority of individuals within these categories (Sv and Hp) have a sylvatic origin, which coincides with the infestation/movement process documented in rural landscapes in this region (Dumonteil et al., 2002, 2007; Nouvellet et al., 2011). Our findings suggest that occupying stable landscapes (i.e., urban landscape class), where hosts do not have seasonal changes in abundance and are thus predictable throughout the year, would allow a reduction in triatomine investment in structures of reduced use.

The morpho-functional pattern documented in thorax size is similar to that of antennal sensilla, showing a clear morphological simplification in *T. dimidiata s. l.* in the urban landscape. The antennal phenotype of this species is characterized by a high diversity of chemoreceptor sensilla in the pedicel (Catalá et al., 2005; Arroyo et al., 2007; May-Concha et al., 2016, **Supplementary Table S9**), which coincides with the complexity of the niche it occupies throughout its geographical distribution (Ibarra-Cerdeña et al., 2014; Villalobos et al., 2019). Given that TK and TH chemoreceptors are sensitive markers of habitat influence in our study species (Catalá et al., 2005; Arroyo et al., 2007; May-Concha et al., 2016, 2018), our findings indicate that these odor-detecting traits may be responding to selection via a reduction in complexity in urbanized environments. Interestingly, abundance of TK flagellum in *T. infestans* (Catalá and Dujardin, 2001) and *T. dimidiata* (Catalá et al., 2005) was higher in urbanized areas in South America. Notwithstanding, we found that the highest abundance of TK sensilla was observed in individuals from Msd and Sv and not in stable landscapes as represented by U. Also, we found a complex relationship between antennal phenotype and sylvatic habitats (Msd and Sv), as evidenced by the density of TK and TH sensilla and the total abundance of sensilla per segment. In this regard, sensillum density in the pedicel and FI was higher in Msd and Sv than in Urban and Hg for both sexes. Additionally, unlike females, males showed a nature-urban gradient in the total abundance of sensilla in FII and significant differences in Ba abundance between landscapes. These sexual differences may be related to mate searching activities by males because Ba is related to the perception of sex-pheromone components (Bohman et al., 2018). A functional explanation for the decrease in the density of sensilla in urban environments is that a high density of such structures is not needed in stable landscapes as blood sources can be, as indicated above, less diverse and more predictable. This idea would be supported mainly by the sensillum abundance at the pedicel in urban triatomines as this trait has been described in domiciled species that either occupy stable habitats (e.g., *T. infestans*, Catalá, 1997; Catalá et al., 2005) or use humans and other domestic animals (canids, chickens, cats) as the main hosts (Ceballos et al., 2005; Rabinovich et al., 2011; Alvedro et al., 2021).

One remarkable aspect of our findings is that these are compatible with Schofield's domiciliation hypothesis which proposes a pattern of phenotypic/genotypic simplification in triatomines from urbanized areas (Schofield et al., 1999; Flores-Ferrer et al., 2018). Although the methodological approach is different, our detected simplification pattern matches with that documented for *Panstrongylus geniculatus* in Caracas, Venezuela (Aldana et al., 2011). Because both sexes are strictly blood-sucking, host-feeding traits in urban landscapes will show convergence, which is the case of thorax size and antennal phenotype. Moreover, our results are also in agreement with those of other insects, where morpho-functional structures are smaller in urban individuals than their rural counterparts and that such phenotypic variation would be related to the respective foraging landscape (Eggenberger et al., 2019). The phenotypic simplification in *T. dimidiata* may imply a reduced ability during flight/locomotion and dispersion, related to a less diverse yet more predictable hosts in urban landscapes. This logic has a sound background: it has been shown that while bugs make use of a larger range of hosts in rural environments, bugs that infest and colonize urban dwellings make use of humans and domestic animals (Alvedro et al., 2021). In this sense, the phenotypic simplification in urban bugs, could be supported by the blood-feeding profiles so far documented in bugs from the city (Guzmán-Tapia et al., 2007), which contrasts with the full range of host types identified in sylvatic and rural individuals

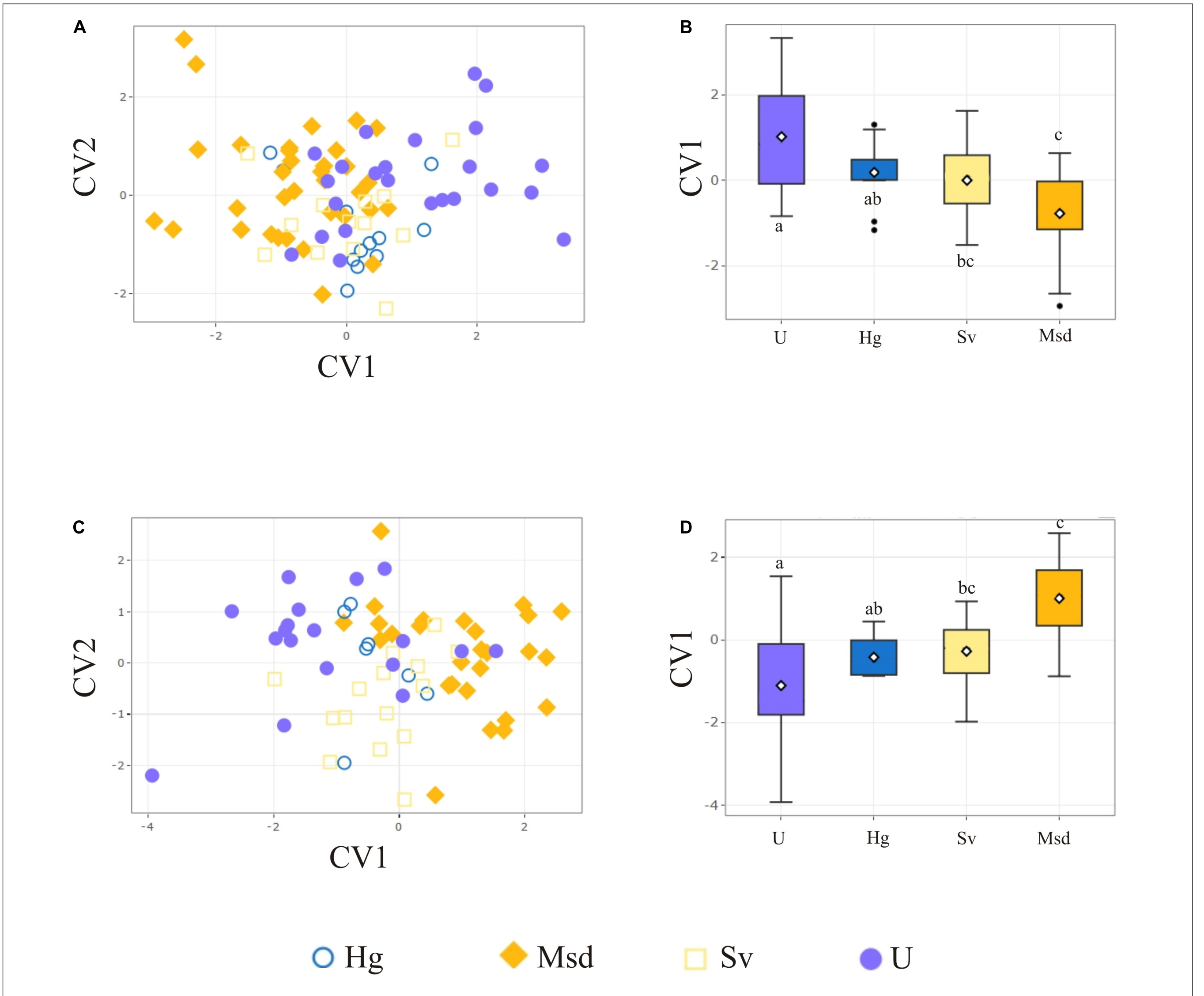

FIGURE 6 | Antennal phenotype variation of *T dimidiata* among landscapes by sex **(A,C)**. Distribution of individuals by landscape class along the first two canonical axes **(B,D)**. Female **(A,B)** and Male **(C,D)**. Groups with the same letter did not present significant difference ($p < 0.05$) according to the Scheffé test. Urban (U), Homegarden (Hg), Secondary vegetation (Sv), and Median subdeciduous tropical forest (Msd).

TABLE 4 | Disparity values of four morpho-functional traits of *T. dimidiata s. l.* along a nature-urban gradient.

Land	Female						Male					
	Wing		Thorax		Proboscis	AP	Wing		Thorax		Proboscis	AP
	Shape	Size	Shape	Size			Shape	Size	Shape	Size		
U (a)	0.0034/ **c, d***	0.0506/ –	0.0022/ **d***	0.0284/ **d***	5.0140/ **b, c***	3.078/ **b, c, d****	0.0004/ –	0.0446/ –	0.0012/ –	0.0290/ **b***	4.4237/ **b, c****	5.199/ **c, d***
Hg (b)	0.0039/	0.0269/ –	0.0008/ –	0.0254/ **d***	7.4041/ **a, d***	6.799/ **a, c, d****	0.0004/ –	0.0326/ –	0.0010/ –	0.2079/ **a***	9.1016/ **a, d****	6.74/ **c***
Sv (c)	0.0005/ **a***	0.0490/	0.0014/ –	0.0468/	6.9166/ a, d*	5.201/ **a, b, d****	0.0005/ –	0.0540/ –	0.0012/ –	0.0372/ –	7.4442/ **a, d****	1.989/ **a, d***
Msd (d)	0.0005/ **a***	0.0375/ –	0.0010/ **a***	0.0821/ **a, b***	4.5841**/** **b, c***	3.240/ **a, b, c****	0.0004/ –	0.0390/ –	0.0011/ –	0.0753/ –	5.1845/ **b, c****	6.086/ **a, c***

Procrustes variance for landscape classes (Upper)/statistical significance represent by letters (lower, $p < 0.05$, 0.001**).*

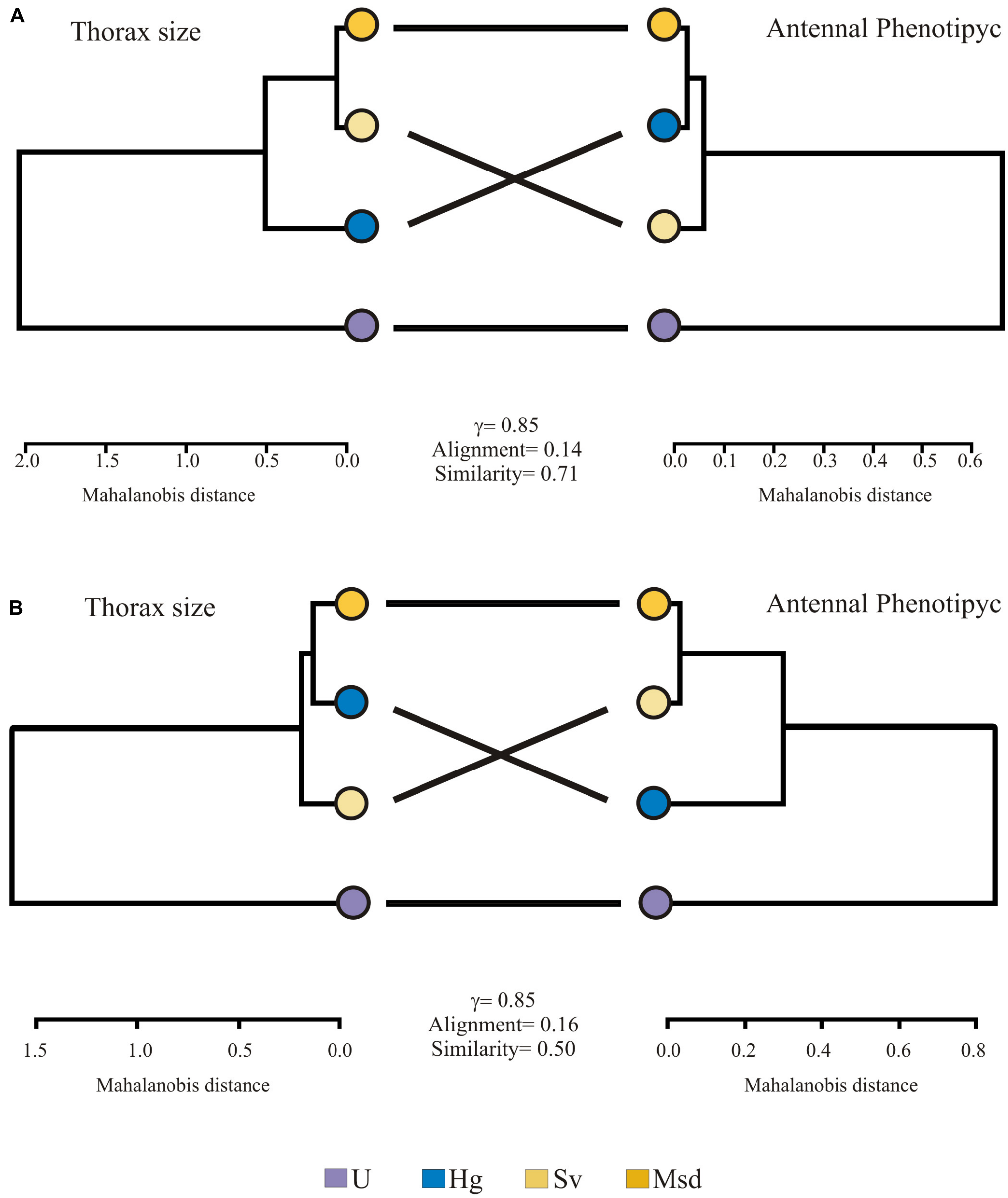

FIGURE 7 | Patterns of hierarchical similarities in thorax size (left) and antennal phenotypic (right) by landscape for *T. dimidiata* **(A)** female and **(B)** male. Landscape: Urban (U), Homegarden (Hg), Secondary vegetation (Sv), Median subdeciduous tropical forest (Msd) Goodman-Kruskal-gamma index (γ); Alignment quality (Alignment).

(López-Cancino et al., 2015; Dumonteil et al., 2018; Moo-Millan et al., 2019).

Opposite to our predictions, we did not find variation in proboscis length and wing morphology related to landscape, possibly showing that these structures are equally economic and functional among the different landscape classes. These results reflect the generalist and opportunistic nature that characterizes this species (López-Cancino et al., 2015; Dumonteil et al., 2018; Moo-Millan et al., 2019). On the contrary, specialization has been the main driver of the differentiation of proboscis length under a rural-urban gradient in other insects (Eggenberger et al., 2019). As for wing morphology, our findings agree with previous studies in the YP (Dumonteil et al., 2007; Nouvellet et al., 2011). Notice that wing shape is a genetically determined character (Birdsall et al., 2000; Zimmerman et al., 2000) so that the absence of differentiation documented in our study match with the low genetic structure detected (Dumonteil et al., 2007; Nouvellet et al., 2011) and the fact that there is mainly a single haplogroup (Hg1) circulating in Yucatan state (Pech-May et al., 2019). Our interpretation of the morphological stability between landscapes and the lack of functional integration with thorax is that the variation in wing morphology does not affect the flight performance and dispersion in *T. dimidiata s. l.* Actually, walking is another important dispersal mechanism in triatomines (Vazquez-Prokopec et al., 2004, 2005; Abrahan et al., 2011) and the muscle involved with such locomotion is also concentrated in the thorax (Davis and Holden, 2015; Hernández et al., 2015). The evidence suggests that through this dispersal mechanism (walking), triatomine bugs could disperse more actively, regardless of their nutritional status or even if they are not fed, which favors the invasion and colonization of new habitats (Lobbia et al., 2019). Thus, possibly a small thorax is related to a lower capacity for locomotion performance in urban bugs.

Nutritional status (as a body weight/body length ratio or feeding status) is associated with population structure and dispersal in triatomine bugs (Lehane and Schofield, 1982; Lehane et al., 1992). Although this study was carried out in different periods of time during certain months, the nutritional status would be an immediate indicator of the individual's physiological state, while the morpho-functional variables quantified in this study would reflect the conditions (environmental, nutritional, etc.) that prevailed during development of nymphal stages (Hernández et al., 2011). The former would also explain the significant differences in the frequency of blood feeding documented in this work. This is, it is a momentary indicator of the physiological state and blood-source availability. Although the frequency of blood feeding between urban versus rural and sylvatic individuals were significantly different, our results show that bugs from all landscapes are fairly successful in gaining access to blood (higher than 65%). For example. this value is higher than that reported for peridomicile bugs of *Rhodnius pallescens* in Panama (less than 40%; Gottdenker et al., 2011), and shows a less contrast between sylvatic and urban/rural bugs (a difference of 14% in *T. dimidiata* in our study vs. 40% difference in *Rhodnius pallescens*).

Several aspects of the morpho-functional simplification in *T. dimidiata* in urban areas are unknown and need to be addressed in future studies. First, would this simplification generate a trade-off with fitness-related traits? For example, could feeding on an urban host generate nutritional stress that affects their development, reproduction, and/or survival? Related to this, developmental instability (Dujardin et al., 1999; Nattero et al., 2015a,b) and reduced size and survival (Gutiérrez-Cabrera et al., 2021) has been documented in triatomines that feed on specific hosts that occupy predictable habitats. However, we are not aware of any fitness-related difference when triatomines use urban vs. non-urban hosts. On the other hand, despite providing data on egg number as a fitness proxy, there is not any apparent benefit of occupying urban landscapes where the availability of resources would be stable. Future studies should include several fitness measures (e.g., number of eggs laid, hatching nymphs and/or survival) to unravel the costs/benefits of occupying urban niches for *T. dimidiata*. Second, because the intraspecific morphological variation in Triatominae does not necessarily have a linear relationship with the genetic structure (Dujardin et al., 1999, 2009), the microevolutionary potential that urban environments could exert on this species remains unknown. *T. dimidiata* is a species sub-complex integrated by at least four evolutionary lineages (Pech-May et al., 2019) with records in several cities of Mexico (Ramsey et al., 2015) and other Central and South America countries (Zeledon and Rabinovich, 1981; Zeledón et al., 2005; Dorn et al., 2007). In this sense, because the ecological pressures in cities would be similar (Santangelo et al., 2018), it is expected that urban traits will be highly canalized, independently of genetically distinct lineages, as has been documented in the south American triatomine, *R. ecuadoriensis* (Abad-Franch et al., 2021). Third, related to the wide distribution of *T. dimidiata*, the question remains as for whether the morphological changes are explained by phenotypic plasticity or are evolutionary responses to selection at the population level. One key feature of vectors that are expanding their distribution range is their phenotypic plasticity (Lefèvre et al., 2009). Future research should answer this for triatomines. This can be done with selection experiments whereby phenotypic plasticity is examined via fitness payoffs in different environments. And fourth, although we have provided a general discussion on how the phenotypic traits we measured are related to urbanization, a more intimate link may occur at the level of microhabitat structure. For example, the antennal traits can be used to detect places with particular humidity and temperature ranges where bugs can hide (e.g., *Triatoma infestans* Vazquez-Prokopec et al., 2002). Thus, one further step would be to link such traits with microhabitat structure.

Ample evidence has shown that the biotic and abiotic characteristics in urbanized environments are responsible for the nearly 1600 phenotypic changes that have been documented in plant and animal species worldwide (Alberti et al., 2017). Cities are fragmented landscapes with a high degree of human disturbance, characterized by pollution, noise, light intensity, where the habitat structure is shaped by artificial buildings that tend to increase the temperature (urban heat island effect) and thus affect important biophysical and physiological processes (Liker, 2020). The availability of food and nutrients in the urban landscape is also another factor that influences inter/intraspecific competition, predation and diversity (Biard et al., 2017; Liker, 2020). All of these ecological conditions can generate strong selection that leads to the evolution of specific urban phenotypes. In this regard, while we know that the triatomine phenotype is modulated by ecological factors in the sylvatic-rural gradient within a context of an anthropogenic drive (domiciliation), the role of cities as new adaptive filters has been little investigated. While in many cases, urban evolution is explained by non-adaptive processes (i.e., genetic drift), there is evidence that suggests that genetic diversity and gene-flow among populations of *T. dimidiata* are not associated to anthropic gradients in Yucatan state (Dumonteil et al., 2002; Pech-May et al., 2019). This gives support to the hypothesis that differentiation in thorax size, density of sensilla, and their morphological convergence, cannot be explained by genetic drift. The feeding characteristics of *T. dimidiata* (i.e., opportunistic and host generalist status), and assisted movement of bugs by rural human migration to Mérida (Chi-Méndez, 2016) could help sylvatic insects to move into the urban landscape, thereby increasing genetic diversity and reducing the genetic structuration (e.g., "Urban facilitation models," Miles et al., 2019). Since infestation and transmission of CD in urban settings has increased elsewhere (Vallvé et al., 1996;

Levy et al., 2006; Medrano-Mercado et al., 2008; Carvalho et al., 2014; Parente et al., 2017; Alvedro et al., 2021), there is a potential signal for urban evolution in triatomines, due to the presence of a suitable niche where susceptible hosts are abundant and not intermittent (Bradley and Altizer, 2007). What factors could be driving such niche search? Food resource availability in cities appear to be a leading factor driving phenotypic adaptation in several species (Serruys and Van Dyck, 2014; Alberti et al., 2017; Biard et al., 2017; Eggenberger et al., 2019; Liker, 2020) including, possibly, our study species. We do not rule out that other evolutionary mechanisms (e.g., orthogenesis) or phenomena (e.g., epigenetics/pleiotropy) could be related to morphological simplification in urban individuals of *T. dimidiata*. However, our findings should foster research linking genetic bases of this morpho-functional structuring and possible fitness benefits in the urban landscape. Also key is to answer if the availability of food resources (as host diversity, nutritional stress, etc.) is one driver underlying these covariations and phenotypic structuring (Donihue and Lambert, 2015; Santangelo et al., 2018; Lambert et al., 2020).

From an epidemiological point of view, the morpho-functional simplification of *T. dimidiata* in the natural-urban gradient could suggest a "domiciliation" event which implies that this species maintains resident urban populations. This actually challenges the view that *T. dimidiata* occurrence is restricted to the warmer months (dry season: March to June) uniformly for the Yucatan peninsula (Dumonteil et al., 2007; Waleckx et al., 2015). Thus, a more realistic estimation of Chagas disease risk should take into consideration the temporal patterns of abundance in the appropriate landscape (Ribeiro et al., 2021). In the region, the potential of *T. dimidiata* to establish colonies has not yet been clearly determined (Ibarra-Cerdeña et al., 2020). For instance, some studies have shown that *T. dimidiata* persists in the city throughout the year (albeit with low densities) with infection rates even higher than those documented in rural communities (48%, Guzmán-Tapia et al., 2007; 90%, Ibarra-Cerdeña et al., in preparation) where presence of abandoned backyards are associated with the house infestation by providing temporary shelter, where synanthropic and domestic animals are available as a source of blood (Guzmán-Tapia et al., 2007). This could be the result of two non-exclusive processes, the urban expansion, and the passive transport of bugs due to rural human migration from the countryside to the city (Chi-Méndez, 2016). Since urban expansion and interstate migration in this region continues to increase (Goujon et al., 2000; Colditz et al., 2017), it is likely that bug infestation will also augment. In any case, our approach can be used to track the process of bug domiciliation.

CONCLUSION

Our study reveals site-specific, simplified morpho-functional architecture that renders similar fitness pay offs in urban environments for an insect disease vector. We admit that selective forces, other than urbanization, may be acting and thus explaining our results. However, urbanization properties are so pervasive that it is hard to think about such other forces. It is necessary to promote and increase research on the "evolvability" of other morpho-functional and "plastic" traits of *T. dimidiata* in urban landscapes. Given a functional specialization in urban areas, there are implications for triatomine control. One example would be that urban populations of this species perceive different wavelengths than those that inhabit rural environments. Related to this, it has been documented an intense attraction of triatomines to incandescent light whose use is typical of many Mexican and Latin American villages (Pacheco-Tucuch et al., 2012). One may learn whether light sensitivity to such light sources is part of the novel adaptive repertoire for urbanized triatomines and, if it is the case, how we can use such information for bug attraction and control.

DATA AVAILABILITY STATEMENT

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

AUTHOR CONTRIBUTIONS

CI-C and ACMO-A contributed to conceptualization, methodology, and wrote original draft. RC-G, AG-M, PI-L, and SS-Á performed the methodology and data collection. ACMO-A performed the analysis. CI-C, ACMO-A, and AC-A reviewed and edited the manuscript. All authors approved the final manuscript for submission.

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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.2022.805040/full#supplementary-material

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Children's Green Infrastructure: Children and Their Rights to Nature and the City

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The development of green spaces in cities has corresponded to a need to deal with a series of socio-environmental and health problems felt in urban spaces. However, these are often fragmented or somewhat disconnected interventions that leave out vulnerable and subaltern groups like children, being also commonly based on strictly formatted designs, with more urban furniture than natural elements. In view of the need to make urban spaces healthier, safer, more resilient, and at the same time more child-friendly, in this Conceptual Analysis paper we build from the literature on Urban Green Spaces, Child-Friendly Cities and environments, and Children's Infrastructure to propose the concept of Children Green Infrastructure (CGI), and discuss its application to urban planning, foregrounding the need for fairer, more inclusive and participatory approaches. GGI derives from the Children Infrastructure concept but it puts at the center of the debate the idea of connecting children to nature where they live, learn and play. CGI is based on the assumption that nature should be transversal in urban planning processes, and that it must be perfectly integrated within urban infrastructures, ensuring access to all. Understanding children's needs and integrating their voices in urban planning and design processes are necessary conditions to moving forward to a fairer, more inclusive and truly collective urban project.

Keywords: children's rights, salutogenesis, inclusive urban planning, green infrastructure, Urban green space

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INTRODUCTION

Significant evidence has pointed out the multiple benefits of Urban Green Spaces (UGS). Much of the debate has been developed on the ecosystem services framework provided by UGS (Haines-Young and Potschin, 2018), related to provision (Kazemi et al., 2018), regulation and maintenance (Mathey et al., 2011; Graça et al., 2018) and cultural services (Jennings et al., 2016; Liotta et al., 2020). Nature exposure also contributes to making us more "immune" to urban stressors, improving mental health outcomes (Vidal et al., 2020b; Lencastre and Farinha Marques, 2021). Beyond these multiple benefits, UGS have been proven to be an avenue to health promotion, namely in deprived communities, also by enhancing social cohesion and a sense of belonging (Jennings and Bamkole, 2019). It is widely acknowledged that urban spaces face socio-environmental inequities, affecting the most vulnerable groups (Hoffimann et al., 2017; Vidal et al., 2021a). Children, as a social category of generational type, are also exposed to a variety of social, economic and environmental risks, limiting their opportunities for agency and development (Mansfield et al., 2021). Although on average, urban children enjoy better access to essential services as well as to cultural, educational and other opportunities for development, this "urban advantage" masks enormous disparities and inequities among urban residents (UNICEF, 2018).

Children's access to quality UGS and their use of the same, is also subjected to these inequities, with income disparities, social class and racial and ethnic belonging all having an important role in this regard (Abercrombie et al., 2008; Boone et al., 2009; Johnson-Gaither, 2011; along with gentrification, commodification and other neoliberal processes with impact in urban planning (Karsten, 2007; Formoso et al., 2010; Karsten and Felder, 2015).[1] Furthermore, the way that UGS are designed results frequently in a limited opportunity for children to fully explore these spaces due to the high presence of urban furniture, instead of natural elements (Woolley, 2008; Ferret, 2021; Vidal et al., 2021b). All these constraints threaten the need to ensure universal access to safe and inclusive green spaces in cities, especially for children, women and unprivileged social groups, as framed by the UN 2030 Agenda (United Nations, 2015). Indeed, children's studies have called attention to the disappearance of urban children from the public arena, relating this to processes of institutionalization, privatization and insularization of space (Zeiher, 2003; Leverett, 2011; Sarmento, 2018) as well as to exacerbated representations of the urban public places as dangerous and children as vulnerable (Gill, 2007; Tomás, 2007). In his book, Gill (2007) presents some social statistics that illustrate this evidence: "In 1971 eight out of ten children aged seven or 8 years went to school on their own. By 1990 this figure had dropped to less than one in ten. Again, in 1971 the average 7-year-old was making trips to their friends or the shops on their own. By 1990 that freedom was being withheld until the age of ten, meaning that in just 19 years children had 'lost' up to 3 years of freedom of movement" (Gill, 2007, p. 12). The author furthers his argument by stating that the amount of time that parents spend looking after their children "... has quadrupled in just 25 years, from 25 minutes per day in 1975 to 99 minutes in 2000, and one of the reasons for this is a fear of letting children play unsupervised" (Gill, 2007, p. 13). On the other hand, recent literature has revealed middle-class preferences for an "urban lifestyle" that includes practices of "family outing" and of "consuming the city" and their public spaces by children and their families (Karsten, 2007; Karsten and Felder, 2015).

Given this evidence, it is undeniable that cities must be planned with and for children, and even more when it is projected that by 2030 more than 60% of the population residing in urban areas will be under 18 years of age (UNICEF, 2018). However, although the studies carried out on the topic of UGS aimed to provide up to date knowledge with the goal of improving urban environmental quality and promoting the wellbeing of the urban population, there has been a neglect of the structural deficiencies and inequalities that are constantly reproduced within the urban fabric (Jennings et al., 2021). Cities are spaces of production and reproduction of inequalities (Lefebvre, 1974), and among these, environmental inequalities are gaining expression amongst urban settlements (Laurent, 2011; Kabisch et al., 2016; Liotta et al., 2020). Moreover, cities tend to be characterized by "... adult-centric legislation, policies, rules and practices that are embedded within social structures and institutions which impact negatively on children's daily lives and result in disadvantage and oppressive social relations." (LeFrançois, 2014, p. 517). This adultism has contributed to the invisibility of children in urban policies and undermined both the realization of children's right to the city and the quality and resilience of the urban fabric since this is also a function of the resilience of its most vulnerable social groups (Castro Seixas and Giacchetta, 2020). As argued by Cordero Arce (2012), a hegemonic children's rights discourse, promoted, in part, by the United Nations Convention on the Rights of the Child, is depicting children as passive actors where their rights are understood as an adult concession. In the same line, Skelton (2007), suggests that UNICEF's concept of children's participation, being framed in an abstract and decontextualized manner, actually contributes to rendering social inequalities and exclusion processes invisible. Thus, although the importance of the Convention on the Rights of the Child as "(...) a legal and symbolic landmark, pointing to a universality of rights for younger citizens" (Sarmento, 2020, p. 23) is indisputable, as it is its acknowledgment of children's rights to participation, there is also a need for a critical reflection on the Convention (Gillett-Swan and Thelander, 2021). As Sarmento, 2020 suggest: "it is necessary to reflect on the improvement of its effectiveness and its own content, which needs to integrate the changes that have taken place in contemporary societies and in the ways of life of children" (p. 23).

Previous studies have shown that children are not provided with the opportunities to contribute to the development of their own environment, thus becoming invisible in the landscape and forced to fit into "unfriendly environments of the adults" (Matthews, 1995; Sutton, 1996; Tsevreni, 2015; Ataol et al., 2019; Zerlina and Sulaiman, 2020). It is against this background that we argue for a shift of the paradigm and intervene in the (infra)structural dimension of the city, including the children in this process. With this goal in view, in this paper, we propose a new concept—Children Green Infrastructure (CGI), and discuss its potential application in urban planning. We suggest that this concept opens new possibilities for the realization of children's rights, namely their right to the city, their right to play and their right to nature. At the same time, CGI foregrounds the need for fairer, more inclusive and truly participatory approaches to urban planning.

TOWARD HEALTHY, CHILD-FRIENDLY, AND INCLUSIVE CITIES

Healthy cities are anchored in the concept of "salutogenesis" (Antonovsky, 1979), which characterizes these as places of

[1] The studies cited in this paragraph do not use, nor build from the UGS concept. In fact, there seems to be two different perspectives in terms of the literature with relevance to this topic: on the one hand, the interdisciplinary studies that build specifically from the UGS concept, and on the other hand, the new children's studies that have given an important contribution to the understanding of children's relationship with urban places and urban nature, and the recognition of children as social and political actors, knowledge-producers and subjects of rights. Our proposal of the Child Green Infrastructure concept builds from the articulation between these two perspectives.

protection from diseases and support for the creation and maintenance of health. The recent experience of the COVID-19 pandemic has highlighted the protective role of UGS, namely those spaces that are fully accessible and provide recreational opportunities (Kleinschroth and Kowarik, 2020; Rodgers, 2020). The multiple benefits for mental health are well documented (Tendais, 2020; Mayen Huerta and Utomo, 2021; Ribeiro et al., 2021), with for example Wortzel et al. (2021) finding that youngers experienced lower COVID-19 related worries as a function of their accessibility to UGS. For children, healthy cities are places where they have the freedom to play, explore and socialize, without restrictions or constraints (Kyttä, 2004). In this regard, encouraging street play by improving the streets and spaces next to the children's homes helps promoting the integrative development of children (Thaler and Sunstein, 2008) and realizing children's right to play (Davey and Lundy, 2011). Children can find affordances for play in both formal, planned spaces and informal/unmanaged places (Jansson et al., 2016). The multifunctionality and richness of planned spaces are particularly appreciated by children, but unmanaged areas are also valued for exploration, play and creating "children's places" (Jansson et al., 2016). With Brown et al. (2019), we see "the focus on child-friendly cities as a valuable entry point for integrated healthy city commitment, policy and action, as set out at the foundation of the WHO Healthy Cities initiative" (p. 1). Thus, a child-friendly city is a city that is healthy not just for children but for all citizens.

In the wake of the U.N. General Assembly's adoption of the Convention on the Rights of the Child (CRC) in 1989, a rights-based approach to creating child-friendly environments prevailed, giving rise to several important UN initiatives such as the UNICEF Child-Friendly Cities (CFC). CFC framework builds from a holistic perspective of children's rights as comprising both their access to urban resources (rights in the city), and to meaningful participation in urban governance (rights to the city), and it has given wider visibility to the need for integrating children's rights into decision-making and city governance. A child-friendly city should promote children's rights, and provide safe spaces to play, allowing for a strong connection with nature, encouraging independent mobility and, above all, including children in the processes of reformulation and design of urban places and policies (Brown et al., 2019). UGS, especially those with public access, have the potential to promote a healthy lifestyle for children (Dadvand et al., 2019), and are important in fostering children's integrative development and wellbeing (Fjørtoft, 2004; Faber Taylor and Kuo, 2009; Ward et al., 2016; Sarmento, 2018; Neto, 2020; Fjørtoft et al., 2021; Ito, 2021). A Danish study (Engemann et al., 2018) has showed that children who lived in environments without green spaces had increased risk for psychosis, compared to those who lived in green areas. Furthermore, literature on biophilia (Wilson, 1984) has suggested that humans have a spontaneous relationship or predisposition to connect with nature, whose intensity varies according to their exposure. This hypothesis has been extended to children (Kahn, 1997; Keith et al., 2021) and supported by studies on children's development showing that throughout their childhood, children experience important milestones related to nature connectedness, which emphasize the relationship with nature in multiple ways (Neaum, 2010; Svetlova et al., 2010; Bensalah et al., 2016), namely by the interaction of children with non-human species that starts in early childhood (DeLoache et al., 2010). Nonetheless, some authors have also suggested that this nature connectedness of children can be lost or perhaps it is not innate after all and is instead learnt and depends on nature exposure (Hand et al., 2017).

In the last 10 years, other international initiatives were designed to promote the inclusion of children, their voices and perspectives in city planning: Urban95, which sought to re-imagine cities from 95 cm tall (Vincelot, 2019), or the publication "Cities alive: designing for urban childhoods" (ARUP, 2017) that put children in the spotlight to respond to the main social and environmental challenges in contemporary cities. Nevertheless, and despite its importance in giving visibility to children, their rights and capabilities, the UNICEF CFC framework has also proved to be insufficient and liable to co-option in many cases (Racelis and Aguirre, 2002, 2006). Co-option is facilitated by the fact that there is no single definition of what a child-friendly city is or ought to be, and therefore, different conceptualizations and approaches to CFC emerge within the context of neoliberal globalization. In fact, the idea of what a child-friendly city is might vary according to different cultural and socio-economic contexts. Van Vliet and Karsten's proposal (van Vliet and Karsten, 2015) is key in this respect as it reveals the different approaches to children's relationship with the city, with children being framed as consumers, users, entrepreneurs and/or producers. Moreover, although there may be common features of child-friendly environments, there are sociocultural variations in the subjective experiencing of these features and in their contribution for the resilience of children (Derr et al., 2019). The main focus of child-friendly cities initiatives also depends on the economic status of the country where it is applied, as for high-income nations the focus has been on improving the quality of spaces available for children in the city, while in low-income nations priority has been given to survival issues, including access to basic services and children's safety and security (Malone, 2016). These differences point to the importance of taking into account the context, and listening to children and their families within the process of building child-friendly cities and environments. Horelli's (2006, cit in Haikkola et al., 2007, p. 322) definition of environmental child-friendliness is relevant here as the author considers it to be "a complex multi-dimensional and multi-level concept", which "refers to settings and environmental structures that provide support to individual children and groups who take an interest in children's issues, so that children can construct and implement their goals or projects." Although participation of children is part of the framework, this again is no guarantee of protection against co-option and manipulation of children within the process.

When thinking of UGS, we are not limiting these to the traditional spaces of urban parks but rather, as Pincetl and Gearin (2005) suggest, use this as a broader concept as children also value informal and unmanaged areas for their free play and exploration (Jansson et al., 2016). The possibility of manipulation of environments, both unmanaged and managed, needs to become recognized as part of children's play and met with understanding among managers who must deal with the different perspectives on places among adults and children (Jansson et al., 2016).

Current investments in UGS are not without problems. In this regard it is central to ensure that, when allocating green space to a certain area, the long-term and low-income residents are not displaced. This is a key issue for policymakers as they seek to balance the positive effects of green space allocation and the negative effects of eco-gentrification (Jo Black and Richards, 2020). Eco-gentrification affects mainly those from deprived communities, who then see both their rights to nature, and to housing threatened. In this context, it is worth mentioning the theory of "Just Green Enough" proposed by Curran and Hamilton (2018), which aims to reverse this trend by allowing communities to design their own environmental initiatives, and preventing the expulsion of the most disadvantaged from these re-qualified places.

Public and open green spaces may lead to the opportunity to develop outdoor education programs, which have been shown to have significant effects on learning (Hamilton, 2017), contributing also to the diminution of behavior and socialization problems (Chiumento et al., 2018; Engemann et al., 2019), improving cognitive development (McCormick, 2017), enhancing prosocial behavior (Putra et al., 2020) and reducing the risk of diseases characteristic of urban societies (McCracken et al., 2016; Roslund et al., 2020). More importantly, the planning process can itself be place-based and place-conscious, in a way that values local knowledge and it is done with and for the local community (Villanueva et al., 2016; Lloyd et al., 2018).

MINDING THE GAP: CHILDREN'S GREEN INFRASTRUCTURE

As stated before, UGS are vital for promoting safe and healthy urban spaces, especially for children (Vidal et al., 2020a). Nevertheless, it is worth remarking that for a city to be healthy, it must first be fair and inclusive. And to be fair and inclusive, the city needs to be a collective construction of several voices. Kalache and Kickbusch (1997) suggest that the foundations for healthy living are established in the first years of life. Therefore, urban planning aimed at promoting a healthy environment for children brings benefits in the short, medium and long term. However, in general, children have seldom been called to participate in the processes of urban planning and design (Bishop and Corkery, 2017). This, in spite of the above mentioned UNICEF programs as well as evidence from the new sociology of childhood/children's studies that have shown how children are capable of a critical understanding of place and of making good and realistic contributions to urban planning and policy (Cele and van der Burgt, 2015; Jansson et al., 2016; Ataol et al., 2019; Hanssen, 2019; Mansfield et al., 2021).

Given the ineffectiveness of traditional urban planning (Rittel and Webber, 1973; Rakodi, 2001; Campos, 2015;, the design of the cities by children can be a solution for promoting inclusive values (Krishnamurthy, 2019). However, that may not happen if, as stated before, these are planned as restricted and over-structured places. Power of imagination is a crucial skill that UGS should enhance. However, children are less likely to develop their imagination and fantasy within restricted and over-structured places, strictly formatted with more urban furniture than natural elements (Woolley, 2008; Ferret, 2021; Vidal et al., 2021b). For that reason, there is an evident need to develop an integrated solution that enables children to fully explore cities' spaces. It was in this context that ARUP (2017) proposed the concept of "Child Infrastructure" (CI) to refer to a network of spaces, streets, nature and interventions focused on the city capacity to attract children and remain healthy. This concept goes beyond designated spaces for children like playgrounds, defending an expanded infrastructure that is properly integrated into the multifunctional urban fabric. This proposal aims to place children at the heart of urban planning and to improve the everyday lives of cities' younger residents through intervention at the neighborhood level. Based on this concept, the Gehl Institute (2017) proposed ten principles for the implementation of a CI to combine the accessibility of activities for different ages with daily routes on safer, more welcoming and user-friendly streets, in addition to a connection with nature: (i) give visibility to children and caregivers; (ii) promote curiosity: (iii) encourage children to get dirty; (iv) improve spaces close to their homes; (v) encourage playing in the street; (vi) promoting collective responsibility for children; (vii) develop a community co-creation; (viii) work without borders; (ix) monitor to know where to improve; and (x) strengthen the best ideas. Building from this concept, we suggest that CGI may help reducing the risks for children living in cities not only by promoting the development of safe spaces and routes where cars traffic is reduced or non-existent, but also by allowing children to feel comfortable and encourage independent mobility within urban spaces. This can only be achieved by looking at urban nature experiences as well as security through children's perspectives, and preventing the adultcentric bias of current urban planning.

Infrastructure can be understood as a political, technological and discursive technology of state governance (Kooy and Bakker, 2008). However, agreeing with Berlant (2016, p. 393), "Infrastructure is not identical to system or structure, as we currently see them, because infrastructure is defined by the movement or patterning of social form." In this sense, infrastructure can be seen as a "living mediation of what organizes life: the lifeworld of structure. Roads, bridges, schools, food chains, finance systems, prisons, families, districts, norms all the systems that link ongoing proximity to being in a world-sustaining relation" (Berlant, 2016, p. 393). According to the (European Commission, 2013, p. 3), Green infrastructure is "…a strategically planned network of natural and semi-natural areas with other environmental features designed and managed to deliver a wide range of ecosystem services." This network is responsible for the delivery of health-related and social benefits, namely through the creation of a sense of community and by reducing social exclusion and isolation. Notwithstanding the importance of this concept, a growing body of evidence has focused on infrastructures as emergent social-ecological-technological systems that link more-than-human agencies with social processes and technological systems (Star and Ruhleder, 1996; Grabowski et al., 2017; Markolf et al., 2018). Considering that infrastructures are "networks that facilitate the flow of goods, people, or ideas and allow for their exchange over space" (Larkin, 2013, p. 328), it is also possible that infrastructure users can redefine what infrastructure is for, i.e., an unfinished process that allows a co-creation interaction among children, adults and institutions. This aspect can also be related to the idea of "civic infrastructure" as key to the goal of making urban planning a truly inclusive and democratic process. Civic infrastructure can be understood as "formal and informal institutional as well as sociocultural means of connectivity used in knowledge–action collaboration and networking" (Pezzoli, 2018, p. 192). In times of increasing privatization of infrastructure (Viitanen and Kingston, 2014), civic infrastructure is a key notion to put the focus back on community participation and citizenship rights (Perng and Maalsen, 2020). Children's citizenship is particularly undermined by a dominant discursive rationality paradigm of citizen's participation, which also devalues local knowledge (Davies et al., 2012). CGI can benefit from the aggregation of Green and Civic infrastructures through the development of a network that links society to nature in urban areas (Green) through a co-creation and civic participation (Civic) approach (Ito et al., 2016).

The accessibility discussion on UGS is related to the distribution (and availability) of these spaces in the city, but also to mobility, with accessibility being often defined as the availability of green spaces within a short walking distance of the residence area (Boone et al., 2009). However, assessing UGS remains a complex and controversial issue, namely because these spaces are highly heterogeneous in terms of size, design, available structures and services, quality and safety (Wolch et al., 2014). The fact that there are accessible UGS does not mean that they are of quality, safe and suitable for the populations residing in that area, neither, as Iveson (2007) points out, the access to these spaces from a topographical perspective guarantees status as a member of the public, nor the possibility of participating in public actions. Bristol City Council (2008) provided some recommendations regarding the availability of green spaces. The first concerns its quality and represents the main priority. UGS must have the quality to meet the needs of users, where children are included. The second refers to the distance between green spaces and residential/school areas. The third is related to the availability of green spaces in a given geographic area. The relevance of these priorities is based on the principle that quantity should not be the main criterion. Quality and accessibility (distance) are the main priorities to promote democratization in access to green spaces. And the evaluation of these two criteria has to be done with and for children.

Despite the importance of the CI concept and the literature on the UGS and their accessibility, as presented above, there is an urgent need for a cross-cutting dialogue between these perspectives and the CFC experiences. It is with this aim in mind that we propose the concept of CGI (**Figure 1**). In the center of the model are framed the main pillars that support CGI. Thus, the process of co-creation should be the basis for creating quality and accessible UGS that promote children's connectivity to nature from early childhood. Around them, are the main objectives that can be achieved through the four pillars, which are interconnected and mutually influenced. By developing stimulating natural elements and fostering intra and inter-generational socialization CGI should promote space exploration and imagination skills. CGI should also contribute to the development of open and unstructured natural areas, and their inter-connectivity. But for CGI to be effective, children must also be provided more free time to play and the possibility to play outdoors. In the larger external circle are the pivotal principles that guide CGI, namely the close link it holds with Children's Rights that frames it. Key to CGI are also the principles of inclusiveness, integrativeness and fairness (of both the urban planning process and the green spaces that are created) and safeness of these green spaces. These have to be understood from the perspective of the children as to prevent adultism bias, but their meaning varies according to the context, as explained before.

CGI is suggested here as an open concept, giving its users, namely children, the possibility to redefine their meaning and elements, within the framework of children's rights. Although this concept derives from the CI concept, it places the idea of connecting children to nature at the center, focusing on the context where they live, learn and play. Looking back, over the past years, due to the intense urbanization process, increased criminality and changing lifestyles and habits, childhood has moved indoors, generating a disconnection from the natural world and a nature-deficit disorder (Phenice and Griffore, 2003; Louv, 2005; Karsten and Felder, 2015; Tsevreni, 2015; Sarmento, 2018). Nonetheless, a new social dynamic is gaining expression related to the gentrification phenomenon, where middle class families are beginning to reclaim city centers (Lilius, 2019), namely the 'YUPPs'—young, urban professional parents—who are actively choosing to live in city centers, rather than taking the traditional route of moving out to the suburbs as soon as they have children" (Karsten, 2014, p. 14). This process has, on the other hand, led to increased practices of "consuming the city" by children and their families (Karsten and Felder, 2015), and cannot be dissociated from the commodification and privatization of urban public spaces. The latter processes have transformed urban public places as well as children's relationship with the city and urban nature since those living in cities, report fewer nature interactions compared with rural counterparts (Collado et al., 2015).

CGI proposal assumes that when cities become places of nearby nature connection, children, families and the environment thrive. CGI is embodied by the Convention on the Rights of the Child (United Nations General Assembly, 1989). It thus recognizes children's rights to freedom of expression, to be heard and to participate in the decisions that affect them. It also builds from the literature briefly presented in this paper and concerning children's rights to play and to nature as well as their right to the city. However, the inclusion of children in urban planning is a complex task that should be realized through a co-creation approach. This becomes more obvious when current evidence shows that children without nature exposure experiences may have little empathy with or interest in nature (Kaplan, 1989). Despite this, and as previously stated;

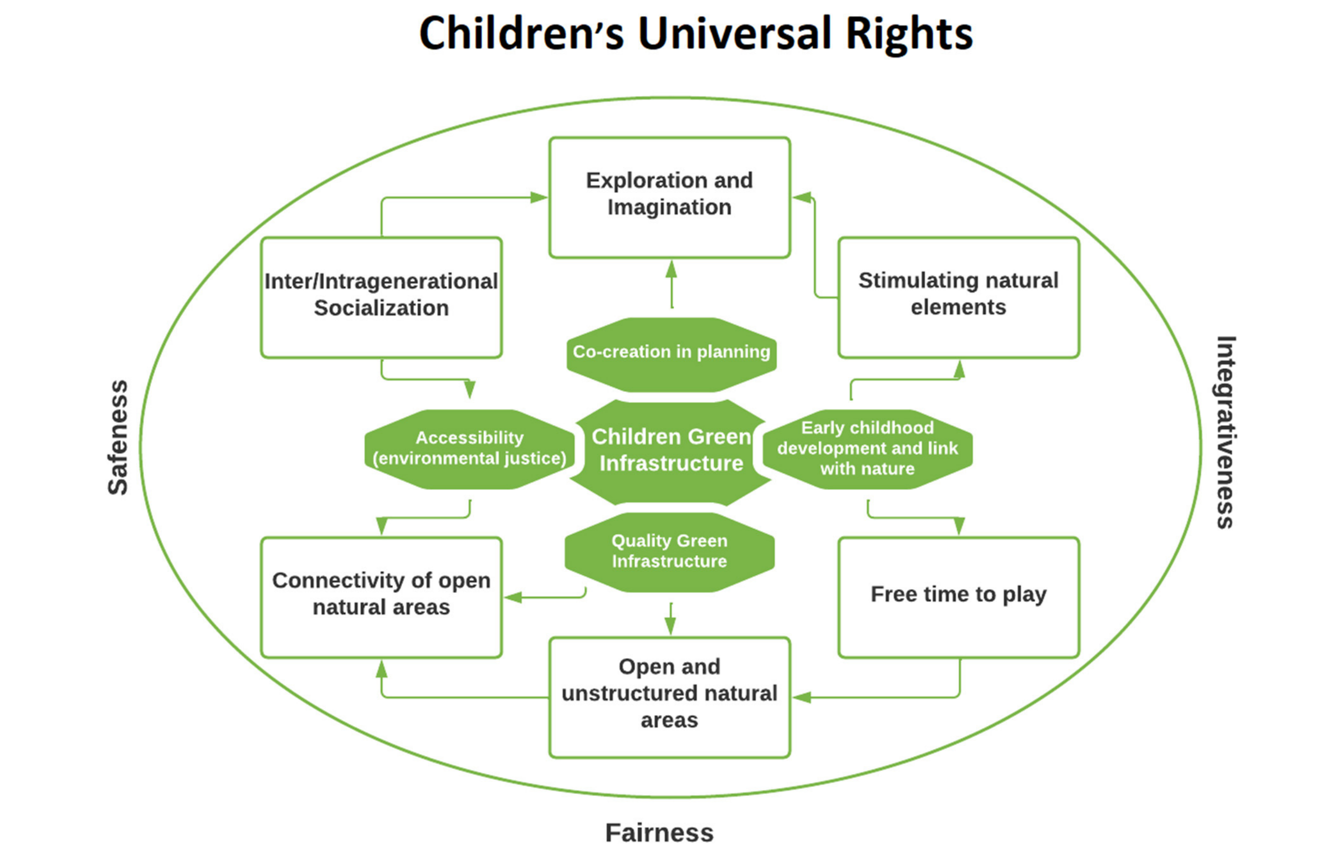

FIGURE 1 | Children green infrastructure model proposed by the authors.

"most children have a natural affinity with nature" (Gill, 2011, p. 8). This means that even though some children have less empathy with nature, if stimulated, they can develop a deeper connection with it. Also, Freeman et al. (2015) argue that the concept of "nature deficit disorder" proposed by Louv (2005) "is adult-determined, and ignores the richness of biodiversity and associated nature connection opportunities available to children within many urban landscapes." Instead, the authors argue for "a more positive interpretation" in order to explore "whether children's urban lives are and can be nature-rich", also because what may appear to an adult to be a nature-deficit may in fact be nature-rich from the child's perspective (Louv, 2005, p. 179).

CGI can act in both ways: firstly, by integrating children's preferences and motivations regarding natural elements in city planning and design, but also their own experiences and perceptions and how they interact with nature in their immediate environment. Secondly, through this integration of children's perspectives, it will be possible to create stimulating and more diverse open nature areas interconnected in cities, benefiting particularly children with poor access to formal UGS. Therefore, CGI is here understood as a place-based community-focused continuous and unfinished process whereby both adults and children can participate. What is important is to ensure that children (and other members of society) have the possibility to influence the final outcome through a democratic /participatory and co-design process, meaning that their voices are not only heard but taken into consideration in the final decisions, meeting children's right to participation as enshrined in article 12 of the Convention on the Rights of the Child (CRC).

At the center of CGI is the need to rewild cities, through a co-creation process where children are included and their voices, needs and rights are taken into consideration. The benefits of rewilding cities go far beyond the potential to promote biodiversity recovery and sustainable development. Due to their informal and unrestricted character, rewilding cities provides stimulating natural elements which enhance the exploration and imagination skills of children (Henderson and Vikander, 2008; Gurholt and Sanderud, 2016; Bento and Dias, 2017). Also, rewilding means that nature takes its course and wildlife can flourish (Xie and Bulkeley, 2020; Lehmann, 2021). This solution may lead the opportunity to reintroduce lost biodiversity back into cities spaces and promote inter and intragenerational communication and a closer connection with nature, goals that can be associated with the Planetary Boundaries proposed by Steffen et al. (2015), and its application to city level proposed by Hoornweg et al. (2016). Planetary boundaries cannot be directly applied at a local level, but an effort should be made since cities are key drivers and those most impacted by global influences. Rewilding allows for connecting small and medium natural areas into a continuum, limiting the negative effects of fragmentation, minimizing physical and symbolic distance for children to these areas. Also, CGI may help to address biodiversity loss and extinction through the creation of new habitats to plants and animals.

CGI aims to place nature where children are present, in an equitable manner. Processes of neoliberal urbanization have increased urban inequality regarding resources' distribution, including nature and UGS provision (Authier and Lehman-Frisch, 2013; Lang and Rothenberg, 2017; Karsten, 2020). Since structural deficiencies are harsh to deal with, policymakers and stakeholders must use the power of imagination and creativity to add quality and child-friendly natural features where is possible. If children spend most of the time at school, this process must include transforming schoolyards into green areas or providing nature play options close to childhood centers, adapting vacant lots and streetscapes and opening community gardens. However, it is also important to try to reduce the duration of classes and the amount of homework that children are given and which consumes most of their free time.

More difficult is to establish green corridors aiming to, first, connect children daily routes—home, school, extracurricular activities—and, second, connect the available natural areas in the city. In order for children to walk or bike to school, one needs to create an infrastructure that enables them to do this route in a safe, interesting and playful way, which implies avoiding the conflict with automobile traffic (Becker et al., 2018). In fact, and according to several authors, the increased mobility for adults, namely by the use of the car, resulted in the reduction of children's mobility (Parr, 1967; Engwicht, 1992; Tranter and Sharpe, 2008). In this line, Cervero et al. (2017) developed the concept of kid-friendly Transit-oriented developments (TOD), arguing that planning cities should have the needs of children in mind. Kid-friendly TOD comprise lush and green communal gardens, playgrounds, tot-lots, and play-inviting open space, where surface parking is progressively replaced by gardens and play areas. But this is not enough, as we also need to change adults' minds so that they let their children walk or bike to school, because even if the routes or the infrastructures are safe, their representation may not be. Hence, we have to look at the infrastructure as not only a material thing, but as an imagined and socially constructed place in a world where adults have power over children and their mobility.

The model here proposed through CGI rejects the KFC design (Kit, Fence, Carpet) (Woolley and Lowe, 2013), or even the presence of just one of these elements—with an emphasis on the fences and a standardized children's equipment kit, which is neither flexible nor very stimulating for the children (Castro Seixas et al., 2022). Unstructured open and natural areas can stimulate children's sociospatial exploration and imagination and as Duhn et al. (2017) say, help "troubling the intersections of nature/urban/childhood" (p. 1358).

Inclusiveness is another key aspect of CGI. Our understanding of inclusiveness goes beyond the age variable to ensure the participation of children who are systematically excluded and not just those who have reduced mobility, who have already attracted some attention from designers. We should focus also on including street children, children from poor and stigmatized neighborhoods, institutionalized children, children with visual, hearing, cognitive disabilities and children with mental disorders. This is of utmost importance since children are also victims of social crimes and, poorly lighted "wild green spaces" could present a risk in this regard (Lyytimäki et al., 2008; Matzopoulos et al., 2020). CGI can help to minimize this risk by creating a dynamic and interconnected network through a green infrastructure. As Wenger et al. (2021) state, inclusion is a complex process that involves an interaction between physical (related to the UGS design), social (related to norms and attitudes) and political (related to options and regulations) dimensions. Social class, ethnicity, skin color, racial and gender identity are other important variables to be considered, which leads us to an analysis of how these open and green spaces can constitute a microcosm of society power relations. Indeed, there is some evidence that different social and ethnic groups value different characteristics of UGS and also use the space differently (Loukaitou-Sideris and Sideris, 2010; Özgüner, 2011; Vidal et al., 2022).

CGI also calls for intra and intergenerational socialization among children, young people and adults as well as between children and non-human species. The inclusiveness and integrativeness of CGI could be somewhat related to the concept of Intergenerational Contact Zones (Kaplan et al., 2020), which appeals to a co-creation process among all generations. Azevedo's proposal (Azevedo, 2020) of an intergenerational space can be easily adapted to the development of CGI, in accordance also to a place-based and place-conscious perspective that values local knowledge and community participation (Malone, 2016; Villanueva et al., 2016), including the knowledge and participation of children and young people. In the case of CGI, it is believed that cocreation between communities and decision-makers (Sanders and Stappers, 2008; Lund, 2018; Šuklje Erjavec and Ruchinskaya, 2019; Costa et al., 2020) may better promote children's participation in urban planning and design process. Furthermore, and of utmost importance, community involvement goes far beyond the creation of CGI. As a continuous unfinished process, the success of CGI implies the commitment of the whole community to assure the sustainability of the initiative. CGI calls for a universal commitment of all adults and institutions for the inclusion of children in urban planning processes.

FINAL REMARKS

In this paper we have built from the literature on UGS, Child-Friendly Cities and environments and Children's Infrastructure to propose the new concept of Children Green Infrastructure (CGI). Our conceptual model is framed upon Children's Rights, although we also recognize a need to critically review official conceptualizations of children's rights as enshrined in the CRC.

We have presented CGI as anchored on several principles that have been highlighted by the literature here reviewed, namely the principle of inclusiveness, which foregrounds the need the include children in urban planning processes, and among these, the most marginalized and excluded children, and the principles of safety, accessibility and quality of the green infrastructures, understanding these features from children's perspectives. This child-centered approach should thus prevent current urban planning adult-centrism and promote children's rights of participation.

As stated before, CGI aims to promote children's rights of participation, to nature and to play. Nevertheless, we believe this model could have beneficial social, environmental and health outcomes for all, namely by helping to rewild cities, providing quality, accessible and child-friendly green infrastructures and fostering inter-generational dialogue and socialization. Indeed, rewilding cities and enhancing biodiversity are essential processes for the promotion of CGI. This perspective implies a different approach than the one inspiring the creation of designated spaces for children, which tend to be highly structured, restricted and regulated, conditioning the exploration of space and the free play of children, especially for the most vulnerable ones (Lehmann, 2021). CGI, through its civic and participatory focus, aims to co-create natural areas by rewilding cities and creating open natural spaces that can be appropriated by children as their own, because children also participate in their design, together with adults and young people. Rewild cities is also a process to reintroduce nature which provides a wide range of benefits to tackle the biodiversity and climate crises in a balanced way that fosters humans and wildlife needs to create better urban landscapes for all.

Access to quality and child-friendly green infrastructures is unequally distributed. The concept here proposed of CGI should be translated in the mission to reconnect children with nature from where they live, learn and play. This implies including the diversity of children's voices in the urban planning process, to create healthier and more inclusive and resilient cities for all to live, grow up in, and realize their full potential. CGI appeals to the need to shape city planning and urban design around children's needs but also their rights, including their right to nature, their right to play and their right to place-making and to participation in urban policies. At the same time though, CGI aims to develop spaces and design processes that promote inter-generational dialogue and socialization, that recognize children as social and political subjects and knowledge producers, capable of contributing to urban design processes together with adults. The green infrastructures created through this approach should also be inter-generational spaces that do not separate children from adults.

Finally, the proposed model should also be considered as a motivation to pursue further research and policy development. In this regard, there are many under-explored areas, such as investigating children's—and especially marginalized children's perspectives and experiences with urban nature; developing child-friendly participation processes and developing methods for fostering inter-generational dialogue, as well as promoting effective dialogue between lay people, experts and policy-makers within urban planning processes. The meaning of the aspects highlighted in the model will be different according to the geographical, historical, cultural and socioeconomic contexts and these aspects also need further analysis. The relevance of some of these aspects may speaks more to the context of post-industrial neoliberal cities of the "Global North", where the need for promoting children's connection with nature from early childhood as well as children's free time to play and to do so in open and unstructured natural areas is more significantly felt. Nonetheless, questions of deep social inequalities, child's labor and child's participation in armed conflicts, more salient in the "Global South" also prevent children's right, and available time to play, by threatening the very right to childhood. Although children's right to the city, to nature and to play relate to different questions and have different expressions according to the context, we believe the value of CGI remains universal.

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

DV and EC contributed to conception and design of the study. DV wrote the first draft of the manuscript. EC wrote sections of the manuscript. All authors have contributed to manuscript's final version, read, and approved the submitted version.

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Exploring the Integration Between Colour Theory and Biodiversity Values in the Design of Living Walls

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Designing green infrastructure in cities requires vegetation that has multiple outcomes and functions, particularly using plants that have both attractive visual or aesthetic features and high biodiversity values. Plantings that have high visual appeal are more highly valued by people and increase their feeling of wellbeing. Increasing biodiversity in cities is one of the major challenges facing urban planning and design. However, balancing biodiversity and aesthetic outcomes in urban planting design is complex, and to date there are few methods that can be used to guide plant selection. To address this knowledge gap, we investigated the use of a colour theory framework for planting arrangements to see if we could design vegetation that is highly aesthetic and has high biodiversity. We did this by configuring planting combinations for living walls in Malmö, Sweden, using principles based on Johannes Itten's colour theories. The plant combinations on each wall were graphically arranged using (1) colour analysis of each plant and (2) design of the plant species into two colour schemes: light-dark colour concept and a complementary colour concept. For each species used in the compositions we created a biodiversity classification, based on its pollination value, "nativeness" and conservation value as a cultivar; and a plant visual quality classification, based on the performance from living walls studies. The graphical colour composition and interlinked biodiversity value were then compared to designs created with randomly selected plant species. The results showed that it is possible to design a living wall based on colour theory without compromising with biodiversity outcomes, namely species richness, pollination and the nativeness of the species. The results also indicate the potential application of this design approach to deliver greater aesthetic appreciation and enjoyment from plantings. While more work is needed, this study has shown that a theoretical colour framework can be a useful tool in designing green infrastructure to improve delivery of both cultural and regulatory ecosystem services.

Keywords: green infrastructure, landscape aesthetics, living wall, green wall, ecosystem services pollinators, environmental appraisal

INTRODUCTION

Urban vegetation can be designed to deliver biodiversity outcomes and provide regulatory ecosystem services, particularly through green infrastructure. In addition, visual qualities derived from urban vegetation can provide aesthetic enjoyment and appreciation, delivering cultural ecosystem services, but are dependent on socio-behavioural pathways for positive public health effects (van den Bosch and Ode Sang, 2017).

Perceived visual quality and visual satisfaction in outdoor environments derives from a viewed landscape and can be evaluated from a formal aesthetic approach where the landscape quality or beauty is seen as an attribute in the eye of the beholder (Lothian, 1999). Perceived attributes, such as landform, water bodies and colours, are part of the formal aesthetic approach and are included in the objectivist paradigm (Daniel and Vining, 1983; Lothian, 1999). Colour as a property has a broad spectrum, and in landscape design compositions the attribute "colour" has its origins in design theory and design principles, sometimes using gestalt theory as an evaluation tool (Lang, 1994; Bell, 2012; Yilmaz et al., 2018).

Yilmaz et al. (2018) states that landscape professionals can influence visual qualities and viewer's aesthetic satisfaction via careful use of design principles, in turn contributing to positively affect the visual quality (Ulrich, 1983; Arriaza et al., 2004; Polat and Akay, 2015) and evoke positive restorative effects (Hoyle et al., 2017). In the planning and design of green infrastructure the aesthetic property colour and it's physical attribute, contrast, give variation to planting composition schemes (Bell, 2004; Dunnett, 2019). A conscious use of colour and colour contrast in the design of urban green installations can improve human experiences of pleasantness (Thorpert, 2019), provide positive visual preferences (Polat and Akay, 2015) and contribute to increased experiences of beauty and harmony (Eroğlu et al., 2012; Oleksiichenko et al., 2018; Huang and Lin, 2019). A common strategy to achieve variation and distinction through colour contrast in compositions is to use disparity in the brightness level, as well as the use of complementary contrast in design compositions (Robinson, 2004; Bell, 2012). For example, the concept of complementary contrast is an effective design parameter in landscape architecture and has proven to contribute to harmonious arrangements (Westland et al., 2007; Bell, 2012) and increase the correlation between visual preferences and complementary colour contrast in urban green spaces (Polat and Akay, 2015). In the context of colour and visual preferences, it is important to highlight that personal attitudes, biases and cultural background or variations in value systems all contribute to how we perceive colours (Patton, 2002). A study by Kendal et al. (2012) showed that diversity in visual preferences depends on features such as foliage colour and flower size as well as non-visual characteristics such as drought tolerance and "nativeness."

Biodiversity, according to Wilcox (1984), is defined as the variation of life forms on earth, including variation in genes, species and biotopes, and functions from factors, the species richness and the distribution of the individuals among these species, the evenness. High species diversity positively affects ecosystems and human life (Hooper et al., 2005, 2012), while biodiversity loss is recognised as one of the most severe human induced global challenges (Rockström et al., 2009). Biodiversity loss causes ecosystems to capture less resources, reducing biomass production and causing instability in the ecosystem (Cardinale et al., 2012) and can mitigate invasive species (Naeem et al., 2000; Kennedy et al., 2002); changes which can be irreversible or extremely difficult and expensive to alter. Biodiversity in urban environments differs from that in rural environments; urbanisation tends to decrease biodiversity (Knapp et al., 2008), especially in city centres (McKinney, 2002). Areas with only moderate urbanisation can show an increase in biodiversity, primarily due to the introduction of non-native species being faster than the extinction of native species (Kowarik, 1995). This increase may also be due to factors like intrinsic high plant species richness in settlement areas, intermediate disturbance theory and introduction by species (McKinney, 2008), resulting in urbanisation leading to homogenisation of native plant species (Kühn and Klotz, 2006).

Urban areas are nowadays a hotspot for threatened species (Ives et al., 2016) and species conservation in these areas can be seen as a practical response to mitigate the loss of species globally (Kowarik, 2011). This is resulting in significant efforts in habitat creation to counteract biodiversity loss in rural and agricultural environments (Knapp et al., 2008). A systematic review by Berthon et al. (2021) found that most studies indicate a positive influence of native plant species on at least one measure of biodiversity, justifying their priority in urban green settings to support native animals. While conservation of native plant species is increasingly important, the conservation of culturally valuable species is also important in the context of heritage values. Especially the conservation of genetic variation within horticultural species, to find varieties that may be more useful in a future changed climate.

Biological diversity includes among other aspects to rehabilitate and restore degraded ecosystems and support the recovery of threatened species, for example through the conscious use of development and implementation of management strategies (Convention on biological diversity, 2021). It is also important to pay attention to the role of local and traditional knowledge in the conservation and sustainable use of biodiversity *in situ* where health is connected to biological diversity, a basic human right, and an important indicator of sustainable development, providing ecosystem services essential to human health and wellbeing (Convention on biological diversity, 2021).

The decline of insects, especially pollinators, is a major factor in the decline of global biodiversity (Goulson, 2019; Wagner et al., 2021). As a result, more attention is being given to the creation of urban habitats that can sustain more pollinators and increase biodiversity (Baldock et al., 2015). Green infrastructure technologies such as green roofs and walls are increasingly being used to support more urban biodiversity (Collins et al., 2017; Mayrand and Clergeau, 2018; Filazzola et al., 2019). Green walls have been shown to benefit birds (Chiquet et al., 2013) and insects (Madre et al., 2015; Ridzuan et al., 2021), although the number of studies regarding pollinators is limited (Chiquet, 2014; Dover, 2015; Ridzuan et al., 2021). Vegetation traits identified as having a positive impact on insect populations include flowering (Ridzuan et al., 2021) and foliage (Chiquet, 2014). Designing green infrastructure installations such as green- and living walls with greater species diversity and with species that support insects could be a way used to increase biodiversity in urban areas and is a major research question in this study.

Vertical greening is categorised as either facade greening, climbing plants growing on a building facade (Perini et al., 2013); or living walls, plants growing directly on a wall surface in either felt-based hydroponic systems or soil-cell modular systems (Riley, 2017). Living walls are becoming difficult to define, due to differences in components, materials and vegetation, as well as advances in design and technology (Perini et al., 2013; Manso and Castro-Gomes, 2015; Riley, 2017; Bustami et al., 2018). Regardless of these differences, living walls are increasingly being used in urban environments because of the many benefits they can provide. These include greater building thermal performance, particularly cooling (Wong et al., 2010; Chen et al., 2013), improvements in thermal comfort and carbon sequestration (Charoenkit and Yiemwattana, 2016), reductions in noise (Wong et al., 2010; Azkorra et al., 2015) and air pollution (Weerakkody et al., 2017), opportunities for localised food production (Nagle et al., 2017; Mårtensson et al., 2014) and improvements to urban hydrology, including grey water reuse (Fowdar et al., 2017). Much of the research into living walls has focussed on plant growth and performance (Riley, 2017), including species across a range of groups and climates (Mårtensson et al., 2014, 2016; Jørgensen et al., 2018; Dvorak et al., 2021) and the potential to support urban biodiversity (Collins et al., 2017; Filazzola et al., 2019). While there have been studies exploring the broader social benefits of living walls (Pérez-Urrestarazu et al., 2017; Bustami et al., 2018), the analysis of their aesthetic values remains a major gap in the literature (Radić et al., 2019).

Living walls are also a useful experimental prototype to examine individual plant performance and different planting designs. Most living wall systems can be adjusted, in terms of irrigation and nutrition, to provide optimum conditions for growing less common species, including native plants (Dvorak et al., 2021) and those with high biodiversity values (Mayrand and Clergeau, 2018). The vertical layout of living walls means that novel plant communities can also be configured under what are fairly uniform conditions, with competition restricted to a few neighbouring plants, with growing conditions that are easy to manipulate. The evaluation of species traits or attributes related to visual traits or to biodiversity and cultural values can be made through short growth experiments.

This study asks the question—can a living wall be designed to be aesthetically pleasing according to Johannes Ittens's colour theory and at the same time support high biodiversity outcomes? The study examines and theoretically analyses if it is possible to design living walls that contain high biodiversity and at the same time obtain an appealing aesthetic expression from a colour theoretical point of view using Johannes Itten's concepts light-dark- and complementary contrast as interpretation models (2002:36).

MATERIALS AND METHODS

To collect data for interpretation of the graphically arranged design models we completed *in situ* plant growth trials, made plant colour observations *in situ*, created theoretical modelling of colour contrast interpretations models and calculated biodiversity outcomes of selected plant species. Three biodiversity factors were calculated from the selected plants and linked to the two redesigned conceptual colour contrast models, pollination value, "nativeness" and valuable cultivars. The individual plant visual performance data was derived from *in situ* living walls based on classification of classifying individual plants growth, vitality and survival.

We used a hypergeometric distribution to test if the biodiversity outcome in the two colour contrast interpretation models (light-dark and complementary) differed from a random choice of available plants. We also tested, through simulation, the two colour conceptual models to see if the used plant species visual performance were different from a random colour model. Finally, we combined calculated biodiversity classification value with the visual plant performance to gain a conceptual design framework for future planting design of living walls.

Plant Materials and Used Living Wall Systems

The data of plant species used in the study were sampled *in situ* from two experiments conducted in southern Sweden (Malmö), in November 2014, using three types of living wall systems. The first of these was a capillary and hydroponic mat system with pockets (wall type 1 in **Figure 1**); each pocket filled with 1500 grams of substrate composed of 90% (vol.%) pumice and 10% compost. One or two plants were planted in each pocket at a density of 31 plants per m^2, the total planted area was 16.8 m^2. The second was a hydroponic Rockwool mineral wool based panel system (wall type 2 in **Figure 1**). Nine plants were planted in each panel at a density of 25 plants per m^2, the total planted area was 22.4 m^2. The third living wall type is a container system (wall type 3 in **Figure 1**). The containers contain a substrate composed of 60% pumice, 20% compost and 20% sand and host 5 plants each. The containers are arranged in columns with a distance of 20 cm between the rows and the total area of the wall covered 8 m^2 with a density of 25 plants per m^2. The three wall systems are classified as a continuous living wall system CLWS according to Manso and Castro-Gomes (2015) and were irrigated and fertilised to optimum levels. All the walls were facing south and installed on buildings in the south of Sweden between 2012 and 2014 at two locations. Wall type 1 and 2 was located on a masonry wall in a closed area and wall type 3 in a residential housing area at public disposal. The criteria used in plant selection for the experiment was based on tolerance to drought and solar radiation, their origin (native to Sweden), edible traits, conservation status and lifeform (e.g.

FIGURE 1 | Capillary mat pocket living wall system (Left: wall type 1, “pockets”); container living wall system (Centre: wall type 3, “containers”) and mineral wool living wall system (Right: wall type 2, “panel system”) growing in southern Sweden used to determine colour and visual plant performance of plants. (Photograph Ann-Mari Fransson).

	Wall type	Dominant colour			Accent colour			Visual performance	Biodiversity classification			Combined value
Botanical name	1=pocket 2=mineral wool 3=containers	L*	a*	b*	L*	a*	b*	mean +/- Standard deviation	Attractive for pollinators	Native plant species	Valuable plant species (POM)	1 = maximum value of visual performance and biodiversity classification.
Achillea millefolium	1	54.2	-22.9	39.7				3.2 ±0.7	1	1		0.9000
Acinos alpinus	2							1.9 ± 1.3				0.2375
Allium schoenoprasum	2	49.3	-18.7	19.4				1.7± 1.1	1			0.5125
Antennaria dioica	1	49.4	-18.9	19.4				1.7±1.7		1		0.4125
Arenaria montana	2	48.5	-16.8	16.9				2				0.2500
Armeria maritima	1	48.5	-16.8	16.9				3.2±1.4	1	1		0.9000
Aubretia x *cultorum* 'Blaumeise'	1	49.3	-18.7	19.4	71.2	-2.2	9.2	1.3±0.7	1			0.4625
Bergenia cordifolia 'Rotblum'	3	48.5	-16.8	16.9	43.5	34.2	17.8	2.5±0.5				0.3125
Bergenia cordifolia 'Möja'	3	48.5	-16.8	16.9	39.9	29.2	11.6	2.8			1	0.4500
Bergenia sp. LON 20090513 01:18	3	52.4	-26.9	29.8	29.8	26.3	4.9	3.8			1	0.5750
Calamintha nepeta	2	64.8	-25.6	44.0	60.8	-11.11	12.5	2.7±0.5	1			0.6375
Carex morrowii 'Ice Dance'	1	54.1	-22.7	39.7	92.5	-8.6	24.1	1.1±0.9				0.1375
Cerastium sp. ISB 20100727 01:5	3	54.6	-10.1	11.1				2.8			x	0.4500
Cerastium sp. KPN 20070619 02:9	3	77.8	-6.62	6.3				2.6			x	0.4250
Cerastium tomentosum	3	78	-5.6	6.2				3.1				0.3875
Chamaecyparis pisifera	2	71.4	-9.1	8.8	60.9	17.2	29.9	3.3±0.5				0.4125
Dianthus arenarius	2	40.4	-16.9	6.8				3.6±0.5	1	1		0.9500
Dianthus deltoides	1	53.1	-19.6	18.7				2.1±1.0	1	1		0.7625
Dianthus plumarius LON 20090514 01:4	3	48.5	-18.9	0.8				3.5	1		1	0.8375
Dianthus plumarius 'Marieberg'	3	56.5	-21	13.1				3.7	1		1	0.8625
Euonymus fortunei	2	58.2	-17.6	20.2	92.5	-3.3	11.1	3.6±0.5				0.4500
Euonymus fortunei 'Emerald 'n' Gold'	3	36.2	-18.9	1.2	70.8	-15.1	57.6	3.7				0.4625
Euphorbia polychroma	2							1.6±1.0				0.2000
Fragaria vesca	2							2.2±0.4		1		0.4750
Fragaria vesca var. semperflorens 'Rügen'	3	65.0	-25.6	43.9				2.8				0.3500
Geranium maculatum 'Espresso'	3	45.5	26.4	28.2				1.9	1			0.5375
Geranium sanguineum var. *striatum*	3	41.4	-15.0	13.2	41.2	26.1	15.9	2.8	1			0.6500
Geranium sanguineum ' Max Frei'	3	44.7	-15.4	15.9	55.6	14.7	35.7	2.6	1			0.6250
Pilosella aurantiaca	1	56.9	-29.2	36.1	54.9	1.3	20.0	2.5±1.3	1	1		0.8125
Hyssopus officinalis	3	70.9	-25.3	47.1				3±0	1			0.6750
Iberis sempervirens 'Appen-Etz'	1	37.1	-8.8	11.8				1.7±1.9				0.2125
Ilex crenata	2	70.9	-25.3	47.1				2.7±0.5				0.3375
Leontopodium nivale ssp. alpinum	2	49.3	-18.7	19.4	54.2	-0.9	12.0	0.1±0.3				0.0125
Luzula sylvatica	1	54.2	-22.9	39.7	54.9	1.3	20.0	2.7±0.5		1		0.5375
Microbiota decussata 'Sibirteppe'	3	52.4	-26.9	29.8				1.8				0.2250
Molinia caerulea	1	54.2	-22.9	39.7				2.2±1.6		1		0.4750
Molinia caerulea 'Variegata'	3	76.5	-21.4	38.1	47.2	-19.8	15.5	1.8				0.2250
Nepeta x *faassenii* 'Superba'	1	79.3	-5.2	11.7	71.2	-2.2	9.2	2.6±0.5	1			0.6250
Rubus x *stellarticus*	2							2.1±0.6	1	1		0.7625
Salvia nemerosa 'Marcus'	1	54.6	1.3	20.0				2.8±0.4	1			0.6500
Saxifraga x *urbium*	2	58.2	-17.6	20.2				1.5±0.9			1	0.2875
Sesleria heufleriana	1	49.3	-18.7	19.4	77.1	-1.4	11.7	2.2±0.6				0.2750
Sesleria nitidia	3	48.4	-16.8	17.17	59.1	-19.8	20.9	3.7				0.4625
Stachys byzantina 'Silver Carpet'	1							1.2±1.6	1			0.4500
Thymus serpyllum	2	70.1	-18.2	19.1				2.7±0.5	1	1		0.8375
Vaccinium vitis-idaea	2	50.0	-14.0	25.9				3.7±0.5		1		0.6625
Vinca minor	2	61.3	-26.0	36.6	54.6	-10.1	11.1	2.9±0.3				0.3625
Waldsteinia ternata	2	57.3	-14.5	31.0	40.6	21.0	15.0	2.3±0.7				0.2875

FIGURE 2 | Compilation of the species studied, and the species mean visual performance, biodiversity classification and the combined value. Plant species marked with grey hue indicates that no colour measurements have been performed. Number 1 in the Biodiversity classification columns indicated that the plant species: is attractive for pollinators; belongs to native plant species or is cultivated culturally valuable plant species (POM).

deciduous: geophyte: tufting: woody: sedge/rush: grass). The list of the 48 plant species planted in the three wall types is provided in **Figure 2** and visible in **Figure 1**.

Colour Determination and Analysis

Colour determinations were performed *in situ*, and the inherent foliage colours of the used plant species and varieties were documented from plants growing in the three living wall systems. The walls were fertilised to remove any possible interactions from nutrient deficiencies.

Out of the 48 plant species included in the three living wall systems, five plant species were excluded due to problems with visibility and accessibility. Colour analyses of the remaining 43 plant species were performed using colour charts from the RHS Colour Chart 5th edition. The colours were documented and systematically ordered in relation to each plant species intrinsic foliage colour and assessment of the plant species' dominant colours. Some species were registered having a dominant and an accent foliage colour (**Figure 2**). Only the foliage colours were documented, any flowers were excluded from the study. To avoid glitter and reflective surfaces caused by direct sunlight, the colour assessments were conducted between 10 a.m. and 2 p.m. in overcast weather.

The perceived 43 colours according to the RHS Colour Chart 5th system were converted into the CIE (L*a*b*) space. The CIE (L*a*b*) system is directly modelled on human colour perception (Tkalčič and Tasič, 2003) and represents differences in colour that correspond to human vision (Kendal et al., 2013). The CIE (L*a*b*) colour space communicate colour perception and show relationship between the two colour dimensions a* (green at negative a* values to magenta at positive a* values) and b* (blue at negative b* values to yellow at positive b* values). CIE (L*a*b*) has a linear measure of lightness (L*) where L* = 0 represents darkest black and 100 brightest white. The CIE (L*a*b*) colour values of the perceived 43 colours is visible in **Figure 2**.

Graphical Redesign of Living Wall Using Colour Theory

In order to be able to theoretically test Johannes Itten's concepts, light-dark- and complementary contrast, two graphical designs were prepared with species from the 2012 to 2014 living wall experiments in Malmö, Sweden. The number of selectable species and in turn available colour choices in the study, limited the number of alternative interpretation models. However, the two graphical redesigns were solely based on Johannes Itten's design principles and we based the colour contrast compositions on what Itten (2002) considered a "well-balanced and colourful arrangement." To ensure a balanced composition, each plant species CIE (L*a*b*) values have been used as a basis and inspiration for the development of the two graphic redesigns; light-dark, and complementary colour concepts. The two redesigns were designed without knowledge of each species biodiversity outcome. This approach enabled the designs to be as close to Johannes Itten's colour contrast concepts as possible, without being influenced by factors other than colour theory.

The two graphical redesigns used 25 plant species per m^2 of a living wall area. This was done to resemble the design structure of two of the studied living walls and the ratio of 25 plant species per m^2, which satisfies a goal of moderate to high species richness in a Northern European context (Crawley, 1986). The graphical

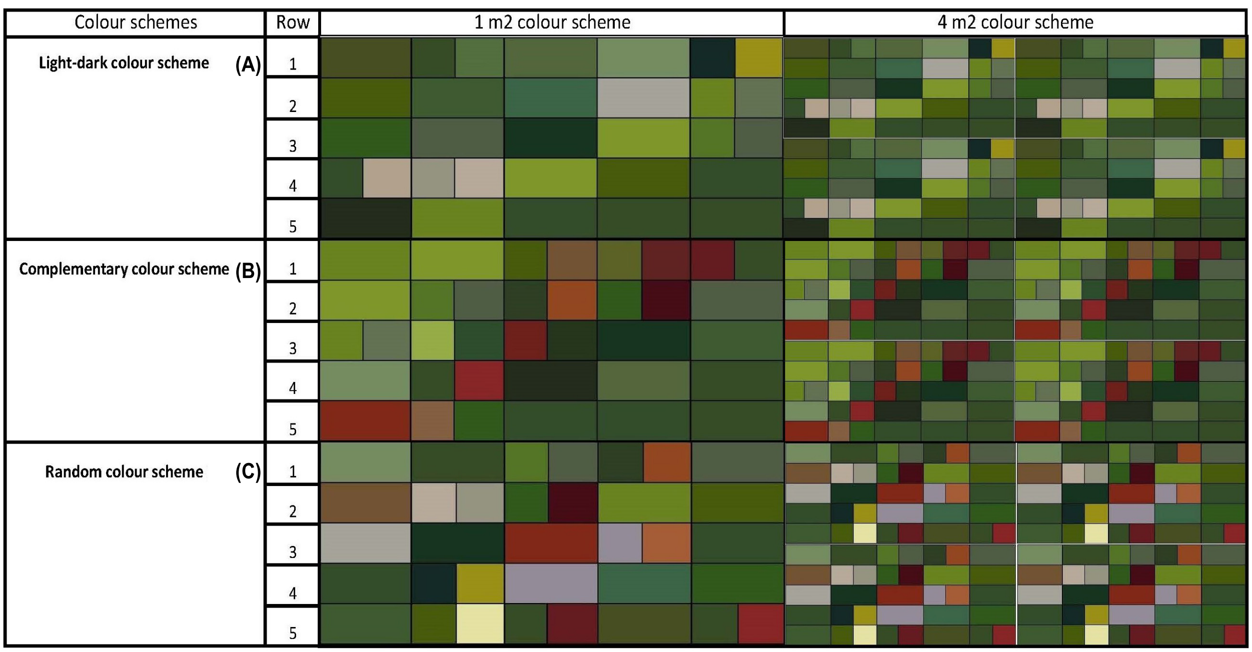

FIGURE 3 | Graphically rearranged plant designs of living walls using two colour schemes, light-dark (A) and complementary (B) and one random scheme (C) illustrated in 1 m^2 and 4 m^2 view.

Colour schemes	Used plant species in the colour schemes
Light-dark colour scheme (A)	Row 1: Vaccinium vitis-idaea; Sesleria nitida; Saxifraga x urbium; Thymus serpyllum; Euonymus fortunei 'Emerald 'n' Gold'. Row 2: Molinia caerulea; Dianthus deltoides; Dianthus plumarius 'Marieberg'; Cerastium sp.KPN 20070619 02:9; Calamintha nepeta. Row 3: Microbiota decussata 'Sibirteppe'; Cerastium sp. ISB 20090513 01:18; Dianthus arenarius; Ilex crenata; Vinca minor. Row 4: Sesleria heufleriana;; Nepeta X faassenii 'Superba'; Hyssopus officinalis; Achillea millefolium;Antennaria dioica. Row 5: Iberis sempervirens 'Appen-Etz'; Fragaria vesca var. semperflorens 'Rügen' ; Allium schoenoprasum; Arenaria montana; Armeria maritima.
Complementary colour scheme (B)	Row 1: Fragaria vesca var. semperflorens 'Rügen'; Hyssopus officinalis; Luzula sylvatica; Waldsteinia ternata; Bergenia cordifolia 'Möja'. Row 2: Ilex crenata; Vinca minor; Geranium sanguineum 'Max Frei'; Bergenia sp. LON 20090513 01:18; Cerastium sp. ISB 20100727 01:5. Row 3: Calamintha nepeta; Molinia caerulea 'Variegata'; Geranium sanguineum var. striatum; Dianthus arenarius; Dianthus deltoides. Row 4: Thymus serpyllum; Bergenia cordifolia 'Rotblum'; Iberis sempervirens 'Appen-Etz'; Saxifraga x urbium; Arenaria montana. Row 5: Geranium maculatum 'Espresso'; Pilosella aurantiaca; Allium schoenoprasum; Antennaria dioica; Armeria maritima.
Random colour scheme (C)	Row 1: Thymus serpyllum; Armeria maritima; Vinca minor; Geranium sanguineum 'Max Frei': Cerastium sp. ISB 20100727 01:5. Row 2: Salvia nemorosa 'Marcus'; Nepeta X faassenii 'Superba'; Bergenia sp. LON 20090513 01:18; Fragaria vesca var. semperflorens 'Rügen'; Achillea millefolium. Row 3: Cerastium sp. KPN 20070619 02:9; Dianthus arenarius; Geranium maculatum 'Espresso'; Chamaecyparis pisifera; Antennaria dioica. Row 4: Allium schoenoprasum; Euonymus fortunei 'Emerald 'n' Gold'; Cerastium tomentosum; Dianthus plumarius 'Marieberg'; Microbiota decussata 'Sibirteppe'. Row 5: Dianthus deltoides; Carex morrowii 'Ice Dance'; Bergenia cordifolia 'Möja'; Vaccinium vitis-idaea; Bergenia cordifolia 'Rotblum'.

FIGURE 4 | The species used to create a design based on light-dark design and complementary design and one random design for a theoretical living wall.

designs based on light-dark contrast and complementary contrast are presented via colour schemes (**Figure 3**), together with the plant species included in each design (**Figure 4**). To examine if either of these two graphical designs: light-dark colour scheme (A) and complementary colour scheme (B) were extreme, with respect to the variables studied, the colour schemes were compared with a randomly selected colour scheme (C) (see **Figure 4**).

Visual Plant Performance Assessment of the Plants Used in the Study

We calculated the visual plant performance using a 5 point scale where 0: 100% dead (no observed living tissue) 1: > 50% dead (just alive), 2: 25–50% dead (severely affected), 3: < 25% dead (slightly affected), and 4: 0% dead (thriving), and was based on Mårtensson et al. (2014). To gain a value of visual plant performance, we assessed between 5 and 8 specimens of the individual species to gain a mean value between 0 and 4. The value represents the plants ability to grow and thrive in combination with the species on these living walls. The visual plant performance data of each plant species in the study is shown in **Figure 2**. To estimate if the mean values of the plants in the light-dark (A) and complementary (B) graphical designs differed from the mean values in a random design we did a simulation of the mean values. The probability distribution of the mean visual plant performance of a random choice of 25 plant species was simulated using 10,000 simulations of plant species compositions out of the 43 plant species included in the study.

Biodiversity Classification

Plant Selection for Biodiversity Classification

Three plant selection criteria were used to define a biodiversity classification in this study—attractiveness for pollinators, native plant species and valuable cultivars. Pollinators within this study are defined as pollinating insects, firstly bees, bumblebees and syrphids but also butterflies. Plants attractive for pollinators data was gained from three main sources: Garbuzov and Ratnieks (2014), Rollings and Goulson (2019), and Goulson (2021). Two further sources, Teräs (1976) and Fussell and Corbet (1992), were used to explore flower visits by bumblebees. Observations from Haaland (2015) were also included. Plant attractiveness for pollinators is not well researched at a species level and not available in the literature (Garbuzov and Ratnieks, 2014). Therefore, in this study we classified plant species regarding their attractiveness for pollinators at a genus level. Further, in this study plant species were included based on evidence to support their potential attraction for pollinators, rather than more quantitative evidence in the literature (i.e., how many pollinators visited a certain plant species in a certain time). This means also that plant species could be classified as being attractive even if the numbers of visiting insects was low, but insect visits were recorded. In this way, the following plant genera were classified as potentially being attractive for pollinators: *Achillea*, *Allium*, *Armeria*, *Aubretia, Calamintha, Dianthus, Geranium, Hieracium, Hyssopus, Nepeta, Rubus, Salvia, Stachys,* and *Thymus* (**Figure 2**).

A total of 12 species in this study are native to Sweden according to Mossberg (2010) (**Figure 2**). Seven plant cultivars with high conservation value were also included based on their "cultural heritage" value. This was based on data collected from the National genebank for the preservation of older varieties of ornamental species, otherwise known as "The program for diversity of cultivated species" or the acronym POM. These plant cultivars have cultural-historical values, as well as the diversity values, attributes that are included in Convention on biological diversity (2021).

Biodiversity Classification and Data Analysis

Each individual species is classified as 1 or 0 for three variables, the "attractiveness for pollinators," whether it is "native plant species" and whether it is "cultivated culturally valuable plant species." To combine these components, a biodiversity value was calculated using the formula

$$\text{“biodiversity value”} = 0.6^{*}\ \text{“attractive for pollinators”} + 0.4^{*}\ \text{“native”} + 0.2^{*}\ \text{“valuable cultivar.”}$$

All components in this equation are either 0 or 1, and since none of the plant species included in the study are classified as "native plant species" and "culturally valuable plant species," the biodiversity value is between 0 and 1. The values were weighted based on that it is twice as valuable if a plant is native than cultivated and three times as valuable if it is attractive for pollinators than cultivated giving a relationship between attractive for pollinators: native: cultivated culturally valuable of 3:2:1, where close to 1 expresses a high value for biodiversity value (see **Figure 2**).

To see if the biodiversity in terms of attractiveness for pollinators, number of native plant species and number of cultivated culturally valuable plant species in light-dark (A) and complementary (B) is in some way extreme, we compared the graphical designs to a random choice of 25 species out of the 43 species available. For this purpose, we use the fact that the number of elements with a given property in a random choice from a finite population is distributed according to a hypergeometric distribution. In this case, the population size is $N = 43$, the number of chosen elements is $n = 25$, and K is the number of elements in the populations with the given property. In this study, the parameters and the observed values in the hypergeometric distribution for the different situations are in **Table 1**. In the analyses, we compared the values obtained in the graphical designs with the probability distribution of the hypergeometric distribution, and were able to see if the values obtained in the graphical designs were extreme.

Biodiversity Classification and Visual Plant Performance Values

To have a same weighted value for the visual plant performance and biodiversity values, the value for visual plant performance was standardised to between 0 and 1 by dividing the values with the maximum value for the visual plant performance (4).

The "standardised visual plant performance" is now on the same scale as the biodiversity value and the combined value can be defined as

$$\text{“combined value”} = 0.5^{*}\ \text{“standardised visual plant performance”} + 0.5^{*}\ \text{“biodiversity value”},$$

Where 0.5 in the formula expresses the fact that the weight for the two components is the same.

The combined value is a value between 0 and 1 where close to 1 expresses a high value for the visual plant performance and the biodiversity value (see **Figure 2**).

TABLE 1 | Parameters used in the analyses with the hypergeometric distribution.

	N	n	K	x, light-dark (A)	x, complementary (B)
Attractive	43	25	17	10	11
Native	43	25	10	8	7
Culturally valuable	43	25	7	4	4

RESULTS

Outcome of Colour Contrast Interpretation Models, Plant Biodiversity Classification, and Visual Plant Performance

The sections below display the design structure using the light-dark (A) and complementary (B) interpretation models as well as the outcome of visual plant performance in each design. The conceptual models are tested against a biodiversity classification and a combined value of biodiversity and visual plant performance.

Design Outcome and Structure of Light-Dark and Complementary Interpretation Models

A total of 39 plant species, from 43, were used in the preparation of light-dark (A) and complementary (B) graphical redesigns with a species richness of 25 plants/m^2. In the design of the colour schemes A and B, Johannes Itten's contrast of extension has been used as a base, where contrast of extension is correctly expressed, a contrast of proportion. Consideration and attention to the coloured areas in design A and B has been an important issue in the actual choice of plant species from **Figure 2**. To achieve a brightness effect in colour scheme A and B (**Figure 3**), a clear distinction and graduation of light and dark values are incorporated, and repetition and grouping are used to increase the brightness effect.

Light-dark colour scheme (A) is composed using structures of diagonal direction with contrasting and groupings of light and dark green/grey values in coloured boxes (**Figure 3**). A clear diagonal orientation in the composition creates perspective illusions, movement and depth into the arrangement (Itten, 2002). The rectangular boxes to the left and the right of the diagonals consist of medium green values, where the medium values in the composition increases the light-dark contrast effect. The complementary colour scheme (B) is arranged via two visually clear areas of light and dark green values in coloured boxes. The dark values are placed in the lower part of the composition and the light green values in the upper part. The dark values placed in the lower part of the composition stabilises the colour elements in the arrangement. In contrast, yellowish lighter green values are perceived as weightless colour elements and are placed in the upper part of the colour arrangement. The coloured rectangular boxes in colour scheme C are randomly placed with 25 randomly chosen plant species. Green, red and grey hues with a mix of light, medium and dark values are visible in a non-designed arrangement.

The order of the species in each colour scheme follows a structure with 5 × 5 rectangular boxes, in horizontal and vertical directions respectively. The species in each colour scheme (A, B, C) are arranged from left to right in the composition, with row 1 at the top on the left hand and row 5 starting at the bottom on the left hand of the colour schemes. In cases where a selected plant species were registered in the colour assessment having both a dominant and an accent colour, both the dominant and the accent colours are represented in the coloured rectangular boxes respectively. To show the outcome of the light-dark- and complementary compositions and the randomly selected model in a broader contextual situation, each 1 m^2 colour scheme is visible via a 4 m^2 colour scheme (see **Figure 3**).

Outcome of Visual Plant Performance in Light-Dark- and Complementary Interpretation Models

The two graphical design interpretation models visual plant performance were compared with 10 000 randomly chosen simulations (**Figure 5**), to examine if any of the colour schemes, A and B were extreme in relation to the factor visual plant performance. The mean visual plant performance for the 25 plant species in the graphical design light-dark (A) is 2.66 and for complementary (B) 2.51. The distribution of the mean value of a wall with 25 randomly chosen species out of the 43 was simulated using 10 000 simulations and the mean is 2.58 (**Figure 5**). The simulation show that the visual plant performance for light-dark (A) and complementary (B) are far from extreme and not significantly different from the random choices.

Outcome of Plant Biodiversity Classification of Tested Plants in the Study

For each of the 39 plant species we have created a biodiversity classification. Of the plant species, 17 were classified as being potentially attractive for pollinators (**Figure 2**). The light-dark (A) design included 10 plant species attractive for pollinators, while the complementary design (B) had 11 plant species (**Figure 6**, left). Compared with the hypergeometric distribution, neither of them were significantly different from a random choice of species. Of the plant species used in the colour schemes in **Figure 2**, 10 were classified as being native plant species. The light-dark design (A) included 8 native plant species and the complementary design (B) included 7 native plant species. Compared with the hypergeometric distribution, this is not significantly different from a random design (**Figure 6**, central). Of the plant species in **Figure 2**, **7** plant species were categorised as being valuable cultivars. The graphical designs light-dark (A) and complementary (B) both included 4 plant species valuable cultivars by POM. Compared with the hypergeometric distribution, this is what you could expect from a random choice, illustrated in **Figure 6** (right).

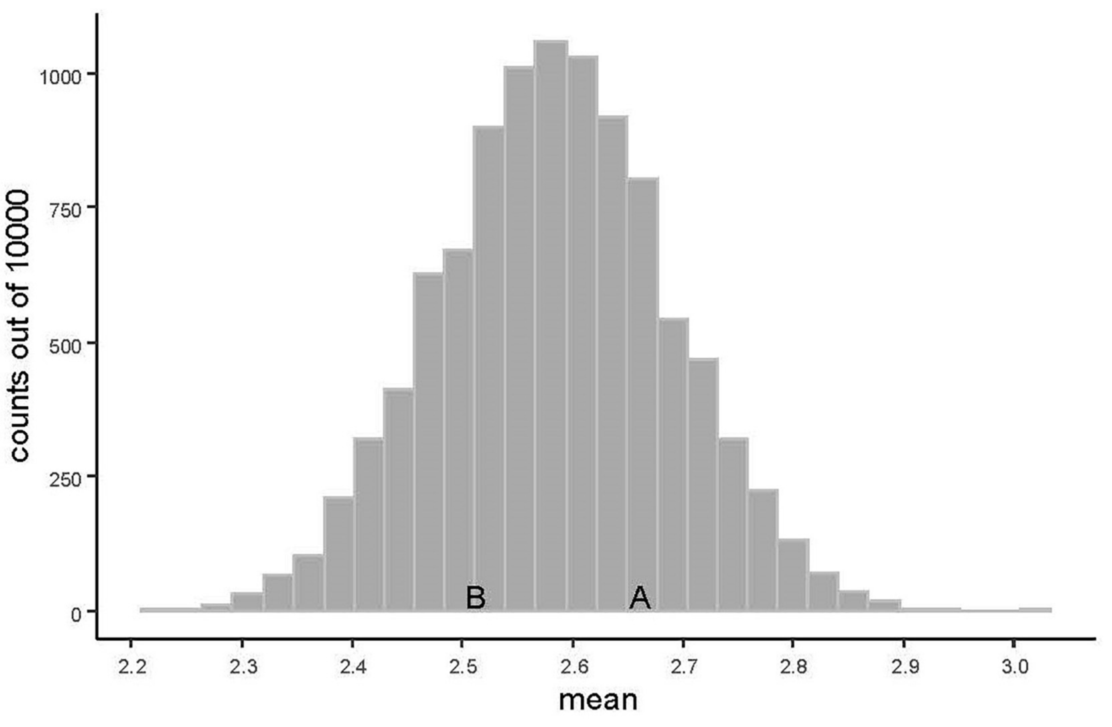

FIGURE 5 | The distribution is based on 10,000 simulations of the mean value of 25 randomly chosen numbers from the values of visual plant performance. The letters correspond to the mean values for the graphical designs: light-dark (A) and complementary (B).

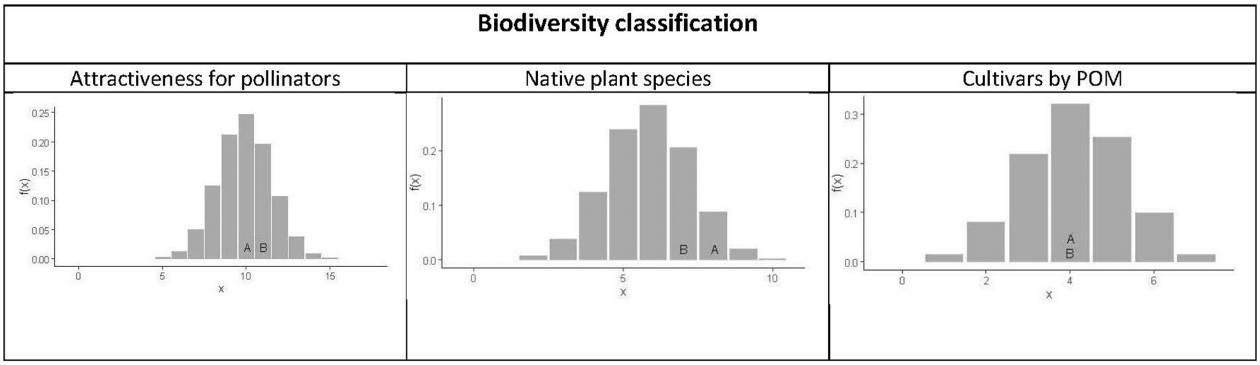

FIGURE 6 | The probability function for the number of attractive plant species for pollinators (left), native plant species (central) and cultivars with conservation value of a random selection of the 43 species possible to use in the design. The letters A and B in the figures correspond to the number of species in the designs based on light-dark (A) and complementary (B) colour theory.

In conclusion, none of the graphical designs light-dark (A) and complementary (B) had an extreme number of plants attractive for pollinators, were native or culturally valuable, when compared to a random choice of plant species.

Outcome of Plant Species With a High Combined Value of Biodiversity and Visual Plant Performance

The top three species that had the highest combined biodiversity value and value for visual performance were: *Dianthus arenarius*, *Achillea millefolium,* and *Armeria maritima*. All three species are included in the graphical design light-dark (A) and two species in the complementary design (B).The combined values of the biodiversity value and visual plant performance is between 0 and 1, where 1 expresses the best possible combination of the visual plant performance and biodiversity value (attractiveness for pollinators, native plant species and valuable cultivars). The genus Dianthus with the species: *Dianthus arenarius*; *Dianthus plumarius* "Marieberg" and *Dianthus deltoides* had the highest sum between 0.95 and 0.76 (**Figure 2**). The calculations show that the values for the genus Geranium also were generally high, with a sum between 0.65 and 0.54.

DISCUSSION

Colour Concepts as a Method for Achieving Aesthetics and Biodiversity Outcomes on Living Walls

The results from this study show that it is possible to design species rich living walls by implementing colour contrast concepts without compromising biodiversity. Both these outcomes, high colour contrast and biodiversity, were preferred by visitors to a botanical garden when they were asked to design meadows (Lindemann-Matthies and Bose, 2007). Flowers in particular have been shown to increase the attractive preference and the restorative effect of digitally designed urban green space (Wang et al., 2019), while flower colour diversity has been shown to influence both people's aesthetic responses and pollinator interactions in direct sown meadow plantings (Hoyle et al., 2018). Colourful designs with a high diversity were created and selected as the most attractive when people could choose. Including both colour and diversity in designs of urban green space is of high importance to aesthetic preference and restorative potential in landscape design.

While neither of the graphical designs (i.e., light-dark and complementary) in this study had an extreme plant biodiversity classification compared to a random selection of plant species, the method provides consideration of this as a tool to improving plant selection in living walls. Even small patches of urban vegetation can support high species richness and diversity in cities, even for uncommon species (Vega and Küffer, 2021). Providing greater connections between these patches is crucial to achieve more urban habitat for biodiversity (Vega and Küffer, 2021) and living walls could be a way of providing these connections, especially as they have the potential to become "vertical corridors for wildlife" (Mayrand and Clergeau, 2018).These results were based on the colour determinations of plants growing on living walls facing south in the northern hemisphere (north-facing in the southern hemisphere). Plants colour will be related to the light intensity due to increased cuticle thickness and chlorophyll content on locations with high solar radiation as well as nutrient limitations. These specific designs are therefore restricted to living walls facing similar directions. Similarly, the visual plant performance will be directly related to the type of living wall system in use. However, the method used here—to design living walls guided by colour theory and using plants that are attractive for pollinators, have a level of "nativeness" and/or are valuable cultivars is more general and may be applied to any living wall system.

An intensification of cultural ecosystem services in green infrastructure installations could also lead to an increase in identity and senses of place (Eliasson et al., 2018). And while more aesthetically pleasing green infrastructure installations could foster more social cohesion and community, they are also not without some significant challenges. The use of colour concepts as a method in the design process can be particularly challenging for practitioners. We currently do not fully know the best way to design an outdoor composition focusing on the visual effects of colour contrasts. More research is needed on the perceived effects of a colour contrast composition giving contrasts to the perceived three-dimensional situation. In addition, studies of the overall colour performance of green installations in different contextual situations is required. It is also essential that practitioners involved in the planning- and design process have knowledge of the impact and potential effects of colour contrasts on human health and wellbeing.

Human Experiences Relative to Theoretical Colour Concepts on Living Walls

Previous studies show that the method for aesthetic outcome using colour contrasts as a design principle has the ability to affect human values, such as aesthetic enjoyment and appreciation. It has furthermore the capacity to contribute to increase visual quality and experiences of beauty, which in turn can be beneficial and positively associated with human health. This means that the conceptual design proposals developed in this study have a strong potential to affect humans positively, be highly valued by people and benefit human wellbeing. The design concepts use strategies already evaluated and proven to achieve harmonious arrangements and have the ability to influence visual qualities and viewer's aesthetic satisfaction (e.g. Westland et al., 2007; Yilmaz et al., 2018). The conscious use of colour contrasts in plant compositions also has the potential to contribute to wellbeing outcomes, and increase experiences of pleasantness, beauty and harmony (Eroğlu et al., 2012; Oleksiichenko et al., 2018; Huang and Lin, 2019; Thorpert, 2019; Zhuang et al., 2021). In the light of the above reasoning, it is motivating to notice that green infrastructure installations, designed in a careful and conscious way, have the ability to be supportive for humans and contribute to the viewed quality in our daily life. However, it is important to point out that this study has been based on a two-dimensional way of thinking, which may suit living wall compositions, but will need complementary approaches in three dimensional planting designs.

Methodological Considerations

This study should be seen in the light of being explorative where the method allows the process to be creative in order to gain insightful information on the studied subject. We have investigated the possibilities to find a way to balance biodiversity and aesthetic outcomes from a theoretical point of view, and to bridge the knowledge gap in the design and planning process. The results give some indications that the method used could be applicable and a tool in the design of living walls and similar green infrastructure installations. However, the method has some limitations and should be seen as an attempt to create new ways to meet the requirements of designing with biodiverse vegetation that has multiple outcomes and functions as a goal.

In this study, we measured the colours from plants living in three types of living wall systems, where the systems were located facing south, respectively. How central each living wall system is for the intensification of vegetation colours has not been included in this study, but is something that can be prioritised in future studies. Also studies of the location of living walls in different latitude directions and its impact on plant colours is of importance to be able to create well-balanced aesthetic decisions in the future. A limitation in this study has been that only one time of determination has been included. A deeper understanding about seasonal changes and lightning conditions impact on plant colour performance could help ensure a balanced choice of plant mixtures and contribute to a holistic approach in relation to human health and wellbeing.

The methodology used to classify plant species with respect to their attractiveness for pollinators has some challenges. As noted, there is little empirical evidence of plant attraction for pollinators at a species level, in accordance to this we in this study included species based on their genus from the available literature. However, we are aware that attractiveness of plant species for pollinators can vary considerably within a genus, such as Geranium in Rollings and Goulson (2019). It has also to be noted that this way of classification leads to the fact that certain plant species as for example *Dianthus deltoides* were here classified as being attractive for pollinators, while the species investigated in the empirical study was a different one, here Dianthus barbatus (Rollings and Goulson, 2019). Nevertheless, it is assumed that the methodology chosen is justifiable in the context of this study. A further differentiation of plant species regarding their attractiveness would have been desirable but was dismissed because of the lack of data for all plant species.

This study has clear demarcation in the selection of choosable plant species and hence a limitation in the possibilities to develop design solutions. It is likely that additional plant species available in the design process could lead to an increased biodiversity value. It is also likely that an increase in the number of choosable plant species might provide major opportunities to create colourful design solutions and sustainable colour contrast effects. This in turn would lead to higher visual attractiveness and be more valued by people and increase their feeling of wellbeing.

The methodological approach used in this study, takes into consideration that both social and ecological plant traits is a useful way to make the plant choices in urban plantings more based on scientifically knowledge. However, the method needs to be validated, especially the weighting among the different values needs more research and inclusion of more parameters is recommended.

CONCLUSION AND FUTURE STUDIES

Each plant species selected for use in the two colour schemes were consistent with Itten's concepts of "aesthetic attractiveness." Based on knowledge from this study, we suggest that the use of proven theoretical colour models in the planning and design of species rich living walls and comparable green infrastructure installations have the possibility to give visually attractive urban installations, which in turn can be beneficial and positively associated with human health. The study furthermore shows that it is conceivable to design living walls containing multiple outcomes and functions. The study shows that the biodiversity value in the graphical colour designs used in the study is not significantly different from a random choice. Meaning that it is possible to add benefits of colour theory to a design of living walls without compromising biodiversity outcomes.

In conclusion, living walls have the potential to be more thoroughly designed and have the possibility to be more highly aesthetically pleasing and highly valued by people and increase their feeling of wellbeing, and at the same time deliver a high level of biodiversity values and species per m^2.

The study also shows that green infrastructure installations designed in a well-balanced way have the potential to contribute toward both cultural- and regulatory ecosystem services. However, more work is needed to get a comprehensive representation of the visual feature, colour contrast and its potential as a design parameter in green infrastructure installations.

A deeper understanding about plant colours in various environmental settings during different light- and seasonal conditions could help ensure a balanced choice of colourful plants attractive for both humans and pollinators. This could also help identify optimal placement of coloured plants in relation to the surrounding contextual situation as well as in relation to colour theory. A follow-up from this study could be human subjective studies to assess preferences for different colour contrast models, also comparing studies of perceived plant colour, plants attractive for pollinators, indigenous and culturally valuable plant species and human experiences might enable an understanding of differences and similarities between perceived colour contrasts and biodiversity values. Longitudinal studies could also give indications about perceived colour contrast and biodiversity values in relation to human health and wellbeing and contribute to a more comprehensive understanding of the integration between biodiversity and colour theory in the design of living walls.

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

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REVIEW
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Planning, Designing, and Managing Green Roofs and Green Walls for Public Health – An Ecosystem Services Approach

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Installing green roofs and green walls in urban areas is suggested to supply multiple ecosystem services of benefit to human health and well-being. In a three-step literature review, we examined current knowledge on the link between public health and green roofs and green walls. A systematic search identified 69 scientific articles on green roofs/walls with a public health discourse. These articles were categorized according to type of health path covered (reduction of temperature, air pollution, noise or environmental appraisal) and coverage of issues of relevance for strategies on planning, design/construction, and maintenance of green roofs and green walls. Articles identified through the structured search were complemented with reviews (with no explicit public health rationale) covering reduction of noise, temperature, or air pollution and environmental appraisal. Other relevant studies were identified through snowballing. Several of the articles provided guidelines for optimizing the effect of green roofs/walls in supporting ecosystem services and maximizing well-being benefits to support health pathways identified. These included specifications about planning issues, with recommended spatial allocation (locations where people live, sun-exposed for maximum ambient temperature reduction) and with physical access needed for environmental appraisal. Recommendations regarding design parameters covered substrate depth (deeper generally being better), plant choices (more diverse roofs providing more services), and maintenance issues (moist substrate positively correlated with heat reduction).

Keywords: temperature regulation, air pollution regulation, noise regulation, environmental appraisal, public health and well-being, green infrastructure, nature-based solution, living walls

INTRODUCTION

Green roofs and green walls have been promoted as features to improve the amount of urban green space, mainly within the dense city, motivated by their contribution to improving the urban environment (Norton et al., 2015). Lately, urban vegetation has received increased attention through the European Union's launch of nature-based solutions (NbS), where different forms of green infrastructure (such as green walls and green roofs) are seen as a measure for dealing with environmental and social issues within urban environments (e.g., Raymond et al., 2017).

Green roofs are a constructed system comprising vegetation growing on horizontal panels that are incorporated into existing built infrastructure and generally intended for environmental benefits, such as stormwater mitigation (Bengtsson, 2010). A green wall is part of what is called vertical greening or green facades (Köhler, 2008), defined as a building envelope based on living plants, and the term refers to all forms of vegetated vertical surfaces (Manso and Castro-Gomes, 2015). Green walls divided into green facades and living walls (Radić et al., 2019). In this review, we mainly focused on the living walls type, which can also be referred to as modular green walls (Köhler, 2008). Many previous review studies have identified a link between public health/well-being and natural environments, with e.g., a review by van den Bosch and Ode Sang (2017) summarizing the evidence on public health benefits from NbS. Most published studies assess health and well-being using a socio-ecological approach, where both environmental and social determinants are seen as contributing to health and well-being.

Section "Regulating ecosystem services – reduction of heat, pollution, and noise" summarizes the current understanding of actual health pathways for NbS, based on the framework presented by van den Bosch and Ode Sang (2017; see also **Figure 1**).

Regulating Ecosystem Services – Reduction of Heat, Pollution, and Noise

Increased *urban heat* has been shown to be a strong predictor of a range of diseases [e.g., mental health, cardiovascular disease (CVD)] and all-cause mortality (Berry et al., 2010; Basagaña et al., 2011; Benmarhnia et al., 2015). Recent reviews have also indicated an association between heat load and decreased birth weight, although the evidence is inconsistent (Beltran et al., 2013; Poursafa et al., 2015). Urban heat affects mortality and morbidity through a combination of exposure to greater heat and vulnerability to extreme heat events, with heat sensitivity varying within populations and globally (Campbell et al., 2018). Another important factor influencing the effect of urban heat is behavioral exposure, i.e., the number of people using public open space (Norton et al., 2015).

Urban green space is reported to have potential in mitigating the urban heat island (UHI) effect, although the degree of mitigation is dependent on spatial location, vegetation type, and urban morphology (e.g., Bowler et al., 2010b; Norton et al., 2015). For instance, the mitigating effect of vegetation on UHI has been shown to be greater in densely built-up areas than in more sparse developments, with variations due to prevailing wind direction and time of day (Žuvela-Aloise et al., 2016). There is also seasonality in the effect of urban vegetation, with stronger effects in summer than early spring (Zhou et al., 2014). In addition to these broad differences in cooling, there is also variation linked to the level of soil sealing and amount of vegetation, which could explain micro-climate effects (Lehmann et al., 2014).

Air pollution adversely affects human health, resulting in an increase in respiratory illnesses such as asthma, a higher incidence of CVD (cardiovascular diseases), and impaired neural development and cognitive capacities (e.g., PopeIII, Burnett et al., 2002; Fann et al., 2012; EEA, 2016; WHO, 2016). The European Union has introduced legislation to improve human health by restricting different pollutants (e.g., SO_2, SO_x, NO_2, NO_x, NH_3, PM, CO, O_3, heavy metals, BaP, PAH and VOCs) (EEA, 2016). The main source of air pollution within cities is motorized traffic, but waste incineration, heating (domestic and thermal power generation), agriculture, and industry can have strong local effects (EEA, 2016).

Vegetation has the potential to mitigate air pollution. The density, height, thickness, and coverage of the vegetation, as determined by relative tree cover, tree size, and density, are the main characteristics determining the effect (Escobedo and Nowak, 2009; Tiwary et al., 2009; Tsiros et al., 2009; Dzierżanowski et al., 2011; Tallis et al., 2011; Nowak et al., 2013; Baldauf, 2017). However, there are differences between species (Benjamin et al., 1996). Uptake of gaseous and particulate pollutants is related to the morphology and physiology of plants (e.g., Beckett et al., 2000; Klingberg et al., 2017). For instance, traits such as compactness, plant hair density, plant leaf density, leaf wax, leaf surface area have been shown to influence particulate matter mitigation (Dzierżanowski et al., 2011; Hwang et al., 2011; Sæbø et al., 2012; Speak et al., 2012).

A WHO report published in 2011 concluded that noise has a negative impact on human health and that there is sufficient evidence of a relationship specifically with annoyance, sleep disturbance, CVD, cognitive impairment, and tinnitus (WHO, 2011). Noise exposure depends on the space-time behavior of individuals and differs between the residential, commuting, and work environment, with all three contributing to overall exposure and subsequent health implications (Díaz and Pedrero, 2006). Several studies have highlighted the positive influence of noise reduction through quiet urban areas, such as green spaces, and their possibility to act as a mitigating measure (Öhrström, 1997; Öhrström et al., 2006), while the vegetation itself also provides mitigation (Ow and Ghosh, 2017). Noise exposure in the urban context is largely due to motorized transport, but also construction and industry, community sources, and social and leisure sources, with the contribution of these sources varying spatially and over time (Moszynski, 2011).

Natural vegetation can reduce noise and contribute to a better soundscape (Viollon et al., 2002). Studies on traffic noise have shown that if the natural vegetation is sufficiently high, wide, and dense, it can decrease recorded noise levels (e.g., Viollon et al., 2002; Fang and Ling, 2003; Ow and Ghosh, 2017). For instance, it has been suggested that 30 m width vegetation can reduce noise levels by up to 8 dB (Huddart, 1990) or that 3 m width of dense vegetation can result in an attenuation of 5 dB (Kragh, 1981).

Cultural Ecosystem Services – Socio-Behavioral Pathways

A range of health benefits are closely associated with socio-behavioral pathways and supply of cultural ecosystem services, requiring individual interaction with the environment. Studies on specific natural elements have provided knowledge about visual qualities and their impact on people's emotions and preferences associated with *environmental appraisal*. Recent reviews provide

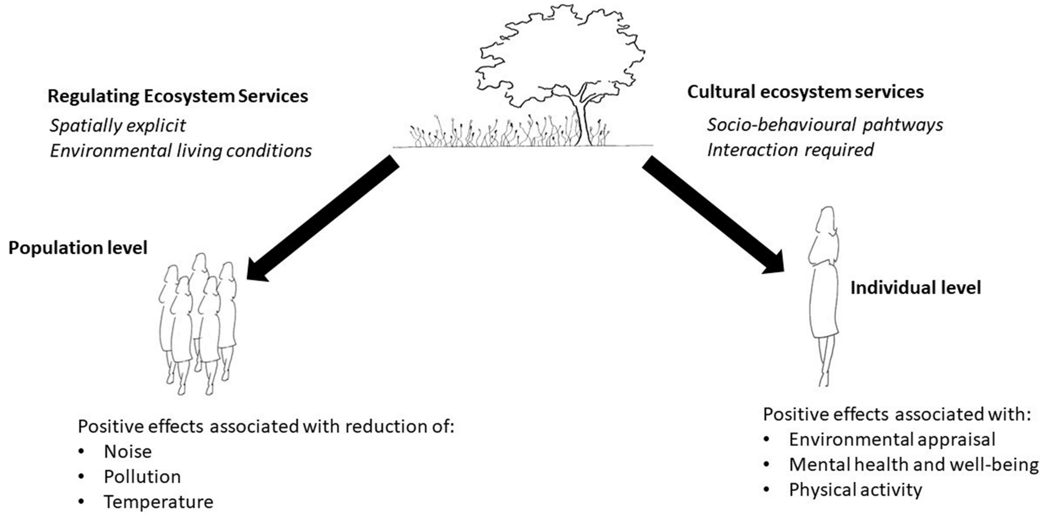

FIGURE 1 | Positive health benefits from urban green spaces, which are mainly associated with regulating and cultural ecosystem services. Regulating ecosystem services are spatially explicit and work on population level through improving environmental living conditions, while cultural ecosystem services are dependent on socio-behavioral pathways requiring interactions on individual level.

strong evidence of a positive impact of natural environments on behavioral affects and reduced levels of anger and sadness (Bowler et al., 2010a). Positive behavioral affect is in turn strongly related to CVD and all-cause mortality (Shirom et al., 2010; Lamers et al., 2012; Mroczek et al., 2013).

Visual properties such as openness (Hanyu, 2000; Jorgensen et al., 2002; Motoyama and Hanyu, 2014) and coherence (Motoyama and Hanyu, 2014) in a natural setting may have the potential to generate affective appraisals such as safety (Hanyu, 2000; Jorgensen et al., 2002; Motoyama and Hanyu, 2014) and relaxing and pleasant (Hanyu, 2000; Motoyama and Hanyu, 2014). Experience of naturalness has also been found to act as a mediator to experience of well-being (Knez et al., 2018). Ground cover features such as lawns are positively correlated with "rest and restitution" (Peschardt, 2014). A study by Nordh and Østby (2013) found that a "lot of grass and flowers/plants" were liked qualities, whereas absence of vegetation and presence of hard surfaces (pavement and buildings) decreased the perceived restorative qualities. Perceived visual quality has been found to increase with a high percentage of vegetation and presence of color contrasts (Arriaza et al., 2004), with a positive correlation also between visual preferences and medium to high plant species diversity and plant color composition (e.g., complementary color) (Polat and Akay, 2015). Recent reviews of the link between natural environments and health indicate that there is strong evidence of a positive relationship between natural environments and *mental health and well-being* (e.g., van den Berg et al., 2015). Several studies have reported restorative qualities of urban green space for coping with mental health problems (Barton and Pretty, 2010) and as a pre-emptive measurement for decreasing stress and providing "instorative" effects (e.g., Barton and Pretty, 2010; Nordh et al., 2011). In environmental psychology, restorative environments refer to environments that can trigger a psychological and/or physiological recovery process (Joye and van den Berg, 2013). Studies have shown that exposure to nature has restorative effects (Hartig, 2007), with healthy, unstressed participants reporting improvements in their subjective energy levels (Ryan et al., 2010), mood states, and ability to reflect (Mayer et al., 2009). While most studies to date have focused on the visual qualities of natural environments as a stress pre-emptive measurement, recent studies have highlighted the importance of natural sounds (such as birdsong) (e.g., Alvarsson et al., 2010; Annerstedt et al., 2013).

Aim and Objective

The studies cited above were based on green spaces more generally, with few exploring specific types of green elements. Within the dense urban fabric, retrofitting of green roofs and green walls is often the only way of increasing the amount of green.

Our aim in this study was to explore whether, and to what extent, research findings on relationships between green spaces generally and public health apply to constructed green infrastructure such as green walls and green roofs. Our starting hypothesis was that when adequately planned, designed, and maintained, green roofs and green walls have the potential to supply the same human health and well-being benefits as green space in general. Therefore, we specifically examined evidence-based guidelines on green roofs and green walls issued within urban governance structures concerned with planning, design, and management issues.

METHOD

In order to understand the discourses on public health benefits of green roofs and green walls, we conducted a three-stage literature review that combined a main systematic literature search with an additional systematic literature search (to identify additional relevant reviews) and a non-systematic literature search (snowballing technique; Almenar et al., 2021).

The systematic literature search was conducted in August 2021, using Scopus. We limited the search to peer-reviewed articles in English and used the following search term: (“**green roof***” OR “**green wall***”) AND (**health** OR **wellbeing** OR **well-being** OR **aesthetic**). After careful consideration, we decided not to include the term ‘barrier’, which is often used in relation to green walls in street-level settings, due to the multiple meanings of the term in relation to green space (e.g., physical barrier hindering access). Thus the search is likely to have missed some articles focusing on street-level features.

Since the aim of the review was to compile evidence of health and well-being benefits from green roofs and green walls, during screening of titles and abstracts for the initial search hits, we excluded articles with no connection to health-related performance and well-being. We categorized each remaining article according to type of study (review, empirical, simulation), the public health benefits discourse (noise, pollution, thermal, mental health/well-being, environmental appraisal), and aspect of green walls and green roofs (presence/location, design with regard to form, material, vegetation, performance as regards status and health of vegetation). In addition information on spatial context and measurements carried out was included when available.

Our second structured literature search was carried out in Scopus on 15 September 2021 and, in addition to the search terms Green wall* OR Green roof*, used the terms presented in **Table 1**. The aim of this second search was to identify reviews covering the health-supporting ecosystem services pathways identified by van den Bosch and Ode Sang (2017). **Table 1** shows number of published articles identified for each search string, the number of reviews within the total, and reviews relevant to our topic and hence included in our analysis.

The hits from the first systematic literature search were classified into the following three groups, based on type of study and where in the urban governance the results can be implemented:

(1) Planning: Location and placement of green roofs and green walls.
(2) Design: Design of components of green roofs and green walls.
(3) Management: Maintenance of green roofs and green walls.

RESULTS

Discourses on Green Roofs and Green Walls for a Health-Promoting City

The first systematic search in Scopus resulted in 207 articles, with 69 articles identified as relevant and included in our analysis (**Table 2**).

In the two structured literature searches we identified 20 reviews, of which several provided broad overviews of the benefits deriving from green roofs (Berardi et al., 2014; Francis and Jensen, 2017; Liu et al., 2021) and green walls (Medl et al., 2017; Ghazalli et al., 2019). For temperature and air quality regulation, there were numerous reviews dealing solely with these aspects, with 11 reviews focusing on temperature regulation (Bowler et al., 2010b; Hunter et al., 2014; Norton et al., 2015; Santamouris, 2015; Charoenkit and Yiemwattana, 2016; Santamouris et al., 2016; Francis and Jensen, 2017; Pisello et al., 2018; Cascone et al., 2019; Jamei et al., 2021; Liu et al., 2021) and six reviews covering different aspects of pollution (Rowe, 2011; Li and Babcock, 2014; Francis and Jensen, 2017; Corada et al., 2020; Liu et al., 2021; Ysebaert et al., 2021). For noise, we identified one review (Yang and Jeon, 2020). In relation to environmental appraisal and mental health, we found two reviews covering aspects of this (Fernandez-Canero and Gonzalez-Redondo, 2010; Williams et al., 2019).

The first structured literature search, which included a specific public health discourse (**Table 2**), mainly identified papers focusing on temperature and air quality regulation (25 and 23 hits, respectively). The second literature search (which did not include search terms related to public health) identified 705 and 420 articles dealing with the regulating services temperature and air quality regulation, respectively. For noise and environmental appraisal/mental health, fewer studies were identified in both structured literature searches. For noise, we identified eight articles in the first search and 91 in the second search, while for environmental appraisal we identified 10 articles in the first search and 166 in the second search.

Analysis of the hits revealed a direct discourse on the contribution of green roofs and green walls to public health and well-being in terms of regulating services such as reduction in temperature, pollution, and noise, but also in terms of positive environmental appraisal (**Table 2**). Within papers identified in the first structured literature search, the potential for green roofs and green walls to contribute to improved public health was discussed in relation to different spatial scales and contexts (**Table 2**). Most of the 37 articles reviewed focused only on the presence and location of green roofs and green walls, either measuring their impact in comparison with conventional roofs or measuring/simulating their impact on the surroundings. A number of the papers explored different aspects of design, such as species composition or substrate structures of green roofs and green walls (28 articles). Only a limited number of papers explored the status (either vegetation or substrate) of green roofs and/or green walls (five articles) in relation to effectiveness in providing public health-promoting ecosystem services.

The studies represented different governance structures, spatial scales, temporal phases for work with green roofs and green walls in health-promoting cities:

(1) *Planning for green roofs and green walls*. This included studies examining the presence and spatial location of green roofs and green walls for provision of health and well-being benefits on a more general level. These studies often compared conventional roof/walls with green roofs/walls or measured/simulated their impact on the surroundings, but without comparison of different variables such as vegetation and substrate.

TABLE 1 | Results of the second structured literature in terms of number of articles identified for each search string, number of review papers within the total, and reviews considered relevant for our analysis.

Search terms	Hits	Reviews (of which relevant reviews included)
Pollution OR pollutant	420	56 (Francis and Jensen, 2017; Corada et al., 2020; Liu et al., 2021; Ysebaert et al., 2021)
Noise OR sound	91	13 (Yang and Jeon, 2020)
Temperature	705	45 (Hunter et al., 2014; Charoenkit and Yiemwattana, 2016; Francis and Jensen, 2017; Cascone et al., 2019; Jamei et al., 2021; Liu et al., 2021)
Aesthet* OR preferenc* OR emotion OR appraisal	166	22 (Williams et al., 2019)

(2) *Design of green roofs and green walls.* This group of studies discussed and compared different types of green roofs and green walls, with regard to species and substrate and how they contribute to pathways for health and well-being associated with urban nature.

(3) *Management and maintenance of green roofs and green walls.* These studies touched upon the influence of maintenance and management of green roofs and green walls and their effect on the health pathways associated with urban nature.

Green Roofs and Green Walls as Part of a Health-Promoting City

Planning of Green Roofs and Green Walls Within the City – Location and Placement

A large proportion of the articles identified through the first structured literature search mainly focused on comparing green roofs/walls with conventional, not distinguishing specific types or just using one type in the empirical measurement/simulation.

Concerning the possibility of green roofs and green walls to contribute to temperature regulation, some of the studies provided support for this, mainly based on different types of simulations (e.g., Smith and Roebber, 2011; Herath et al., 2018; Gao et al., 2019, 2020; Huang et al., 2019; Zhu et al., 2021), but also experimental studies (e.g., He et al., 2020). Reviews by Bowler et al. (2010b), Santamouris (2015, 2016), Francis and Jensen (2017), Medl et al. (2017), Ghazalli et al. (2019), Jamei et al. (2021), and Liu et al. (2021) concluded that green roofs and green walls have potential for heat reduction, with the highest potential for temperature reduction in a dry climate (e.g., Smith and Roebber, 2011; Peng and Jim, 2013; Gao et al., 2020). The studies in our dataset also provided consistent evidence that the effect on temperature is highest during peak Urban Heat Island (UHI) periods, both in relation to season and to time of day (Bowler et al., 2010b; Speak et al., 2013; Santamouris, 2015; Solcerova et al., 2017; Gao et al., 2019, 2020).

Location in relation to wind direction also had an impact, with simulations by Zhang et al. (2019) showing that green roofs in upwind zones have better temperature reduction capacity for overall UHI mitigation. The review by Jamei et al. (2021) noted that few studies have been carried out specifically at pedestrian level or considering the thermal comfort for people. The few studies carried out showed that green roofs on lower-rise buildings have an impact on pedestrian level (Alexandri and Jones, 2008; Peng and Jim, 2013; Scharf and Kraus, 2019) and that installing green roofs on taller buildings has limited or no effect on pedestrian-level thermal comfort (Santamouris, 2015; Detommaso et al., 2021). For green walls/facades, some studies showed that their impact on the near surrounding outdoor air temperature is limited (e.g., Katsoulas et al., 2016), mainly occurs during peak solar hours (Cameron et al., 2014; Tan et al., 2014) and is limited to the close proximity of the wall (e.g., Wong et al., 2010; Cameron et al., 2014). Several studies emphasized the role of green roofs and green walls as part of an overall green infrastructure strategy to deal with the UHI effect, rather than use of green roofs and green walls as the only mitigation technique (Norton et al., 2015; Jamei and Rajagopalan, 2017; Santamouris et al., 2017; Herath et al., 2018; Zhu et al., 2021). Although green roofs and green walls were seen as less effective for thermal regulation than other green elements such as pocket parks and street trees, some studies concluded that they may play an important role when retrofitted in the dense urban environment (where space is limited) and spatially allocated to areas hosting groups vulnerable to heat exposure (Norton et al., 2015; Sanchez and Reames, 2019).

A factor related to cooling of the surrounding environment is *air quality*. Cold air is heavier than warm air and cooling slows down air circulation, while low temperatures also reduce the activity of the plants and their absorbing and filtering effect on air pollution. Cooling by green elements affects the airflow in such a way that air pollutants are dispersed close to roads, improving air quality (Baik et al., 2012). Less light also reduces surface-initiated photochemical reactions such as formation of ozone (Rowe, 2011), which is highly damaging to human health. Ozone is formed when NOx is transformed and reductions in gaseous pollutants is another way to reduce the amounts at ground level. Green roofs and green walls were reported to remove some gaseous pollutants (e.g., NOx, SOx, CO_2) from the air, but their effectiveness in removing other gaseous pollutants was less clear (Li et al., 2010; Rowe, 2011; Speak et al., 2012; Francis and Jensen, 2017; Medl et al., 2017; Liu et al., 2021).

Particulate matter (PM) in the air causes respiratory damage and negatively affects human health. Green roofs and green walls are reported to be efficient in reducing PM concentrations in air (Rowe, 2011; Francis and Jensen, 2017; Medl et al., 2017; Ghazalli et al., 2019; Weerakkody et al., 2019; Pettit et al., 2021). In general, larger particles are filtered from the air more efficiently by vegetation than smaller particles (Weerakkody et al., 2019; Tomson et al., 2021). Srbinovska et al. (2021) found that living wall plants removed up to 99% of coarse particles (PM 2.5), but the removal rate was less than 1% for particles less than 0.5 μm in diameter. A comparison with a non-vegetated surface in that study showed that the contribution

TABLE 2 | Articles (*n* = 69) recovered through a search in Scopus using the terms "green roof" OR "green wall" AND health OR wellbeing OR well-being OR aesthetic and after a scan of the abstracts.

Author	Type of study	Focus	Type	Context	Species used	Specific details
General						
Berardi et al., 2014	Review		GR			
Ghazalli et al., 2019	Review		GW			
Medl et al., 2017	Review		GW			
Temperature						Temperature measurement
Bowler et al., 2010b	Review		GI incl. GR and GW	NA	NA	NA
Detommaso et al., 2021	Simulation	Planning	GI incl. GR	Urban	Extensive – no species listed though height 30 cm, LAI 1.50.	Outdoor temperature and mean radiant temperature, predicted mean and physiological equivalent temperature
Gao et al., 2019	Simulation	Planning	GR	Metropolitan region	Not specified	Surface temperature
Gao et al., 2020	Simulation	Planning	GR	Metropolitan region	Not specified	Surface temperature
He et al., 2020	Experiment	Planning, design	GR	Urban	Sedum	Surface temperature, heat flux and humidity ratio
Herath et al., 2018	Simulation	Planning	GI incl. GR and GW	Urban	Grass	Air temperature at 1.5 m height
Huang et al., 2019	Simulation	Planning	GR	Metropolitan area	Not specified	Air temperature at 2 m height
Jamei and Rajagopalan, 2017	Simulation	Planning	GR	Urban	Not specified	Physiological equivalent temperature Mean radiant temperature
Li et al., 2014	Simulation	Planning, management	GR	Metropolitan area	Not specified	Surface and near-surface temperature, atmospheric moisture
Lin et al., 2017	Experiment	Design	GR	Urban	12 ornamental plants most commonly used for extensive green roofs in Taiwan	Surface temperature, solar radiation intensity, substrate water content
Norton et al., 2015	Review + observation	Planning	GI incl. GR and GW	Urban	Not specified	Not specified
Peng and Jim, 2013	Simulation Experiment	Planning	GR	Peri-urban to urban	Grass Dense trees	Physiological equivalent temperature (PET) 1.2 above roofs and 1.2 m above street level ground
Pisello et al., 2018	Review		GR	NA	NA	NA
Sanchez and Reames, 2019	Simulations	Planning	GR	Metropolitan region	Not presented	Land surface temperature
Sangkakool and Techato, 2017		Design	GR			
Santamouris et al., 2016	Review		GR + mitigation tech.	N/A	N/A	N/A
Santamouris, 2015	Review		GR	N/A	N/A	N/A
Scharf and Kraus, 2019	Simulation	Planning, design	GR	Urban	Extensive	Air temperature, PET
Smith and Roebber, 2011	Simulation	Planning	GR	Urban	Not specified	Apparent temperature
Solcerova et al., 2017	Field experiment	Planning, management	GR	Urban	Sedum	Air temperature (15 and 30 cm above)
Speak et al., 2013	Field experiment	Planning, management	GR	Urban	Intensive-mixed species: Rubus fruticosa, Buddleja davidii, Plantago lanceolata, Juncus sp., Aster novi-belgii, Senecio jaobaea, Agrostis stolonifera, Festuca rubra	Air temperature (30 cm above)

(Continued)

TABLE 2 | (Continued)

Author	Type of study	Focus	Type	Context	Species used	Specific details
Tan et al., 2017	Experiment	Design	GR	Urban	Cyathula prostrata	Surface temperature, substrate temperature
Zhu et al., 2021	Simulation	Planning	GR and GW + cool	High residential	Hedera helix, Funcia sp.	Building air temperature, canopy air temperature
Air pollution						Type of pollutants
Abhijith et al., 2017	Review	Planning, design	GR, GW	Urban pollutant reduction efficiency	Trees, perennials	General
Alsup et al., 2013	Simulation/ observation	Planning	GR			
Baik et al., 2012	Simulation	Planning	GR	Street canyons	Model green roof vegetation	NOx
Baraldi et al., 2019	Experiment	Design	GR	Lab study Plant physiology and morphology	Perennial green roof species	General
Currie and Bass, 2008	Simulation	Planning, design	GR			
Gnecco et al., 2013	Field experiment	Planning	GR	Campus park building	Grass herbaceous plants	PM, metals,
Joshi and Ghosh, 2014	Simulation	Planning	GW	Road side	Climber, Vernonia elaeagnifolia	SO2
Jung et al., 2016	Experiment/ simulation	Planning	GR	Urban	Model green roof vegetation	BOD
Li et al., 2010	Experiment, simulation	Planning	GR	Urban	Perennial green roof vegetation/Ixora chinensis	CO2
Li and Babcock, 2014	Review		GR	Urban	Perennial roof vegetation	General
Morakinyo et al., 2016	Simulation	Planning	GW	Road-side	Model green wall vegetation	PM 2.5
Ottelé et al., 2011	Simulation	Design	GR and GW			
Pandey et al., 2015	Field Experiment	Design	GW	Urban Pollution tolerance	Climbers	SO2, NO2, ozone, PM10
Paull et al., 2018	Experiment	Design	GW			
Paull et al., 2021	Experiment	Design	GW	Urban field	Perennial green wall plants	Ambient pollution
Paull et al., 2019	Experiment	Design	GW	Laboratory	Native Australian perennials	PM, VOC, CO2
Pettit et al., 2021	Experiment	Design	GW	Road side reduction efficiency	Westringia fruticosa (coastal rosemary), Myoporum parvifolium (dwarf native myrtle), Strobilanthes anisophyllus (goldfussia) and Nandina domestica (heavenly bamboo)	NO2, O3, PM 2.5
Rowe, 2011	Review		GR	Urban		General
Srbinovska et al., 2021	Field monitoring	Planning	GR	Open urban areas	Urban vegetation	PM
Speak et al., 2012	Field experiment	Planning, design	GR	Urban city center	Creeping bentgrass (Agrostis stolonifera), red fescue (Festuca rubra), ribwort plantain (Plantago lanceolata) and sedum (Sedum album)	PM 10
Tomson et al., 2021	Experiment	Design	GR and GW	Lab experiment	Black she oak (Allocasuarina littoralis), monkey rope vine (Parsonsia straminea), fringed wattle (Acacia fimbriata dwarf), and grass tree (Xanthorrhoea johnsonii)	PM

(Continued)

TABLE 2 | (Continued)

Author	Type of study	Focus	Type	Context	Species used	Specific details
Vera et al., 2021	Experiment	Design	GR and GW	Plant reduction efficiency	Sedum album, Lampranthus spectabillis, Sedum spurium P, Lavandula angustifolia, Erigeron karvinskianus, Aptenia cordifolia, and Sedum palmeri.	PM
Weerakkody et al., 2019	Experiment	Design	GW	Road side reduction efficiency	Buxus sempervirens	PM
Yang et al., 2008	Simulation	Planning	GR		NA	
Ye et al., 2013	Simulation	Design	GR	Plant upptake	Sedum lineare Thunb, Sedum sarmentosum Bunge, Portulaca oleracea L.,	Metals
Zhang et al., 2015	Field experiment	Planning, design	GR	Urban	Buddha nail (Sedum lineare Thunb)	Nutrients, OC, Metals
Noise						Noise reduction, decibels
Connelly and Hodgson, 2013	Experiment	Planning, design, management	GR	Lab and in-field	Sedum	Transmission loss dB/Frequency
Connelly and Hodgson, 2015	Experiment	Planning, design	GR	Lab and in-field	Extensive vegetated	Diffuse absorption coefficient/frequency
Jang et al., 2015	Simulation	Planning	GR and GW			
van Renterghem and Botteldooren, 2014	Experiment	Management	GR	Unclear	Sedum	Noise attenuation dB
van Renterghem and Botteldooren, 2009	Simulation	Planning	GR	Street level	Extensive and intensive	Sound pressure level (dBA)
van Renterghem et al., 2013	Simulation	Planning	GR and GW	Street level	Not specified	Absorption coefficient/frequency
Veisten et al., 2012	Simulation	Planning	GR and GW	Street level		Sound pressure level
Yang et al., 2012	Experimental/ simulation	Design	GR	Experiment	Not specified	Absorption coefficient/frequency
Environmental appraisal – mental health						Well-being function (aesthetic or al)
Collins et al., 2017	Experiment	Design	GW	Urban	Variety of plant species	
Fernandez-Canero and Gonzalez-Redondo, 2010	Review		GR	N/A	N/A	N/A
Fernandez-Cañero et al., 2013	Experiment/E	Design	GR	Urban	Variety of plant species	Aesthetic
Jungels et al., 2013	Experiment	Design	GR	Peri-urban to urban	Sedum, grasses, mix of perennial plant species	Aesthetic
Lee et al., 2014	Experiment	Design	GR	Urban	Variety of low growing plant species	Aesthetic Psychological
Liberalesso et al., 2020	Review	Design	GR, GW	Hostel buildings	Not specified	Aesthetic Psychological
Loder, 2014	Interviews	Planning, design	GR	Metropolitan area (Chicago, Toronto)	Prairie-style Sedum, grass-like Meadow-like	Aesthetic Psychological
Mesimäki et al., 2019	Experiment	Design	GR	Urban	Mosses Sedum	Aesthetic Psychological Multisensory experiences
Vanstockem et al., 2018	Case study	Design	GR	Imagine context situation	Sedum Herbaceous species	Aesthetic
Washburn et al., 2016	Field experiment	Planning	GR	Peri-urban (airport area buildings)	Stonecrop species Sedum, Phedimus, Hylotelephium	
Williams et al., 2019	Review	Planning, design	GR	Metropolitan area	Variety of plant species	Aesthetic Psychological

Each article was classified with regard to type of study, focus (e.g., planning, design and/or management), and type (GR, green roof; GW, green wall).

of the green area was on average 25% reduction for PM 2.5 and 37% for PM 10 (Srbinovska et al., 2021). Unfortunately, smaller particles are more damaging to human health than larger particles (WHO, 2016), reducing the benefits from plants in this regard.

However, removal of pollutants from the air may be damaging to the plant, and some studies argued that the plants which are most efficient in removal would not survive long in polluted environments due to accumulation of contaminants in plant tissues (e.g., Paull et al., 2019). However, a later study by the same authors found that many species of plants exposed to urban air can withstand the polluted environment and show no signs of reduced vitality compared with non-exposed plants (Paull et al., 2021).

Overall, the articles in our dataset indicated that green roofs and green walls have the potential to reduce pollutants with negative effects on human respiratory functions, particularly when these green elements are located in the proximity of pollution sources such as traffic (Medl et al., 2017). A study by Morakinyo et al. (2016) highlighted the role of different horizontal/vertical patterns and magnitudes of upwind and downwind flow on relative concentrations of pollutants. These factors are depending on wind conditions and green element type and dimensions. Green walls may be able to reduce air pollution at pedestrian height, as indicated in the study by Morakinyo et al. (2016), but the interaction between filtering capacity and aerodynamic effect still needs to be evaluated. The location of a living wall or green roof is probably most critical for their efficiency in removing pollutants, with proximity to the pollution source and the effect on local wind conditions being the most important factors (Medl et al., 2017).

Studies exploring the *noise reduction* effect of green roofs and green walls showed that they can potentially have a positive affect on absorption and decrease transmission of noise (Veisten et al., 2012; Yang et al., 2012; Connelly and Hodgson, 2013, 2015; van Renterghem et al., 2013; van Renterghem and Botteldooren, 2014; Medl et al., 2017). Green roofs were reported to be more effective in reducing noise in quiet courtyards (van Renterghem et al., 2013) and at traffic noise levels over 1 kHz (Jang et al., 2015), while vegetated façades were reported to be better for narrow city street canyons (van Renterghem et al., 2013) and for reducing low-frequency traffic noise (Jang et al., 2015). While this gives some guidelines at city scale, Yang et al. (2012) identified a need for site-specific analysis of configuration and position of the system in order to maximize the reduction of noise. While those studies highlighted the noise reduction potential of green roofs and green walls, these green elements can also contribute toward positive noise in the form of bird life (Washburn et al., 2016), which has been shown to have a positive effect on estimated well-being (Hedblom et al., 2017).

Visual and physical access is evidently a key location-related aspect affecting the possibility of green roofs and green walls to contribute to positive preferences and emotion and to psychological well-being, human experiences associated with *environmental appraisal*. Several studies in our dataset emphasized the positive effect that viewing green roofs can have on esthetic enjoyment and provision of restorative experiences (Williams et al., 2019), such as emotions, affect, and psychological well-being (e.g., Ghazalli et al., 2019; Wong et al., 2010; Lee et al., 2014; Loder, 2014). Installation of green roofs and green walls at the planning stage could also contribute to local esthetic improvement (Liberalesso et al., 2020), and in the long-term leads to intensification of cultural ecosystem services and increased level of identity and sense of place (Eliasson et al., 2018).

Implementation of Green Roofs and Green Walls – Design Parameters Affecting Health Pathways

Analysis of the literature in our dataset showed that substrate and species composition are the main design parameters affecting health pathways. Substrate type, and particularly the depth of substrate, were shown in some cases to have a positive relationship on provision of regulating services such as *temperature* (Santamouris, 2015; Charoenkit and Yiemwattana, 2016; Jamei et al., 2021), *air quality* (Rowe, 2011), and *noise regulation* (Connelly and Hodgson, 2015). The depth and composition of the substrate are also key factors determining the number of species that can be grown and their potential to thrive in green roofs and green walls. Studies comparing the effect of different species showed a positive relationship between canopy density and leaf area index (LAI), and regulating effects such as *thermal reduction* (Kolokotsa et al., 2013; Cameron et al., 2014; Hunter et al., 2014; Lin et al., 2017) and *air quality improvement* (Rowe, 2011; Baldauf, 2017).

With regards to *temperature regulation*, in relation to substrate types the study by Tan et al. (2017) showed effects of substrate composition, with the highest temperature found when topsoil was used as substrate and demonstrated that incorporation of a water retention layer can have positive effects in retaining soil moisture and on evapotranspiration rate. Vegetation coverage ratio was reported to be an important factor in the ability of green roofs or green walls to contribute to temperature reduction (Fang, 2008; Berardi et al., 2014; Lin et al., 2017; Ghazalli et al., 2019; Zhang et al., 2019). However, vegetation traits such as LAI and leaf morphology were also shown to play an important role (e.g., Fang, 2008; Morau et al., 2012; Berardi et al., 2014; Jamei et al., 2021), increasing the shading effects of the vegetation. The studies in our dataset were limited in number of species included, and with few repetitions, something also highlighted as a limitation by Ysebaert et al. (2021). Species reported to have good cooling effects in green walls include *Salvia* (Monteiro et al., 2017; Jamei et al., 2021), *Hedera* and *Stachys* (Cameron et al., 2014). Different species were found to contribute to cooling in different ways, with *Fuchsia* spp. providing evapo-transpiration cooling, while *Jasminum* and *Lonicera* contributed shade cooling (Cameron et al., 2014). Other studies showed that the vegetation must be dense to provide cooling effects (Chang et al., 2007; Bowler et al., 2010b). According to Lin et al. (2017), locations with high intensity solar radiation and a more sophisticated design of green roof (with high plant coverage, plant height, albedo, and canopy volume) can be expected to contribute more toward cooling capacity, and hence the effect would be more beneficial.

In relation to *air quality*, the structure and layout of the vegetation is important and studies have shown that vegetation density, LAI, thickness, and height are important for air flow and speed, where particles are deposited when the air speed is reduced, and vegetation alters the flow of the air (Baldauf, 2017). Species traits has been shown to influence ability for air quality purification (Liu et al., 2021; Ysebaert et al., 2021). For instance, leaf morphology is an important factor in PM removal (Speak et al., 2012; Paull et al., 2019) and species with high photosynthetic capacity, stomatal conductance, and transpiration are efficient in removing all gaseous pollutants from the air (Baraldi et al., 2019). A review by Liu et al. (2021) concluded that air quality purification ability is higher for intensive green roofs than for more extensive green roofs. But also differences among species occur. Speak et al. (2013) showed that *Agrostis stolonifera* and *Festuca rubra* are more effective than *Plantago lanceolata* and *Sedum album* at PM 10 capture. However, plant vitality may be negatively affected by plant uptake of pollutants. Several studies (Paull et al., 2018, 2021) reported that the most common green wall plant species are able to withstand highly polluted environments. There were no conclusive results on species differences on the best capacity for survival and pollutant removal (Paull et al., 2021), although native Australian species seem to have lower capacity (Paull et al., 2019). In one study, edible plant species (*Sedum lineare*, *Sedum sarmentosum,* and *Portulaca oleracea*) grown on green roofs were found to accumulate heavy metals, especially cadmium (Cd), at levels rendering the plants inedible (Ye et al., 2013). The effect of pollutant remediation on the filtering plants needs further research.

The *noise absorption* effect of green roofs and green walls is reported to be dependent on substrate depth, organic matter content, plant establishment, and moisture content (Yang et al., 2012; Connelly and Hodgson, 2015). Noise absorption increases with substrate depth, percentage organic matter, and plant establishment (Connelly and Hodgson, 2015). However, the configuration of the system seems to be more important (Yang et al., 2012), although more systematic studies on different type of layouts are required. Another aspect of noise is its transmission, which decreases for roofs with vegetation, particularly in the case of low-frequency noise (Connelly and Hodgson, 2013).

Design parameters of green roofs and green walls, such as species composition, are linked in the literature to health pathways and associated *environmental appraisal*. An attitude study by Liberalesso et al. (2020) found that potential hostel users support integrated installation of green infrastructure and consider that green roofs and green walls could provide esthetic improvement and stimulate sense of well-being. Analysis of the literature indicated that building-integrated vegetation, such as ivy facades and meadow-inspired roof vegetation appear to be more aesthetically pleasing and have the ability to generate higher restorative qualities than e.g., sedum or turf roof vegetation (White and Gatersleben, 2011). However, Mesimäki et al. (2019) indicated that low-grown grassy vegetation surrounded by a dense urban area can also generate recreational benefits, while Jungels et al. (2013) found that grass-dominated green roofs generate negative esthetic reactions compared with sedum-dominated or mixed perennials, which were experienced as fresh, innovative, and beautiful. One explanation for this is their messiness (Jungels et al., 2013; Loder, 2014). Green roofs

Planning	Design	Management - Maintenance
Regulatory ecosystem services - reduction of heat, pollution and noise • Spatial match problem (pollution, noise, UHI) with vulnerable groups • Positive effects are within short distance for pollution, noise and temperature (0.4 m) – placement should be where people are = at pedestrian level • Placement should be in dry climate with high air quality and sunexposed for maximum ambient temperature reduction • Green elements affect the air flow to spread near roads, which improves the air quality. • Green roofs for high and low frequency traffic noise • Green walls at street canyons and green roofs on courtyards for noise reduction. ***Cultural ecosystem services – socio-behavioural pathways*** • Visual/physical access needed for environmental appraisal	***Regulatory ecosystem services - reduction of heat, pollution and noise*** • High organic matter increases noise absorption and air quality • Structure and layout of the vegetation influence air quality • Deeper substrate is positively correlated to heat, noise and pollution reduction, and support a more layered vegetation. • More vegetation layers/LAI/vegetation depth is positively correlated to heat, noise and pollution reduction. ***Cultural ecosystem services – socio-behavioural pathways*** • Variety in vegetation structures (trees, shrubs, perennials, succulents) has a higher contribution to environmental appraisal and in turn emotion, affect and psychological well-being. • Flowering species and variety of colours has a higher contribution to environmental appraisal and in turn emotion, affect and psychological well-being.	***Regulatory ecosystem services - reduction of heat, pollution and noise*** • Well irrigated substrate is positively correlated to heat reduction, and provide more and healthier vegetation. • Well irrigated substrate risks leaking pollutants. • A healthy vegetation with a large leaf biomass will capture and take up pollutants more efficiently. ***Cultural ecosystem services – socio-behavioural pathways*** • Messiness of the vegetation affects the environmental appraisal and in turn emotion, affect and psychological well-being.

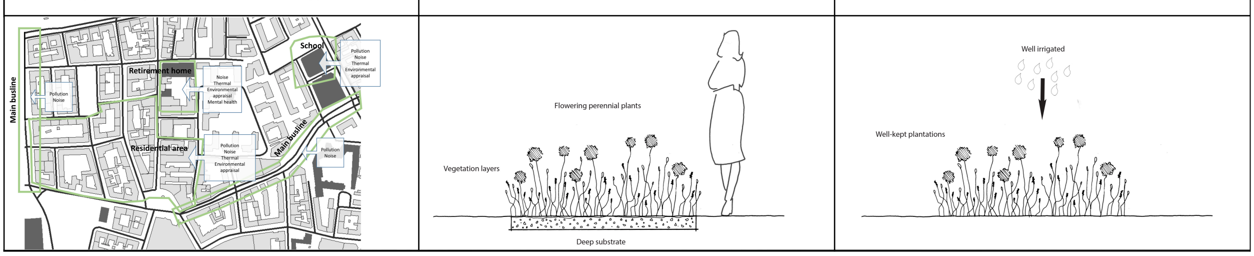

FIGURE 2 | Guidelines for optimizing the effect of green walls and green roofs in delivering public health-supporting ecosystem services.

that contain a variety of colors and vegetation structures are reportedly more likely to be preferred if well designed and regularly maintained (e.g., Fernandez-Cañero et al., 2013). Some studies indicated that flowering vegetation has higher restorative value than succulents (Lee et al., 2014), as well as being positively associated with creative thinking, health, well-being (Loder, 2014), and increased sustained attention (Lee et al., 2015). These findings stress the need for careful design of green roofs and green walls to supply environmental appraisal, especially since research has shown variations in their provision of psychological benefits (Williams et al., 2019) and fulfill people's desires (Mesimäki et al., 2017).

Maintenance and Management of Green Roofs and Green Walls to Maximize Public Health Benefits

Concerning the effects of maintenance and management on the performance of green roofs and green walls and the relationship to public health effects, very few studies in our dataset looked specifically at those aspects. Similarly, few studies looked at positive health-contributing factors over the longer term with regard to roof life-span, despite a call for such studies (e.g., Buffam et al., 2016).

In efforts to ensure that green roofs and green walls fulfill their potential to provide public health benefits, a key aspect is irrigation, which affects both substrate and vegetation health. Well-saturated substrate has greater *cooling capacity* (Li et al., 2014; Santamouris, 2015; Solcerova et al., 2017; Jamei et al., 2021) compared with dry substrate. In relation to *pollutant*, the study by Todorov et al. (2018) showed that a well-irrigated system may leach pollutants to a higher extent, and hence systems for handling this negative effect need to be put in place. For *noise reduction*, the effect of substrate water content is suggested to be limited (e.g., van Renterghem and Botteldooren, 2014).

Irrigation also affects the health of vegetation, manifested by a higher LAI density and hence better effect when it comes to the reduction of *temperature*, *pollution*, and *noise* (Speak et al., 2013; Hunter et al., 2014). Several studies on the effect of green roofs and green walls on *environmental appraisal* concluded that maintenance is an important aspect for preferences (e.g., Fernandez-Cañero et al., 2013; Jungels et al., 2013; Loder, 2014), with the presence of scruffy and dried vegetation considered negative.

CONCLUSION AND FUTURE RESEARCH

Our analysis of the relevant literature showed that green roofs and green walls can supply regulating ecosystem services and cultural ecosystem services, supporting pathways for public health. This effect could be maximized by adequate planning, design, and management of the resources (**Figure 2**).

Analysis of the literature provided strong evidence for regulating ecosystem services (heat reduction, improved air quality and noise reduction) and cultural ecosystem services (improved affect) from green roofs and living walls. *Heat reduction* through vegetation has shown to have a strong link to reduction of CVD (cardiovascular diseases) mortality, as well as all-cause mortality and mental disorders. The reviewed literature provided good evidence of the potential contribution of green roofs and green walls within an overall green infrastructure strategy in mitigating UHI, and thereby potentially reducing CVD mortality, all-cause mortality, and mental disorders in urban areas. This potential appeared to be greatest for urban centers, which are predicted to be significantly affected by increases in temperature due to future climate change. Due to the positive effects occurring only within a short distance from the installation, adequate planning should be carried out before investments in green roofs and green walls is done, and urban morphology and locations of vulnerable human groups to achieve the best health effects in relation to UHI should be taken into account. Through appropriate design of these systems, with plants with a high LAI in well-irrigated and deep substrate, the positive effects could be increased compared with more extensive and thin types of green roofs. However, providing moist systems with optimal plants is a challenge in areas suffering from water shortages, and for which the benefits would be greater.

Air pollution affects human health mainly through an increase in respiratory illnesses such as asthma, a higher incidence of CVD, and impaired neural development and cognitive capacities. Green roofs and green walls have the potential to mitigate air pollution, through reduction of gaseous pollutants such as NOx, SOx, and CO_2, but also PM (with higher particles more effectively filtered), and with the largest effect received close to the air pollution source. When designing green roofs and green walls, the effectiveness of these could be improved through design and layout of the system taking the pollutant source and dominant wind conditions into account. This includes also the type of substrate as well as vegetation types used. For instance, using a heterogeneous topography of the vegetation layer and species with high LAI, photosynthetic capacity, stomatal conductance and transpiration, the filtering and uptake of gaseous pollutants could be improved. Providing a system that is moist provides healthier plants that in turn has better air filtering potential.

There is evidence that *noise* has a relationship with public health aspects such as annoyance, sleep disturbance, CVD and cognitive impairment. Studies have shown that green roofs and green walls could absorb and decrease transmission of noise, though some suggestions are that green roofs works best in already fairly quiet court yards and green walls being better for narrow street canyons. Both have the potential of adding positive sound through the potential habitat for birds. The depth of substrate as well as percentage organic matter is positively influencing the ability of green roofs to deliver noise regulating, while level of water content is less clear. With regards to the vegetation, plant cover, but also canopy density and leaf area index are also important to maximize in order to gain maximum noise regulating effects.

For *environmental appraisal*, the study by van den Bosch and Ode Sang (2017) showed strong evidence of a reduction in all-cause mortality and CVD mortality, but weak evidence of a reduction in mental disorders. In our review, only a limited number of studies focused on environmental appraisal from green roofs and no studies were identified for green walls. The available evidence indicates that, to achieve the environmental appraisal effect from green roofs and green walls, they need to be visually accessible at a minimum. The status and character of the vegetation also appear to be important in maximizing this effect. This means providing a sufficiently deep substrate level to support healthy, well-irrigated vegetation with a variety of vegetation structures and colors. However, these studies included in this review in relation to environmental appraisal have mostly been carried out in Europe, North America and Australia, so the global validity is unclear. Recent studies on global validity of landscape preference has questioned this to be the case (e.g., Hägerhäll et al., 2018).

Our review of the literature focused on positive health impacts, but green roofs and green walls could also have negative effects in some cases, such as emissions of nutrients and heavy metals through stormwater run-off or contamination of edible species, posing potential health risks. Other disservices identified for green infrastructure in general included exposure to allergenic pollen, presence of animals as disease vectors, and discomfort from the presence of animals or their droppings, although research on these in relation to green roofs and green walls was sparse.

Compared with similar vegetation on the ground, green roofs and green walls are more costly in terms of installation and maintenance, which limits their cost-effectiveness as a city-wide NbS strategy. However, in a dense city setting they can provide a space efficient NbS for areas where other types of urban vegetation are not feasible and can provide public health benefits for residents in those areas. In general, the ecosystem services from green roofs and green walls appear to be most efficient in providing public health benefits when located close to where people live or work. Placing these structures close to where people spend time, i.e., on low buildings and structures close to the ground, will have the greatest effect on human health. In the case of noise and particle pollution sources such as roads, a location close to the source would be most efficient.

When analyzing the positive contribution that green roofs and green walls could make to public health and well-being, we focused on the public health pathways provided by regulating and cultural ecosystem services. However, our analysis showed that, in order to achieve these positive effects, well-functioning green roofs and green walls are needed. For example, a thin substrate layer with insufficient level of irrigation loses its cooling ability and noise reduction capacity, as well as limiting the variation in vegetation structure (and in turn in flowering species and variety of colors), providing a lower level of environmental appraisal compared with non-vegetated roofs.

While the literature includes empirical studies covering most aspects relating to public health pathways, there is a lack of studies on the long-term effect of ecosystem services supplied by green roofs/walls. For instance, the long-term effects of green roofs and green walls in sustained reduction of noise, temperature, pollution, and esthetic values are not well studied. There is also a lack of research on how to maximize ecosystem services that support public health and well-being through adequate and cost-effective maintenance of green roofs and green walls across the seasons.

AUTHOR CONTRIBUTIONS

ÅOS initiated the manuscript, carried out the structured searches, and was the main responsible for the writing of the manuscript. PT contributed to the development of the analytical framework, was main responsible for the section on environmental appraisal as well as the illustrations, and contributed to the development of the conclusions and future research. A-MF was main responsible for the sections on reduction of air pollution and contributed to the conclusion and future research. All authors contributed to the article and approved the submitted version.

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Spatiotemporal Dynamics of Ecosystem Services and the Driving Factors in Urban Agglomerations: Evidence From 12 National Urban Agglomerations in China

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The natural environment provides multiple ecosystem services for urban development and human quality of life. Given that current cities interact with each other and form urban agglomerations, understanding the spatiotemporal changes in ecosystem services and the driving forces is crucial for sustainable urban development. Using 12 national-level urban agglomerations as a case study, this paper quantifies the spatial patterns of multiple ecosystem service values from 2000 to 2015 and assesses how natural and socioeconomic factors contribute to such changes by using ordinary least squares (OLS) and geographically weighted regression (GWR). The results show the following: (1) spatial discrepancies of ecosystem services exist both in and between urban agglomerations, and ecosystem service values are reduced in more than 70% of urban agglomerations at a rate ranging from 0.02 to 4.27%; (2) elevation, precipitation, and fraction of woodland have positive impacts on ecosystem service values in urban agglomerations; while gross domestic product (GDP), population, and proportion of built-up area have negative effects; (3) both natural and social driving factors impact the ecosystem services of different urban agglomeration in different ways, according to the differences in their driving degrees. We categorized 12 urban agglomerations in China into six typical types: natural-factor dominated, socioeconomic-factor dominated, policy dominated, balanced, natural-factor inclined, and socioeconomic-factor inclined. Our results can be used to inform decision makers and urban planners to propose explicit location strategies to balance natural protection and socioeconomic development and ultimately promote sustainable urbanization across the nation.

Keywords: urban agglomeration, ecosystem service, driving factor, spatiotemporal dynamics, urban ecology

INTRODUCTION

With the continuous growth of urbanization, the marginal benefits brought by the individual spatial expansion of cities will be greatly reduced (Han et al., 2019). Clustered urban agglomeration development has been adopted as a major form for all countries to enhance urban competitiveness (Fang, 2015; Shi et al., 2020). Scholars from Great Britain, France, the Soviet Union, Germany, and China formed various concepts of urban agglomeration based on the situations of their home countries. Examples include "Metropolitan Regions" (Fawcett, 1932), "Megalopolis" (Gottmann, 1957), "Ecumenopolis" (Doxiadis, 1970), and "City-region" (Ng and Tang, 2013), which feature different research perspectives with respect to demographic, economic, society, function, industry, and the natural environment. Until the end of the 20th century, the United Nations (UN) officially used "urban agglomeration" to describe the phenomenon of urban development (Fang and Yu, 2017). Currently, most scholars refer to an "urban agglomeration" as a metropolitan area led by one or two cores of megacities that exert impacts on several peripheral cities and towns economically, socially, and ecologically *via* multiple infrastructures with regards to transportation, energy, communication, logistics, and natural ecosystems, ultimately creating a clustered regional complex (Song, 2010; Fang and Yu, 2017; Liu et al., 2018; Sun and Zhao, 2018; Chen et al., 2019; Wang J. et al., 2019; He, 2020). In 2006, China's 11th 5-Year Plan proposed that urban agglomeration would be one of the key national strategies for the country's long-term urbanization (NPC, 2006). The subsequent 13th Five-Year Plan (2015–2020) further specified the strategic layout of nineteen major urban agglomerations to be established in the future (NPC, 2016). The urban agglomeration model has become an important national development strategy for China to enhance the international competitiveness of cities.

Although an urban agglomeration is advantageous in terms of optimizing urban-rural resources and regional industrial structures (Cao, 2015; Wang and Cui, 2017; Tian, 2019), their rapid expansion of built-up areas can easily lead to many problems such as ecosystem degradation, habitat disturbance and environmental pollution in surrounding natural systems (Ma et al., 2021). Through assessing the environmental vulnerability of the Yangtze River Urban Agglomeration during 2005 and 2017, Peng et al. (2019) identified that the driving factors of the ecological environment vulnerability of the Yangtze River city group included natural, socioeconomic and policy factors. Based on longitudinal studies on China's 342 cities during 2001 and 2016, Fan et al. (2019) found that China's urban agglomeration development was strongly associated with peripheral air pollution and that this association was gradually growing. Liu's research on the Changchun urban agglomeration revealed that highly clustered urbanization exerted ecological and environmental pressure due to increasing industrial investment and urban sprawl (Liu et al., 2017).

In 2005, the UN published *The Millennium Ecosystem Assessment* (Dooley, 2005; Carpenter et al., 2006) with the World Health Organization and United Nations Environment Program, in which the term "ecosystem service" was proposed to define the overall benefits humans obtain from the natural environment and ecosystem and subsequently established a mechanism to comparatively study the linkage between human wellbeing and the natural system. Scholars evaluate the interrelationship between urban agglomerations and natural ecosystems by adopting the concept of "ecosystem services." For example, Haase explored the spatiotemporal dynamics of ecosystem services in the Leipzig-Halle Region, Germany, and identified how external factors impact each individual ecosystem service, based on which an adaptive integrated multiscale framework is proposed for regional development (Haase et al., 2012). Sun et al. (2018) analyzed how ecosystem services respond to urban sprawl in the Atlanta Metropolitan area from 1985 to 2010 and simulated the development trend of ecosystem services in 2030. Chen W. et al. (2020) explored how natural and social driving factors affect the ecosystem services in the Yangtze River Urban Agglomeration in China during 1995 and 2015. The present empirical evidence relying on the concept of ecosystem services not only generates academic knowledge about how urban agglomerations and natural ecosystems interact with each other and the underlying mechanism but also informs decision makers and urban planners with supportive references for spatially sensitive policy and planning interventions.

A burgeoning number of studies have evaluated ecosystem services in China's urban agglomerations. Zhang et al. (2015) quantitively assessed how the value of ecosystem services changed in urban agglomerations along the coast of the Bohai Rim during 2000 and 2010. Li Z. et al. (2019) investigated the spatiotemporal patterns and cold/hot spots of ecosystem services in the Yangtze River Delta Urban Agglomeration in 2000. Gao et al. (2019) calculated the ecosystem service benefits and losses in the Yangtze River Delta urban agglomeration. These quantified evaluations of the ecosystem services of different urban agglomerations provide reference information and theoretical support for crafting urban agglomeration-related strategies and policies. In addition to monitoring the spatiotemporal dynamics of different ecosystem service values, emerging literature has begun to analyze the potential driving factors of ecosystem services. Liu et al. (2020) investigated how landscape patterns affected urban agglomerations in the Yangtze River and proposed that cross-regional collaborative governance among different regions is necessary to improve the development of ecosystem services in the entire region. Peng et al. (2019) explored the linkages between human activities and ecosystem services in Yangtze River Urban Agglomeration and revealed the heterogeneous effects of natural and social factors on different cities within the agglomeration at different time periods. Chen et al. (2019) assessed the interaction between ecosystem services and driving factors with respect to socioeconomic and policy making and provided recommendations for adaptive land-use models.

However, most research that has explored the driving factors of ecosystem services has focused on individual urban agglomerations (Cao, 2015; Zhang et al., 2015; Chen et al., 2019; Li T. et al., 2019; Li Z. et al., 2019; Peng et al., 2019; Liu et al., 2020). Due to the varying research time periods, methods, and objectives, it is difficult to compare the changing characteristics of ecosystems and driving factors in different urban agglomerations. In the practical realm, spatially explicit information of different urban agglomerations is needed to better inform policy makers to promote location-sensitive development strategies and planning. Therefore, this study comparatively examines 12 recently established urban agglomerations in China using consistent quantification methods to understand how natural and social driving factors affect the ecosystem services among different urban agglomerations and to provide reference information for location-sensitive policies and planning.

The objectives of this study are (1) to evaluate the ecosystem services of 12 major urban agglomerations in 2000, 2005, 2010, and 2015 using a quantitative approach and analyze the spatiotemporal dynamics, (2) to understand how natural and social driving factors influence the ecosystems of the 12 major urban agglomerations and the spatial characteristics, and (3) to provide location-sensitive policy and planning recommendations based on the analytical results.

MATERIALS AND METHODS

Study Area

The State Council of China's 13th Five-Year Plan (2015–2020) proposed nineteen urban agglomerations to be established in the future. Before June 2020, the development guidelines of 12 urban agglomerations were approved by the State Council and National Development and Reform Commission with specific planning borders and development objectives and were thus included in this study (NDRC, 2016). The 12 urban agglomerations are the middle reaches of the Yangtze River Urban Agglomeration (MRTR), Beijing-Tianjin-Hebei Urban Agglomeration (BTH), Harbin-Changchun Urban Agglomeration (HC), Chengdu-Chongqing urban agglomeration (CC), Yangtze River Delta urban agglomeration (YRD), Central Plain Urban Agglomerations (CP), Beibu Gulf urban agglomeration (BG), Guanzhong Plain Urban Agglomeration (GP), Hohhot-Baotou-Ordos-Yulin Urban Agglomeration (HBOY), Lanzhou-Xining Urban Agglomeration (LX), Guangdong-Hong Kong-Macau Greater Bay Area (GHM), and Yunnan Central Urban Agglomeration (YC). Detailed information on the selected urban agglomerations is in **Figure 1** and **Table 1**.

Data Collection

The study utilizes multiple data, including the administrative borders of each urban agglomeration, land-use maps, economic data of agricultural production, and driving factor data. Following previous studies (Gao and Wang, 2019;

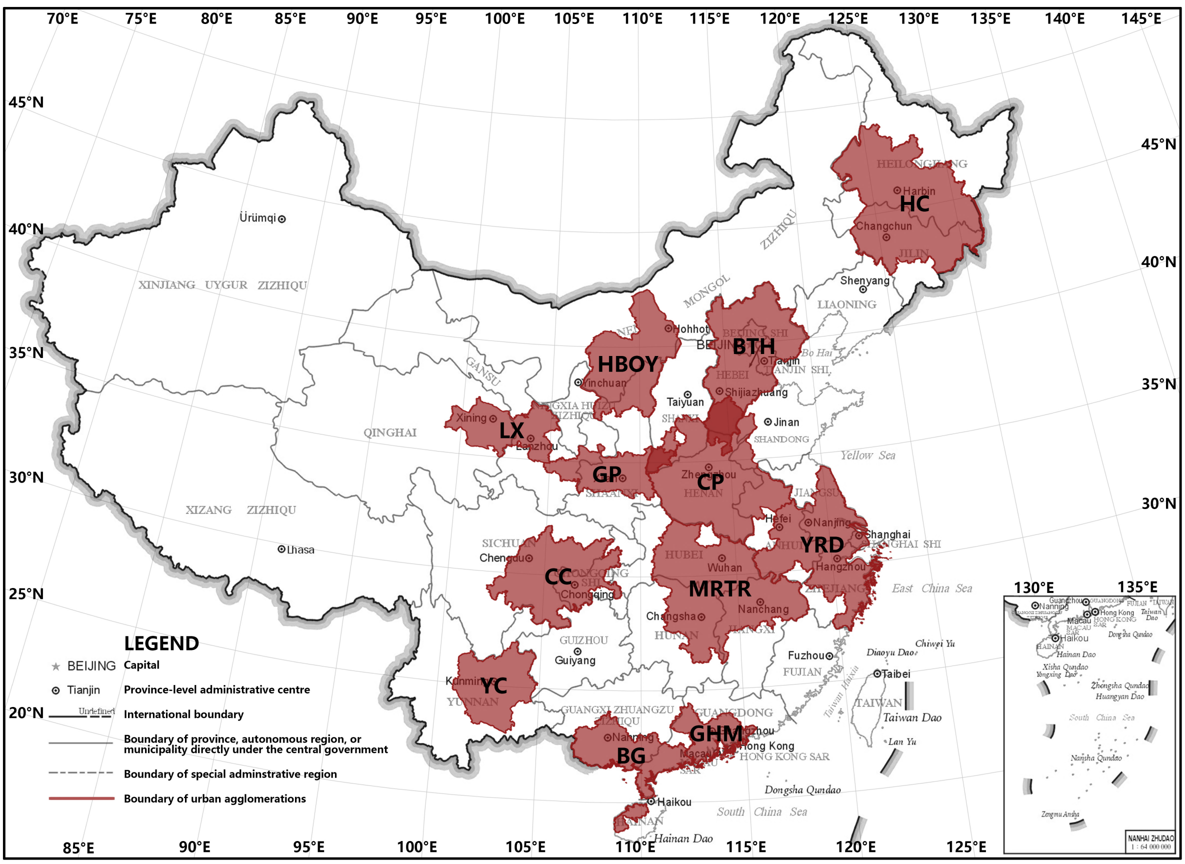

FIGURE 1 | Location distribution map of China's existing urban agglomerations. MRTR, middle reaches of the Yangtze River Urban Agglomeration; BTH, Beijing-Tianjin-Hebei Urban Agglomeration; HC, Harbin-Changchun Urban Agglomeration; CC, Chengdu-Chongqing urban agglomeration; YRD, Yangtze River Delta urban agglomeration; CP, Central Plain Urban Agglomeration; BG, Beibu Gulf urban agglomeration; GP, Guanzhong Plain Urban Agglomeration; HBOY, Hohhot-Baotou-Ordos-Yulin Urban Agglomeration; LX, Lanzhou-Xining Urban Agglomeration; GHM, Guangdong-Hong Kong-Macau Greater Bay Area; YC, Yunnan Central Urban Agglomeration.

TABLE 1 | Overview of China's existing urban agglomerations.

Name	Area (10,000 km^2)	Population (10,000)	Approval year	Proportion of woodland(%)	Proportion of built-up area(%)	Proportion of farmland(%)	Proportion of waterbody(%)	Climate zone	Main vegetation types	Location in China
MRTR	32.61	12,677	2015	47.11–47.43	2.78–4.12	38.36–39.56	7.20–7.57	Subtropical monsoon	Subtropical evergreen broad-leaved forest	Middle east
BTH	22.56	11,270	2015	20.25–20.27	8.12–9.39	50.18–51.15	3.16–3.26	Temperate monsoon	Temperate grassland, Warm temperate deciduous broad-leaved forest	North-east
HC	26.36	4,625	2016	35.71–35.88	3.46–3.69	46.51–46.77	3.74–3.83	Temperate monsoon	Cold temperate coniferous forests, Temperate mixed coniferous and broad-leaved forest	North-east
CC	18.50	10,015	2016	26.51–26.69	1.60–2.94	61.79–63.29	1.80–1.90	Subtropical monsoon	Subtropical evergreen broad-leaved forest	Central to west
YRD	21.17	15,401	2016	29.25–29.52	7.28–10.99	47.76–51.38	8.45–8.67	Subtropical monsoon	Subtropical evergreen broad-leaved forest	East
CP	28.66	16,353	2016	13.90–13.92	10.42–11.52	65.44–66.71	2.34–2.56	Temperate monsoon and subtropical monsoon	Temperate mixed coniferous and broad-leaved forest, Subtropical evergreen broad-leaved forest	Middle
BG	11.66	4,211	2017	55.97–56.24	3.74–4.37	31.38–31.96	3.93–3.99	Subtropical monsoon and tropical monsoon	Subtropical evergreen broad-leaved forest, Tropical rain forests, Monsoon forest	South
GP	10.71	4,038	2018	21.32–21.45	4.02–5.01	44.90–46.28	1.36–1.42	Temperate monsoon	Warm temperate deciduous broad-leaved forest, Subtropical evergreen broad-leaved forest	Central to west
HBOY	17.50	1,151	2018	4.14–4.76	1.49–2.56	18.23–18.71	2.12–2.15	Temperate continental and temperate monsoon	Temperate desert, Temperate grassland, Warm temperate deciduous broad-leaved forest	North-west
LX	9.75	1,526	2018	9.04–9.09	1.38–1.84	18.52–18.97	4.41–4.50	Plateau mountain and temperate monsoon	Alpine vegetation of Qinghai-Tibet Plateau, Temperate desert, Temperate grassland, Warm temperate deciduous broad-leaved forest	West
GHM	5.60	6,957	2019	54.21–55.63	7.79–13.17	22.70–26.15	7.63–8.21	Subtropical monsoon	Subtropical evergreen broad-leaved forest	South
YC	11.46	2,127	2020	49.28–49.35	1.17–1.61	20.37–20.72	1.19–1.21	Subtropical monsoon and tropical monsoon	Subtropical evergreen broad-leaved forest, Tropical rain forests, Monsoon forest	South-west

TABLE 2 | Summary of data sources.

Name of data	Data description	Data source
Administrative border	Obtained from official documents with detailed border.	National Development and Reform Commission of the PRC (https://www.ndrc.gov.cn/)
Land use	Land use map in 2000, 2005, 2010, and 2015. The Institute of Geographic Sciences and Natural Resources of the Chinese Academy of Sciences interprets Landsat satellite remote sensing data according to the national land classification standard. The classification system is shown in **Table 3**. The data accuracy is 1,000 m.	Data Center for Resources and Environmental Sciences, Chinese Academy of Sciences (www.resdc.cn)
Crop production	Crop production, area, and market value for each city in 2000, 2005, 2010, and 2015.	China Statistic Yearbook and Compendium of Chinese agricultural product data
Elevation (m)	Elevation in the urban agglomeration. The data accuracy is 30 m.	Geospatial data cloud platform (http://www.gscloud.cn/)
Coverage of vegetation (%)	Annual average NDVI value in 2000, 2005, 2010, and 2015. The data set is based on continuous time series SPOT/VEGETATION NDVI satellite remote sensing data, and the annual VEGETATION index data set is generated by the maximum value synthesis method since 1998. This data set can effectively reflect the distribution and change of vegetation cover in different regions of China at spatial and temporal scales. The data accuracy is 1,000 m.	Data Center for Resources and Environmental Sciences, Chinese Academy of Sciences (www.resdc.cn)
Precipitation (mm)	Annual average precipitation in 2000, 2005, 2010, and 2015. The data set is based on the daily observation data of more than 2,400 meteorological stations in China and generated through sorting, calculation and spatial interpolation. The data accuracy is 1,000 m.	
Humidity (°C)	Annual group humidity in 2000, 2005, 2010, and 2015. The data set is based on the daily observation data of more than 2,400 meteorological stations in China and generated through sorting, calculation and spatial interpolation. The data accuracy is 1,000 m.	
GDP per unit area (CNY/km^2)	Average GDP per unit area in 2000, 2005, 2010, and 2015. The data set is from the Chinese Academy of Sciences at the national county GDP statistics. It considers factors such as the land use type most closely related to human economic activity, night lighting levels, and density of residential areas. The data set uses the multi-factor weight distribution method with the administrative region as the basic statistical unit. GDP data distribution is allocated on the grid unit published data sets. The data accuracy is 1,000 m.	
Population density (person/km^2)	Average population density within urban agglomeration border in 2000, 2005, 2010, and 2015. The data set is the Chinese Academy of Sciences at the national county population statistics. The data set comprehensively considers factors such as the land use type most closely related to population, night lighting levels, the density of residential areas. The data set uses the multi-factor weight distribution method to the administrative region of the basic statistical unit. Population data distribution to is allocated to online space lattice published data sets. The data accuracy is 1,000 m.	
Proportion of woodland (%)	The proportion of woodland within urban agglomeration border in 2000, 2005, 2010, and 2015. From the land use data classification calculation. The data accuracy is 1,000 m.	
Proportion of built-up area (%)	The proportion of built-up area within urban agglomeration border in 2000, 2005, 2010, and 2015.	Tsinghua University Earth System Science Database (Gong et al., 2019)

Sun et al., 2019; Chen J. et al., 2020; Dai et al., 2020; Luo et al., 2020), driving factors are in two categories—natural and social factors—the former includes elevation, vegetation coverage, precipitation, and temperature, and the latter refers to GDP per unit, population density, and urban built area ratio within the entire administrative border. Detailed data sources and descriptions are provided in **Table 2**.

Data Collection

Assessment of Ecosystem Service Values

Constanza estimated the current economic value of 17 ecosystem services worldwide based on published studies and a few original calculations in 1997 (Costanza et al., 1997), which established a first approximation of the relative magnitude of global ecosystem services and made the potential values of different ecosystems more apparent for further studies. Later, Xie et al. adjusted the methods based on China's socioeconomic and natural ecosystem conditions and widely adopted them in many studies and practices (Xie et al., 2003; Jie et al., 2014; Zhang et al., 2015; Kang et al., 2018; Zhou et al., 2018; Hu et al., 2019).

Following Xie's approach, this study defines one ecosystem service value equal to 1/7 of the average market value per unit area yield of grain in China (Yu et al., 2005). This measurement is based on an ecological assets value table and adjusted price value by biomass, which was established *via* consultation of professional ecologists and adjusted through experiments. The approach can be used to evaluate the entire regional ecosystem

TABLE 3 | The coefficient of ecosystem service value per unit area in China's urban agglomerations [CNY/(Ha * year)].

Ecosystem services	Provisioning service		Regulating service				Supporting service		Cultural service
	Food production	Raw material production	Air purification	Climate adjustment	Hydrological regulation	Waste treatment	Soil conservation	Biodiversity preservation	Aesthetic value
Farmland	1359.84	530.34	979.08	1319.04	1047.07	1890.17	1998.96	1387.03	231.17
Woodland	448.75	4052.32	5874.50	5534.54	5561.73	2338.92	5466.55	6132.87	2828.46
Grassland	584.73	489.54	2039.76	2121.35	2066.95	1794.99	3046.04	2542.90	1183.06
Waterbody	720.71	475.94	693.52	2801.26	25524.15	20193.58	557.53	4664.24	6037.68
Unused land	27.20	54.39	81.59	176.78	95.19	353.56	231.17	543.93	326.36

Farmland refers to the land for planting crops, including ripe cultivated land, newly reclaimed land, recreational land, rotation and rest land, grass field rotation crop land; Land for fruit, crop, agriculture and forestry; Beaches and sea flats cultivated for more than 3 years. Woodland refers to the forest land for trees, shrubs, bamboo and coastal mangrove forests. Grassland refers to all kinds of grassland with grass coverage of more than 5%, including shrub grassland with grazing as the main part and sparsely forested grassland with canopy density of less than 10%. Waterbody refers to natural land area and land for water conservancy facilities. Unused land refers to sandy land, gobi, salt-alkali land, marsh, bare land and rock, alpine desert, tundra and other unused or difficult to use land.

TABLE 4 | Trends of different ecosystem services in 12 urban agglomerations in China during 2000–2015 (Unit: 100 million CNY).

Year	Land	MRTR	BTH	HC	CC	YRD	CP	BG	GP	HBOY	LX	GHM	YC
2000	Farmland	1323.04	1223.87	1635.22	1299.58	1205.81	2053.76	397.30	540.82	354.94	212.03	154.49	260.92
	Woodland	5646.54	1726.82	4490.44	1937.32	2466.01	1525.45	2476.45	886.83	279.72	359.67	1169.91	2209.89
	Grassland	116.40	574.53	270.30	204.36	115.97	296.93	78.63	463.91	1546.81	965.54	19.12	511.12
	Waterbody	1381.99	448.21	773.88	212.02	1138.09	413.55	280.59	91.45	234.40	282.69	278.43	86.27
	Unused land	3.90	3.95	31.61	0.24	0.07	0.44	0.24	0.29	61.11	15.22	0.04	0.31
	Total	8471.88	3977.39	7201.45	3653.52	4925.96	4290.13	3233.21	1983.31	2476.98	1835.15	1622.00	3068.51
2005	Farmland	1309.62	1212.87	1643.33	1289.41	1167.63	2037.60	394.30	533.78	349.14	209.77	140.32	259.66
	Woodland	5637.10	1726.36	4474.80	1949.90	2457.79	1523.58	2483.98	889.70	312.45	360.40	1156.07	2209.62
	Grassland	115.10	571.91	274.36	203.56	115.47	295.91	77.25	467.62	1525.52	966.54	17.96	511.71
	Waterbody	1438.79	437.79	755.13	212.51	1164.67	444.32	283.68	93.74	231.01	283.68	269.37	85.90
	Unused land	3.33	3.84	30.85	0.24	0.07	0.37	0.22	0.28	62.70	15.25	0.04	0.31
	Total	8503.94	3952.78	7178.48	3655.61	4905.63	4301.79	3239.43	1985.12	2480.82	1835.64	1583.76	3067.20
2010	Farmland	1302.90	1207.69	1641.59	1283.27	1143.01	2029.33	392.93	531.76	348.52	209.57	135.96	258.30
	Woodland	5637.52	1725.86	4472.54	1950.74	2452.74	1525.91	2488.00	892.26	314.02	361.81	1150.56	2212.10
	Grassland	112.13	570.61	275.67	202.65	115.53	295.85	76.05	467.99	1529.07	965.81	17.54	510.93
	Waterbody	1442.37	436.00	763.33	216.70	1167.51	448.45	284.54	92.44	231.38	284.54	263.63	86.77
	Unused land	3.57	3.77	30.63	0.36	0.07	0.36	0.22	0.27	62.21	15.25	0.04	0.31
	Total	8498.49	3943.93	7183.77	3653.73	4878.87	4299.91	3241.72	1984.72	2485.19	1836.98	1567.73	3068.41
2015	Farmland	1282.80	1200.66	1644.19	1268.62	1120.73	2014.72	389.97	524.73	345.82	206.98	134.11	256.50
	Woodland	5607.43	1724.52	4469.10	1945.50	2443.03	1524.77	2476.91	891.38	321.47	361.28	1140.20	2209.24
	Grassland	112.56	570.23	273.25	201.97	116.31	295.42	76.97	467.67	1522.77	962.82	19.74	509.26
	Waterbody	1452.30	434.27	758.77	223.98	1161.78	451.85	283.61	95.03	233.72	288.49	258.70	87.45
	Unused land	3.38	3.74	29.75	0.36	0.11	0.37	0.22	0.31	59.96	15.27	0.04	0.34
	Total	8458.47	3933.43	7175.06	3640.43	4841.95	4287.13	3227.68	1979.12	2483.75	1834.84	1552.79	3062.79

in a consistent manner. The ecosystem service value is calculated by the following equations:

$$ESV = \sum_{x,y \times 1}^{a,b} \left(S_x \times C_{xy} \times M\right) \quad (1)$$

$$M = \left(S_{dg} \times M_{dg} + S_{xm} \times M_{xm} + S_{ym} \times M_{ym}\right)/7 \quad (2)$$

In Equation (1), ESV is the total ecosystem service value, S_x refers to the area of xth ecosystem area, C_{xy} is the xth ecosystem'syth service value equivalent coefficient, a is the number of ecosystems, b is the number of ecosystem services, M is one unit standard ecosystem service value equivalent factor, S_{dg}, S_{xm}, $andS_{ym}$ refer to area ratios of Chinese rice, wheat, and corn, respectively, and M_{dg}, M_{xm} and M_{ym} are the market values of the three crops.

The reasons why our study covers the periods of 2000, 2005, 2010, and 2015 are as follows. In the 1990s, due to the introduction of land-use reform and integration of the private sector in the land market, China's urbanization grew at a significant rate. This led to a surge in the built-up area, industrial scale, and volume of revenues of Chinese cities, which was usually

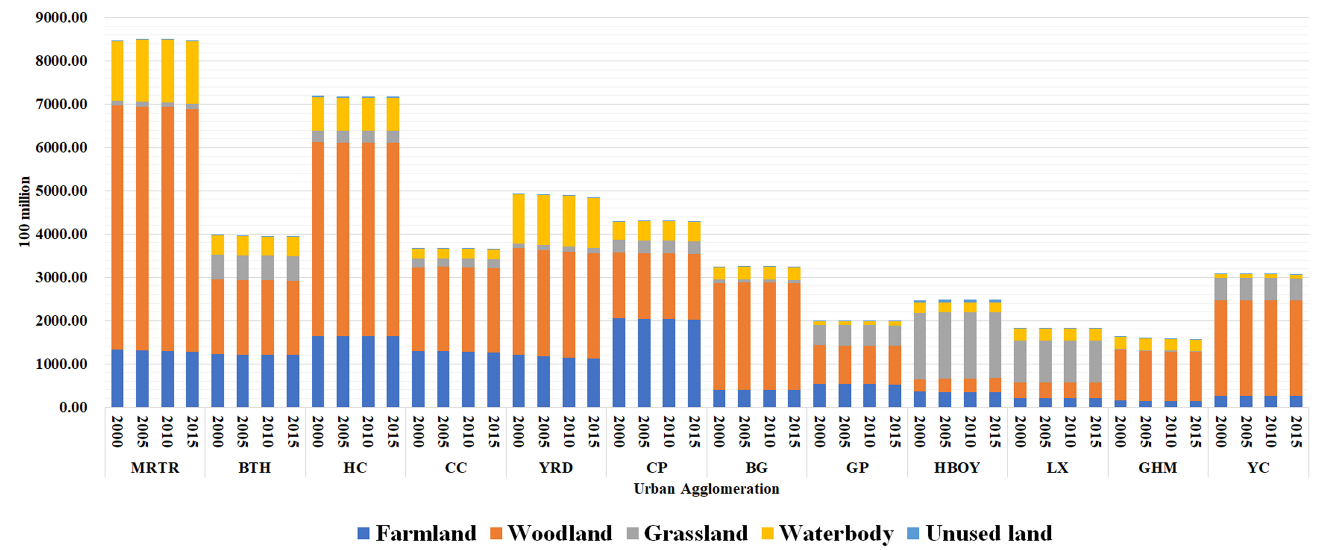

FIGURE 2 | The trend of different ecosystem services in 12 urban agglomerations in China during 2000–2015.

at the cost of cities' natural resources (Gaubatz, 1999). Even though rapid urbanization brings economic development, the country's government realized that the negative impacts on the natural resources and structures of cities were irreversible. Many national-level guidelines were proposed in the early twenty-first century to promote healthy urban development. In the 10th Five-Year Plan proposed in 2000, the central government promoted the coordinating development of large, middle, and small cities as a prototype of urban agglomeration. The 2005 11th Year Plan established integrated urban-rural development, demanding large cities to lead the development of small cities. The 2010 plan stressed the development of urban agglomerations and proposed a "three vertical and two horizontal" pattern of urbanization at the national scale. That being said, during 2000 and 2015, the top-down national policy promoted the stable development of urban agglomerations. Therefore, the study of the state of urban agglomerations in this period can exclude the influence of extreme policies under special circumstances to better reflect the spontaneous evolution characteristics of various urban agglomerations.

In addition, this study calculated the value equivalent factor of ecosystem services in 2000, 2005, 2010, and 2015 based on the China Statistical Yearbook and Compilation of Cost-Benefit Data of Agricultural Products in China, weighting the inflation and market fluctuation in each different year (NBS, 2020), and it used the average values in the 4 years for the standard value equivalent factor of ecosystem service: the value equivalent factor of one standard unit of ecosystem service is 1359.84 CNY per ha. The coefficient of ecosystem service value per unit in China's urban agglomeration is calculated according to Xie et al.'s research (Xie et al., 2003, 2008).

Ordinary Least Squares Model

We used ordinary least squares (OLS) regression to identify the factors that significantly impact ecosystem services within urban agglomerations, and factor screening was conducted based on significance and adjusted R^2. The calculation formula is as follows (Li and Zhao, 2019; Ketema et al., 2020; Zhong et al., 2020):

$$y_i = \beta \sum_{k \times 1}^{n} \beta_k x_{ik} + \varepsilon_i \qquad (3)$$

In Equation 3, y_i is the ecosystem service value at i, β is the interception, β_k is the coefficient of the kth driving factor, x_{ik} is the value of the kth driving factor at i, and ε_i is the error term.

Geographically Weighted Regression

The OLS model is an aspatial regression model. It cannot effectively capture spatial variations in how driving factors impact ecosystem service values when there is of the potential of spatial autocorrelation (Li et al., 2017; Lyu et al., 2019; Huang et al., 2020; Shao et al., 2020).

Therefore, in this study, OLS was performed first to gain the overall associations between driving factors and ecosystem services, followed by spatial regression models, such as geographically weighted regression (GWR), to further explain the local variances (Fotheringham et al., 2002). GWR considers that the spatial variances can be easily visualized to identify the spatial patterns of the relationships, thereby better informing planners and decision makers (Tooke et al., 2010).

$$y_i = \beta_0 (U_i, V_i) + \sum_{k}^{n} \beta_k (U_i, V_i)\, x_k (U_i, V_i) + \varepsilon_i \qquad (4)$$

$$\beta_k (U_i, V_i) = \left(X^T W (U_i, V_i)\, X\right)^{-1} X^T W (U_i, V_i)\, y \qquad (5)$$

$$W_{ij} = exp\left(-d_{ij}^2/h^2\right) \qquad (6)$$

In Equation 4, $\beta_0 (U_i, V_i)$ is the intercept at (U_i, V_i), $\beta_k (U_i, V_i)$ is the coefficient of the kth driving factor at (U_i, V_i), $x_k (U_i, V_i)$

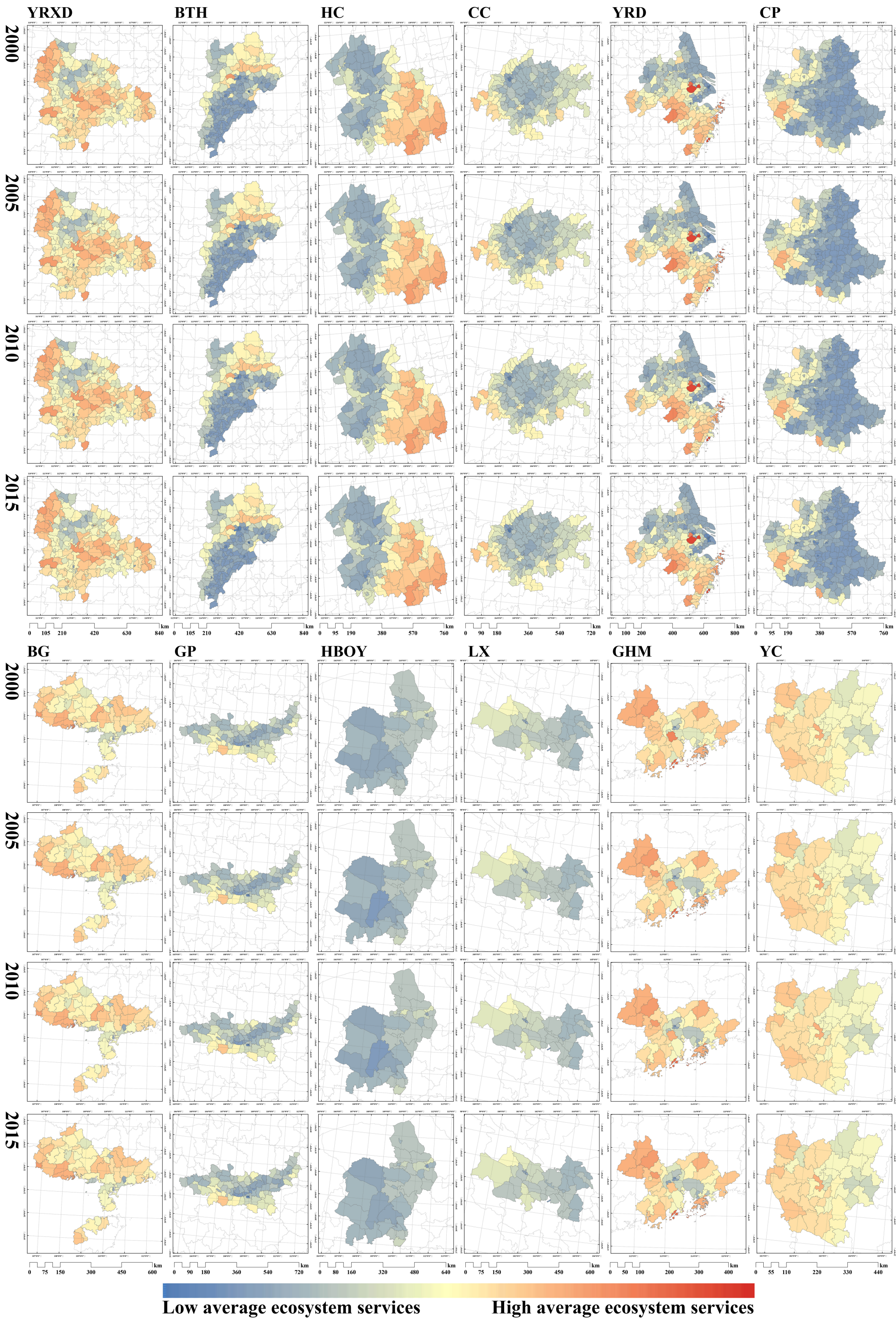

FIGURE 3 | Evolution of ecosystem service value per unit area of cities, counties, and districts in China's existing urban agglomerations from 2000 to 2015 (10,000 CNY/km^2).

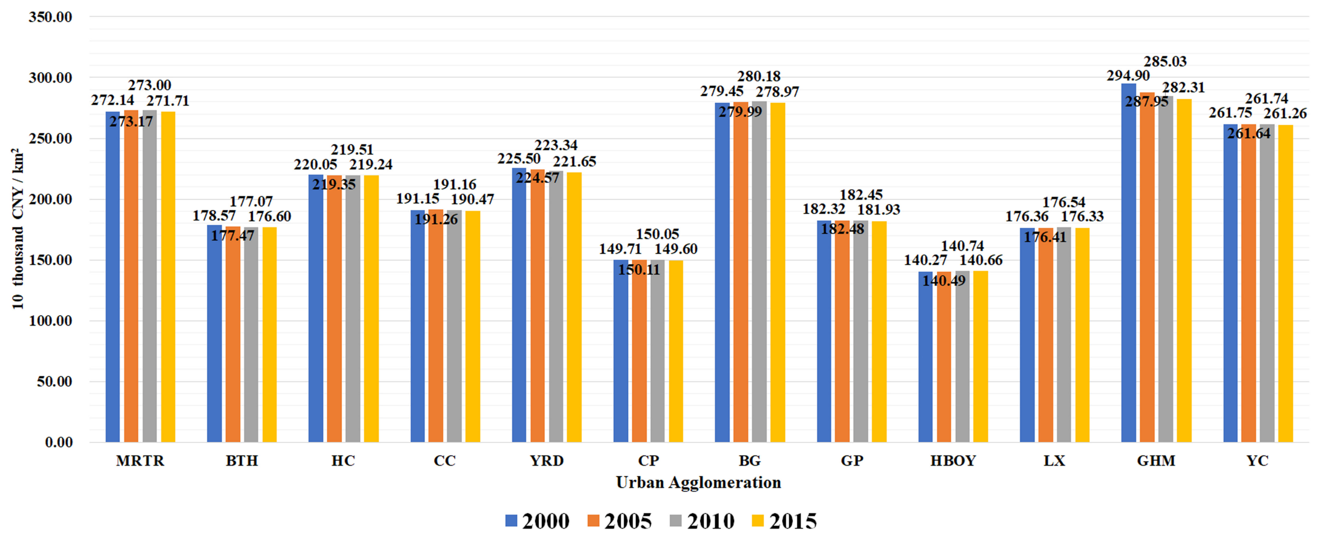

FIGURE 4 | Changes in ecosystem service value per unit area in 12 urban agglomerations during 2000–2015 (10,000 CNY/km^2).

is the value of the kth driving factor at (U_i, V_i), and ε_i is the residual. In Equation 5,X^T is the matrix transpose operation of driving factors, and $W(U_i, V_i)$ is the distance weight matrix. In Equation 6, h is the bandwidth of AIC, and d_{ij} is the distance between i and j.

The OLS and GWR were performed in the ArcGIS 10.6 platform.

RESULTS AND DISCUSSION

Ecosystem Service Values in Different Urban Agglomerations

Based on **Tables 3,4** the values of ecosystem services in the 12 urban agglomerations (at the county-level administration) in 2000, 2005, 2010, and 2015 can be found in **Figure 2**.

With regard to the total value of ecosystem services, urban agglomerations with larger areas tend to have higher ecosystem service values. In most cases, the total value is highest in the MRTR, and the lowest value is in the GHM (see **Figure 2**).

With regard to the ratio of ecosystem services' values, cropland, woodland, and grassland are the three dominant components of ecosystem services values, and their ratios vary in different urban agglomerations. The MRTR, HC, CC, YRD, BG, GHM, and YC urban agglomerations have woodland as the primary provider of their ecosystem services. The ratios of ecosystem services from woodland ranged from 49.60 to 76.95%. Ecosystem services in BTH and GP are also mainly provided by woodland ecosystems, and the ratios ranged from 43.36 to 44.99%. Cropland and grassland ecosystem services occupy approximately 14.34–30.77% of the total ecosystem services in these areas. Cropland ecosystems are the major providers of ecosystem services in the CP urban agglomeration, with ratios ranging from 46.96 to 47.87%. In HBOY and LX, the ecosystem service values were largely derived from grassland, with ratios ranging between 52.47 and 62.45%.

The total values of ecosystem services significantly decreased in all 12 urban agglomerations between 2000 and 2015. In addition to a small increase observed in HBOY, the total values of ecosystem services in the remaining urban agglomerations declined by values ranging from 0.02 to 4.27%.

A few urban agglomerations increased in ecosystem service value based on certain land-use types, while the majority of ecosystem service values from different land uses were reduced in most urban agglomerations. BTH experienced reductions in all types of ecosystem service values, among which the reduction rates ranged from 0.13 to 5.40%. The MRTR and CP urban agglomerations had increased ecosystem services related to waterbodies and decreased ecosystem services derived from other land-use types, with reduction rates ranging from 0.05 to 15.88%. HBOY raised the ecosystem service of forestland, and GHM had their ecosystem service of grassland increased, as the rest of the ecosystem services from other land uses were reduced with rates ranging from 0.29 to 13.19%. HC increased the ecosystem services from farmland and grassland, BG increased the ecosystem services derived from waterbodies and forestland, YC increased the ecosystem services from waterbodies and unused land, and the ecosystem services from the other land-use types in these areas declined by 0.03 to 6.30%. Ecosystem services derived from farmland and grassland in CC and LX and farmland and forestland in YRD decreased by 0.28 to 7.06%. In GP, the major reduction came from ecosystem services derived from farmland, with a reduction ratio of 2.89%.

Considering the widely ranging areas of different county units, we calculated the ecosystem service values per unit area to compare the different urban agglomerations (see **Figures 3**, **4**).

Ecosystem service values per unit area ranged significantly, from the lowest 140.49 in HBOY (2005) to the highest 294.90 in GHM (2000).

Heterogeneous spatial patterns of ecosystem service values per unit area were identified across the 12 urban agglomerations. In the MRTR, BG, GHM, and YC, the values of ecosystem services were low in the central core and gradually increased with distance from the core area. The BTH, HC, YRD, and CP agglomerations also had a core city with low ecosystem service values intruding peripheral forestland and grassland where the values were high. The CC, GP, HBOY, and LX presented lower ecosystem service values in general, with only a few exceptions at the outskirts of cities within the urban agglomerations.

Apart from the heterogeneity of spatial patterns of ecosystem services values within individual urban agglomerations, the difference in ecosystem services in different urban agglomerations prevailed. Most urban agglomerations in the northeastern coastal area of China showed higher values than others. Interestingly, our results indicated that more developed urban agglomerations tended to have higher ecosystem service values. This result provides empirical evidence that urban development and the integrity of ecosystem services are not mutually exclusive.

As cities continue to expand their footprints, more than 70% of areas within each urban agglomeration reduced their ecosystem service values from 2000 to 2015, and the reduction ratio was continually increasing. Taking GHM as an example, the results in **Figure 4** show that GHM has a significant decreasing trend of ecosystem services (294.90–282.31, -4.27%). According to the development status of GHM in **Table 1**, changes in built-up area, changes in forest area and special location seem to be related to the trend of decreasing ecosystem services. At the same time, according to the results of **Figure 3**, this decrease does not only occur around the core areas of urban agglomeration. So we assume that natural and social conditions might contribute to the heterogeneity of ecosystem services within urban agglomerations. The following contents try to identify such impacts.

Driving Factors on Ecosystem Services in Different Urban Agglomerations

Identifying Driving Factors

Although all 12 urban agglomerations experienced declines in ecosystem services to some extent, it is unknown what driving factors were responsible and how they affected such declines. Following previous studies (Liu et al., 2018; Gao and Wang, 2019; Lyu et al., 2019; Sun et al., 2019; Chen J. et al., 2020; Dai et al., 2020; Luo et al., 2020), we aim to adopt eight typical driving factors (see **Table 5**) related to the natural environment and social conditions that might potentially impact the decline in the ecosystem services of the case study urban agglomerations.

Table 6 presents the results of OLS and GWR. The OLS, as an aspatial model, returns only a global coefficient that cannot reflect spatial variances compared with GWR. The adjusted R^2 in OLS ranged from 0.0197 to 0.5285, while the values of GWR ranged from 0.3275 to 0.7478. The adjusted R^2 in each GWR model was higher than that in OLS. The AIC value in GWR was lower than that in OLS by more than 3%, suggesting that the GWR can provide better explanatory power for the driving factors.

Measuring the Strength of Driving Factors

The normalized difference vegetation index (NDVI) and temperature factors had lower explanatory power (adjusted $R^2 < 0.5$), indicating that their impacts on ecosystem service values were minimal; thus, they were excluded from our mapping visualization. To identify the strength of the six factors' impacts on ecosystem service values, their regression coefficients were visualized into ten categories.

Figure 5 shows that elevation has positive impacts on ecosystem services except in the northeastern parts of HC, HBOY, and southern parts of YC. The impact gradually decreases from the central city to the periphery in each urban agglomeration. Coastal urban agglomerations such as the YRD, GHM, and BG tend to be more influenced by elevation than are hinterland agglomerations.

The main reason for the different impacts of elevation on urban agglomeration is subject to the topography of China, where it is higher in the northwest than in the southwest. The HC, HBOY, and YC are affected by the Inner Mongolian Plateau, Loess Plateau, and Yunnan-Guizhou Plateau, respectively. The higher elevations often bring about negative impacts on ecosystem environments. For example, the HBOY has a low-west, high-east topography. As elevation rises, the climate type transitions from continental climate and monsoon climate of medium latitudes to plateau mountain climate, which has low temperature, less precipitation, and intense solar radiation, thus limiting the ecosystem (Wu et al., 2021). In the area where the monsoon climate is dominant, low elevation coastal regions, such as the YRD, GHM, and BG, a higher elevation often means there are more vegetated hilly areas than agricultural lands—the former present higher biodiversity and have more potential for ecosystem services (Bai et al., 2020). An example case is the YRD, where the flat plain area is largely built up, and the ecological protection area is constrained to shallow mountain areas that are hard to develop (Song et al., 2019).

Figure 6 maps how the GDP impacts the ecosystem service values in 12 urban agglomerations. As seen, in most areas (except HBOY and northwestern HC), the higher GDP leads to lower ecosystem service values. The strength of the impact gradually increases from the central city to the periphery in each urban agglomeration. The HBOY, LX, and parts of HC tend to be more influenced by GDP.

The industrial structure of an urban agglomeration might be attributable to the heterogeneous impacts of GDP. The YRD, BTH, and MRTR were established earlier than the others, and their GDP is gradually derived from the less-polluted tier-three industry with high technology and service industry rather than from tier two, i.e., the manufacturing industry. The former is known to be less threatening to the environment. For example, the structure of GDP in BTH relies more on high-tech and service industries than on manufacturing industries; the former demands less land space and poses less stress on the environment (Li et al., 2015). In contrast, in hinterland China, urban agglomerations such as LX were transferred from tier one industry to tier two industry. Their GDP relies more on manufacturing and mining industries and subsequently increases the burdens on ecosystems through excessive mining activity and heavy pollution (An et al., 2008; Ma et al., 2019). In addition, pollution-generating, tier two industries are often located at the outskirt of an urban agglomeration, and thus, the ecosystem service values in peripheral areas are more reduced than those in the central city.

TABLE 5 | Statistical analysis table of driving factors.

Driving factors		Year	Average	Standard deviation	Maximum	Minimum
Environmental factors	Average elevation (m)	——	418.807559	595.697676	3461.855319	0.063944
	Coverage of vegetation (100%)	2000	0.662530	0.123087	0.836652	0.160600
		2005	0.698413	0.132794	0.887718	0.178284
		2010	0.714130	0.126745	0.889560	0.202600
		2015	0.700660	0.140898	0.891590	0.188677
	Precipitation (mm)	2000	992.085215	466.664943	2615.776297	158.226902
		2005	994.355593	449.619113	2251.895570	142.636587
		2010	1063.461032	565.288225	2766.267724	207.604931
		2015	1039.357594	548.195150	2487.480941	174.768644
	Humidity (°C)	2000	14.536538	4.861548	25.171434	-0.016768
		2005	14.599246	4.856380	25.423499	0.686400
		2010	14.671047	4.977356	25.977852	1.141891
		2015	14.805899	4.532835	26.827715	0.936345
Socioeconomic factors	GDP per unit area (CNY/km^2)	2000	1282.900441	3152.772766	43175.223048	0.000000
		2005	1984.744113	3444.982596	27792.154927	5.613884
		2010	2657.348676	3695.846950	20641.479981	13.046842
		2015	11938.836772	49082.815867	787543.612903	0.000000
	Population density (People/km^2)	2000	953.655553	3147.580472	57777.350000	3.737918
		2005	1014.365732	2890.883727	42620.750000	3.000341
		2010	1034.919829	2877.015091	42308.450000	2.853306
		2015	1135.088516	2960.406495	29682.600000	3.711563
	Proportion of built-up area (100%)	2000	0.044777	0.123918	1.000000	0.000000
		2005	0.053745	0.138622	1.000000	0.000000
		2010	0.065113	0.154756	1.000000	0.000000
		2015	0.083257	0.175653	1.000000	0.000000
	Proportion of woodland (100%)	2000	0.213612	0.251788	0.944339	0.000000
		2005	0.210868	0.251628	0.944339	0.000000
		2010	0.208662	0.251742	0.943922	0.000000
		2015	0.207522	0.251927	0.941836	0.000000

TABLE 6 | Comparison between the geographically weighted regression (GWR) and ordinary least squares (OLS).

Driving factors	Models	Adjusted R^2				AIC			
		2000	2005	2010	2015	2000	2005	2010	2015
Elevation	OLS	0.0190	0.0215	0.0215	0.0250	16576.2373	16582.9623	16582.9623	16582.3412
	GWR	0.6894	0.6853	0.6853	0.6829	15006.8139	15035.7046	15035.7046	15050.9673
GDP	OLS	0.0508	0.0889	0.1048	0.0581	16529.3141	16481.4360	16456.3492	16533.2002
	GWR	0.6920	0.6981	0.7074	0.6519	15006.0974	14988.5342	14946.2138	15192.1352
Population density	OLS	0.0657	0.0841	0.0865	0.0996	16506.8432	16488.9659	16485.2619	16469.0271
	GWR	0.6744	0.6794	0.6824	0.6820	15083.4499	15072.7489	15059.4871	15066.0071
NDVI	OLS	0.0430	0.0521	0.0795	0.1242	16540.9650	16537.8108	16496.0905	16429.6354
	GWR	0.4464	0.4548	0.5213	0.5528	15771.2807	15760.3548	15579.3839	15487.7625
Precipitation	OLS	0.2421	0.2715	0.3132	0.2170	16209.0328	16163.2251	16079.4056	16270.2705
	GWR	0.5451	0.5284	0.5360	0.4478	15507.9856	15569.9828	15543.3392	15787.8257
Humidity	OLS	0.0403	0.0418	0.0442	0.0233	16545.0013	16553.1767	16549.6820	16584.7197
	GWR	0.3439	0.3458	0.3379	0.4078	16008.2757	16014.9083	16031.6696	15880.5046
Proportion of built-up area	OLS	0.0709	0.0817	0.0749	0.0775	16498.8402	16492.6724	16503.1159	16503.5412
	GWR	0.6708	0.6732	0.6729	0.6691	15096.9217	15097.0282	15098.6614	15120.2418
Proportion of woodland	OLS	0.5172	0.5199	0.5215	0.5365	15567.3241	15569.8400	15564.9215	15524.1213
	GWR	0.7490	0.7482	0.7513	0.7526	14721.4687	14736.6022	14718.8424	14716.0549

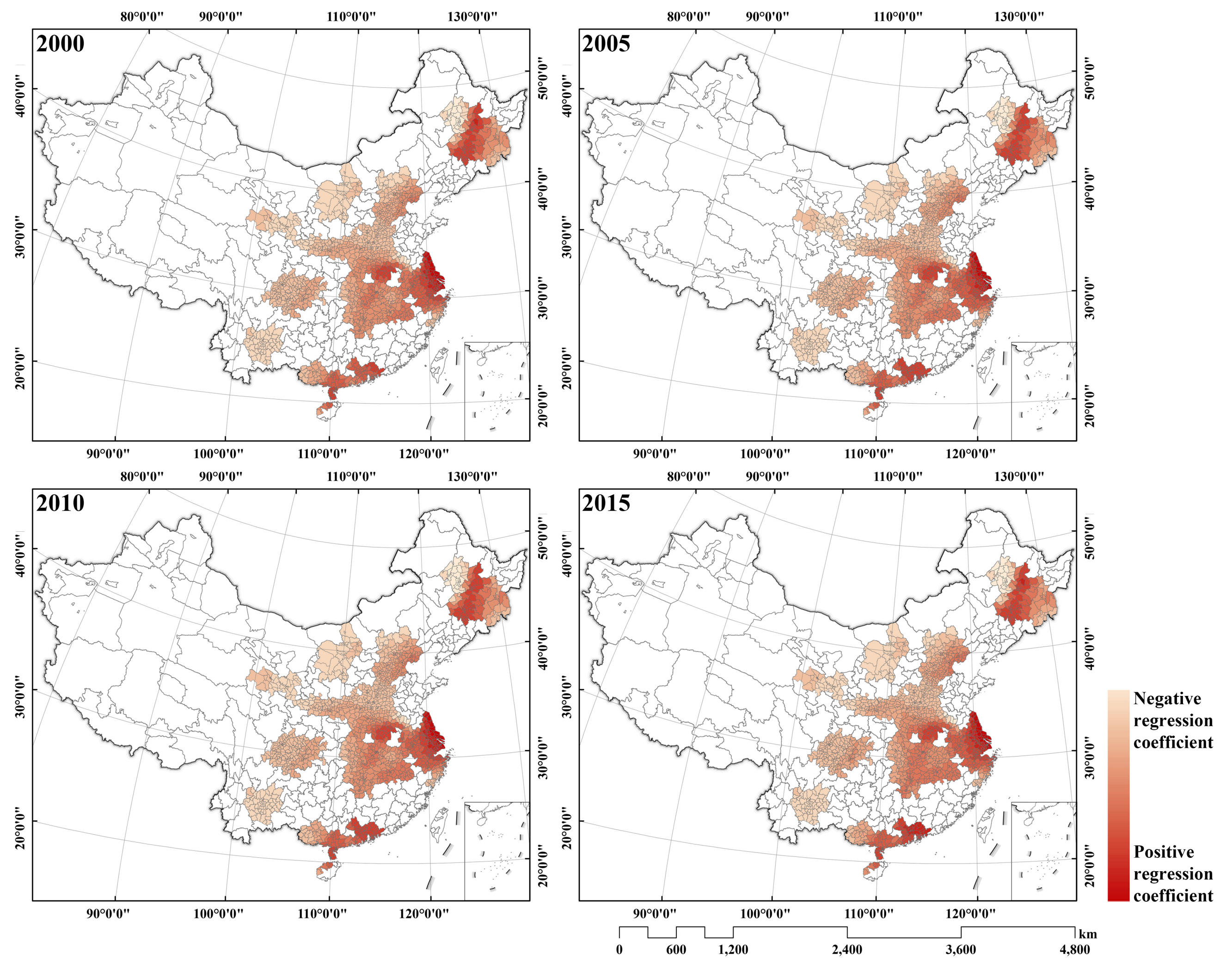

FIGURE 5 | Spatial distributions of the DEM coefficients of GWR models in typical urban agglomerations in China.

In **Figure 7**, the population factor negatively impacts the ecosystem service values in the 12 urban agglomerations except in HBOY. The strength of the impact gradually declines from the central city to the periphery in each urban agglomeration. Regarding the spatial heterogeneity across the country, the GHM, HC, and BG are more negatively impacted by population than the others, while the HBOY is positively impacted more than the other agglomerations.

Higher population density is related to a higher level of human activity, such as agricultural production, industrial construction, and recreational activities, which negatively impacts ecosystem services. An example case is the GHM. It has multiple high-density clusters, such as Guangzhou, Shenzhen, Hong Kong, and Macau, where the environmental pollution caused by household garbage and the service industry has a very serious impact on the ecosystem service values of the peripheral areas of the core-level cities (Bi et al., 2020). Within each urban agglomeration, a higher population density also yields negative impacts on the ecosystem service values. For example, the high-density urban core and low-density mountainous areas in the periphery have distinct differences in ecosystem services in the BTH (Xie et al., 2017), where the former has a low ecosystem services value, and the latter has higher values.

As shown in **Figure 8**, precipitation can positively affect ecosystem services except in the YC, BTH, HBOY, and GHM. The strength of the impact gradually increases from the central city to the periphery in each urban agglomeration. Across the nation, northern and southwestern agglomerations (e.g., HC and CC are statistically significant) showed a positive association, while the central agglomerations (e.g., YC is statistically significant) showed a negative sign.

The nationwide heterogeneous impacts of precipitation on ecosystem services are because the central urban agglomerations are located in warm temperate deciduous broad-leaved forest or subtropical evergreen broad-leaved forest zones with higher biodiversity, and the basin terrain contains more water, which subsequently contributes to higher ecosystem service values. However, subject to precipitation intensity and geographical

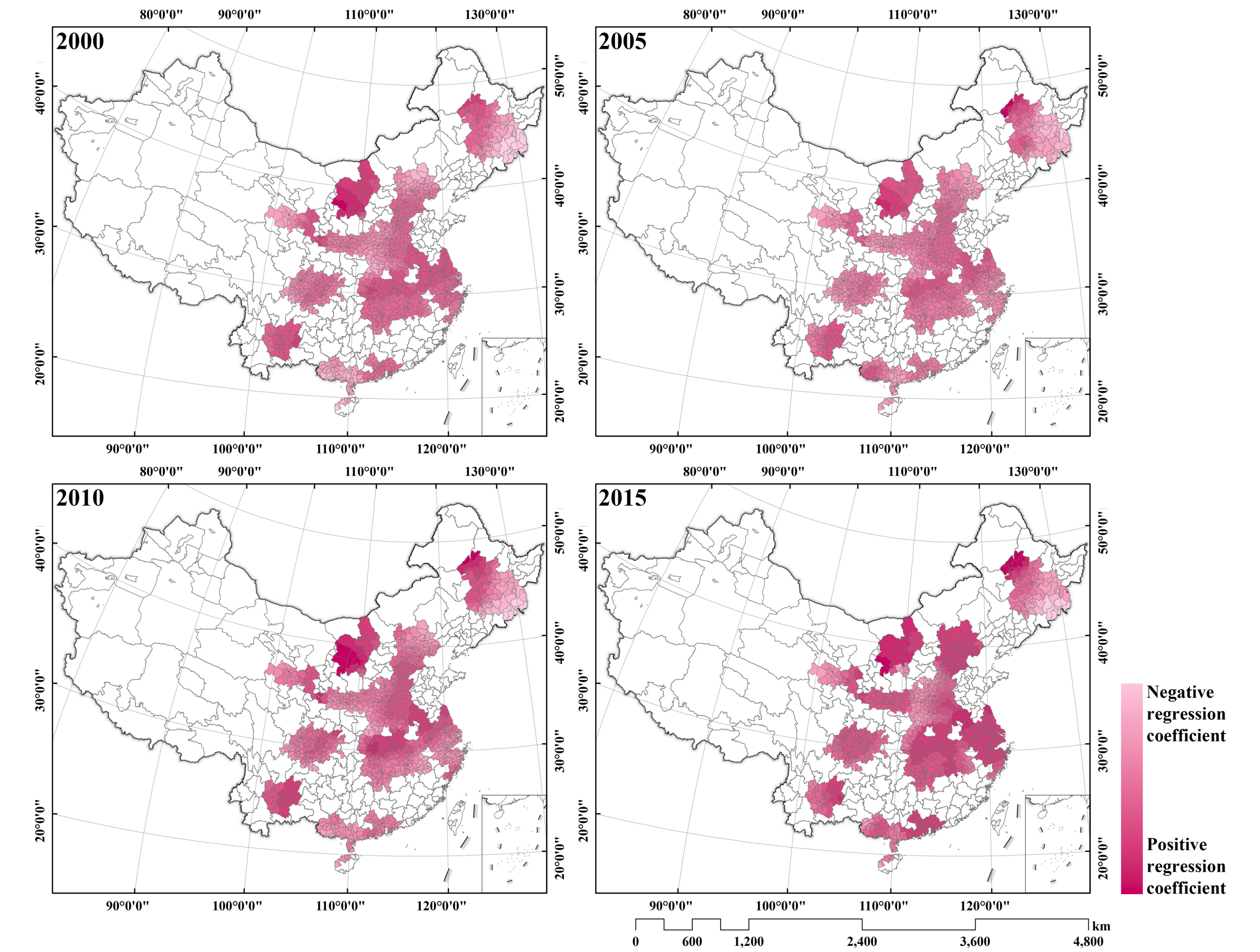

FIGURE 6 | Spatial distributions of the GDP coefficients of GWR models in typical urban agglomerations in China.

conditions, other urban agglomerations with higher annual precipitation might cause flooding issues (Lan et al., 2004). For example, the YC often suffers from concentrated precipitation in a short period of time and leads to frequent flood disasters, which further increase the chances of ecological disasters such as landslides that damage overall ecosystem service values.

In **Figure 9**, the ratio of constructed land use can negatively impact ecosystem services with a few exceptions in the western areas of HBOY. The strength of the impact gradually increases from the central city to the periphery in each urban agglomeration. There is little regional disparity regarding the impacts of construction land use across the country.

The discrepancy in the impacts of built-up areas on ecosystem services across different urban agglomerations is related to the varying levels of built-up area expansion into nearby natural environments, thus causing different levels of disturbances and pressures. Except for Beijing, Tianjing, and Shanghai in the BTH, MRTR, and YRD, most cities in the urban agglomerations in China are at an infant stage of development, and their urban development footprints are expanding. Such expansion is prevalent at the outskirts of cities and thereby intrudes and disturbs existing green-blue spaces such as farmland, grassland, woodland, and waterbodies around the core cities within individual urban agglomerations (Wang Z. et al., 2019). At the national scale, the difference in built-up areas' effects on ecosystem services is due to the governments' urban planning and industrial guidance. Most decision makers prioritize areas for urban and industrial development over spatial connectivity between the natural environment and urban development, which leads to an encroachment on ecological space, the destruction of ecological structure at the urban fringe, and reductions in biological habitats.

In **Figure 10**, the ratio of forestland yields a positive impact on ecosystem service values in all 12 urban agglomerations. The strength of the impact gradually declines from the central city to the periphery in each urban agglomeration. From Northwest to Southeast China, the impacts of the forestland use ratio gradually increased. The BTH, CP, and YRD in south-coastal and eastern China show higher impacts of the forestland use ratio.

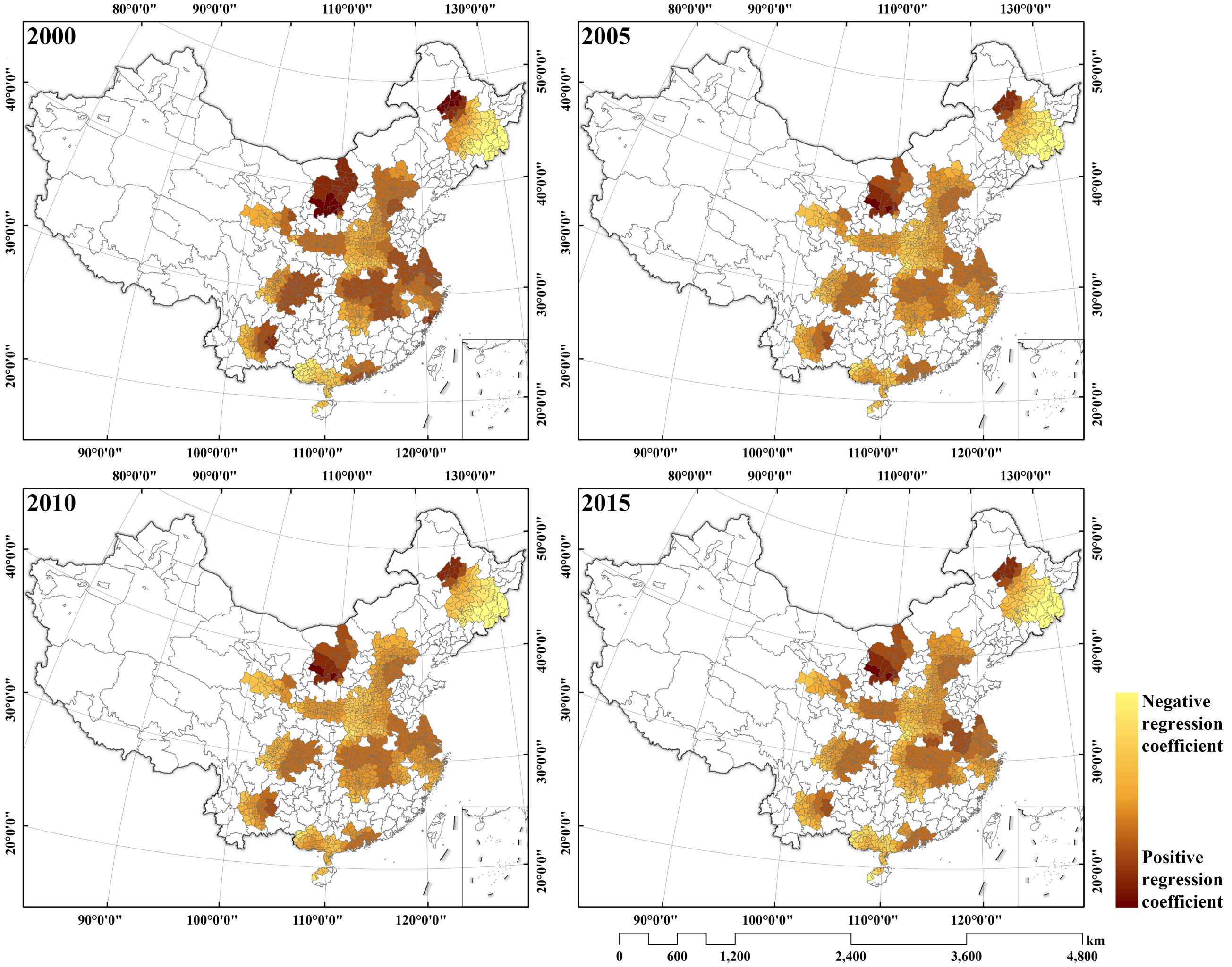

FIGURE 7 | Spatial distributions of the population coefficients of GWR models in typical urban agglomerations in China.

Forest ecosystems often have the highest biodiversity with the most complete community structure, and they are considered one of the most complete ecosystems. From the northwest to the east, the forest structures in China gradually change from temperate grassland to subtropical evergreen broad-leaved forest, their forest biodiversity and community structure complexity are increasing, and subsequent outputs of ecosystem services are raised. Within each urban agglomeration, forestland in mountainous areas is a major provider of ecosystem services (Fujii et al., 2017). For example, by regulating the micro climate and conserving water and soil, Xishan Mountain in western BTH improved ecosystem quality in the northwestern part of the urban agglomeration (Wu et al., 2015).

Policy Recommendations

Elevation, precipitation, and fraction of woodland have positive impacts on ecosystem service values on urban agglomerations, while GDP, population, and proportion of built-up area negatively affect ecosystem service values. The impacts of elevation and fraction of woodland are gradually increasing, as the effects of GDP, population, and proportion of built-up area are declining and transitioned to positive in the most recent year. The effects of precipitation vary in each year.

As seen in **Table 7**, the six types of urban agglomeration are based on how the ecosystem services are impacted by environmental and socioeconomic factors: natural-factor dominated, socioeconomic-factor dominated, policy dominated, balanced, natural-factor inclined, and socioeconomic-factor inclined.

Urban agglomerations with ecosystem services largely driven by natural factors often present fully developed urban patterns. They consist of large cities with stabilized industrial structures and fully developed urban structures. These cities' industrial structures tend not to be significantly restructured during urban development, where minor adjustments are more common in urban agglomeration planning. In other words, socioeconomic factors do not play a major role in affecting ecosystem services. Instead, the changes in natural factors such as precipitation and temperature associated with geographical conditions impact ecosystem services more. Taking YRD as an example, its internal

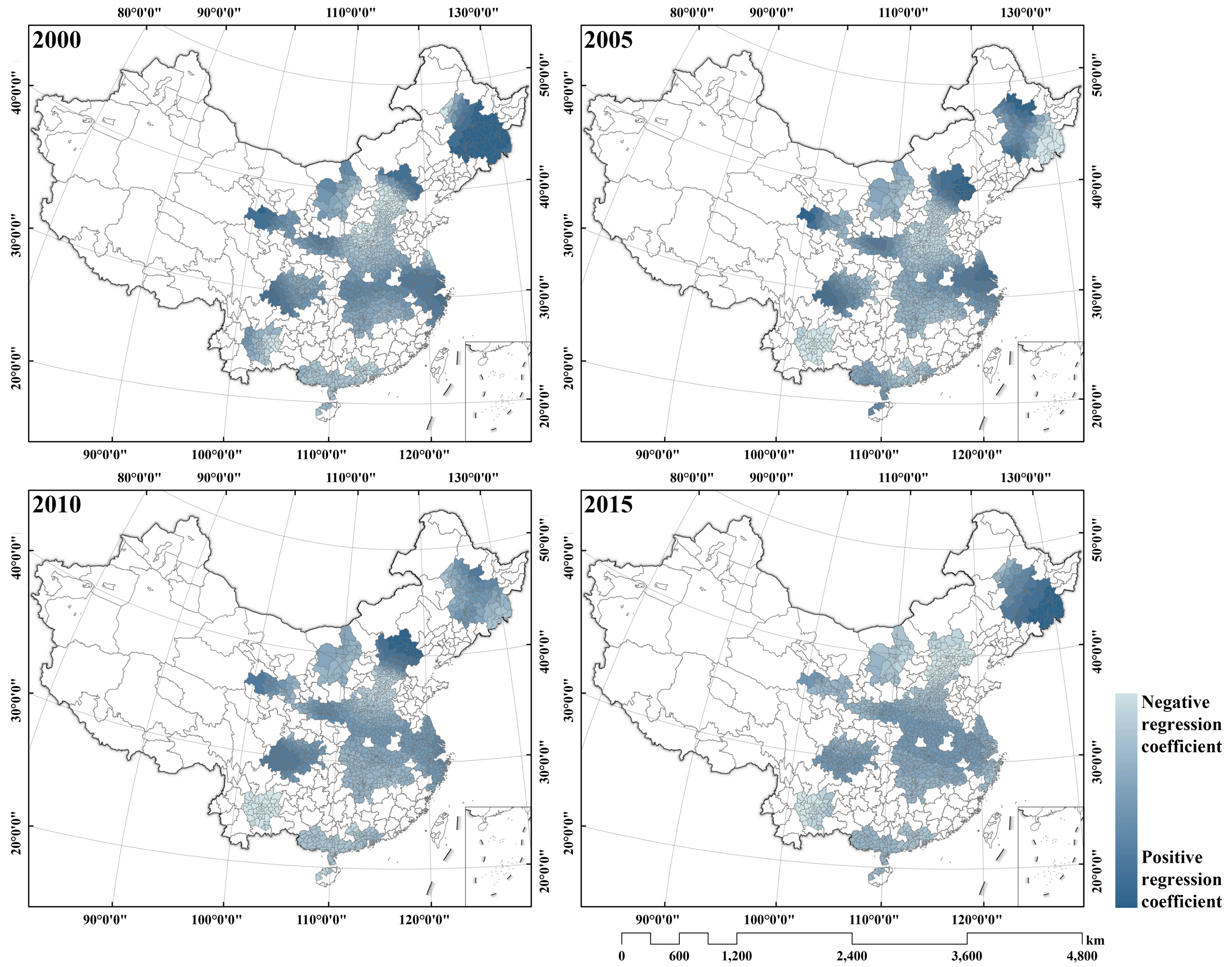

FIGURE 8 | Spatial distributions of the Rain coefficients of the GWR models in typical urban agglomerations in China.

industrial structure and population scale tend to be stable with the core city Shanghai, including Nanjing, Hangzhou, Hefei and other first-tier cities. Its special topography located in the middle and lower reaches of the Yangtze River flood plain has a stronger influence on the development of its urban agglomeration. Therefore, YRD should both make full use of the plain area space within the border and consider strengthening the existing urban space and industrial transformation that could pose less burden on natural conditions.

Urban agglomerations driven largely by socioeconomic factors are still transitioning period to fully developed ones. These cases are mostly led by heavy industries (e.g., manufacturing and energy), while their natural conditions are minimally changed. Urban expansion to accommodate multiple industries is a core cause of ecosystem changes. For example, HBOY's natural environment is homogenous, but the social, industrial and economic development of each city and county within the urban agglomeration are highly differentiated. This is because HBOY is located in northwest China, where the core cities, including Hohhot, Baotou, Ordos, and Yulin, heavily rely on secondary industries. Therefore, in the medium and long-term construction process, HBOY should reasonably plan the urban space and industrial development intensity on the basis of taking the best use of existing economic and industrial models while ensuring less damage to the current ecosystem.

Those urban agglomerations impacted by both natural and socioeconomic factors are increasing and are major focuses of urban agglomeration construction in China to balance the development across different regions in China (e.g., GHM, CC, HC, and GP). These urban agglomerations have well-developed core cities with respect to urban structure and industrial patterns, and the latter also have strong locational characteristics such as major functions (e.g., Hong Kong in the GHM is a financial center, Chengdu in the HC is a logistic hub, Harbin in the HC is a manufacturing base, and Xi'an in the GP is a leading

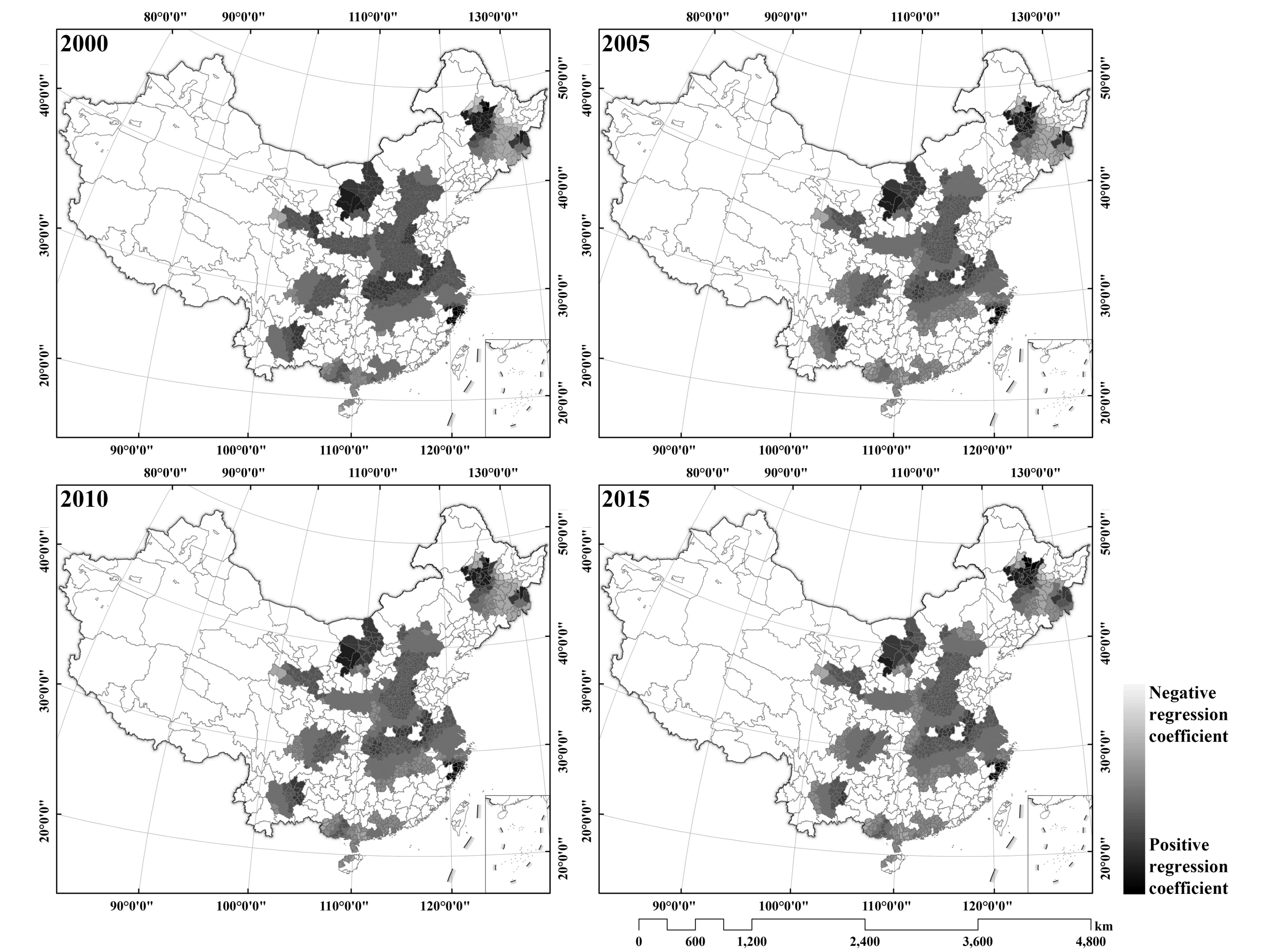

FIGURE 9 | Spatial distributions of the city-area coefficients of the GWR models in typical urban agglomerations in China.

tourism area). Meanwhile, the natural conditions vary widely within these urban agglomerations, which affect the ecosystem services within each. Development in these areas should balance the major goals of economic development and curb potential ecological crises. Ecosystem compensation mechanisms should be established by carefully utilizing internal ecological resources, adjusting industrial structures, and controlling urban expansion and economic development at a reasonable pace. Taking CC as an example, the GNP of its core cities, Chengdu and Chongqing, far exceeds that of the non-core cities within the urban agglomeration. Meanwhile, Chengdu, and Chongqing are located in mountainous areas that call for different development from those urban agglomerations in the plains. CC is recommended to consider developing third-tier industry in the inner core cities to promote the efficient economic development of current land use. Those non-core cities should stress expanding urban land use carefully and more efficiently to reach a harmonious development of the city and nature.

Despite being dominated by natural factors, some urban agglomerations also appear to be impacted by one or two socioeconomic drivers. These cases are often led by relatively developed core cities, but peripheral cities are less developed with regard to industrial structure and development modes. Decision makers should highlight the advantages of natural conditions such as geographical bases and climatic characteristics to protect existing natural resources and consider reasonable development of non-core cities with proper industrial structure and population planning (e.g., MRTR, BTH, CP). Taking BTH as an example, the urban agglomeration is surrounded by the Xishan mountains and Bohai Bay; meanwhile, the congregation effect of the core cities of Beijing and Tianjin is significant. However, the peripheral 13 cities in Hebei province are uneven regarding the economic development levels (Li et al., 2021). Thus, the subsequent development should first highlight the role of the core cities of Beijing and Tianjin in terms of natural environment protection. The strong economic power of the core cities can provide complementary revenues for neutralizing the environmental damage due to less developed peripheral cities in Hebei province.

Some urban agglomerations are impacted mostly by socioeconomic drivers and one or two natural factors at the

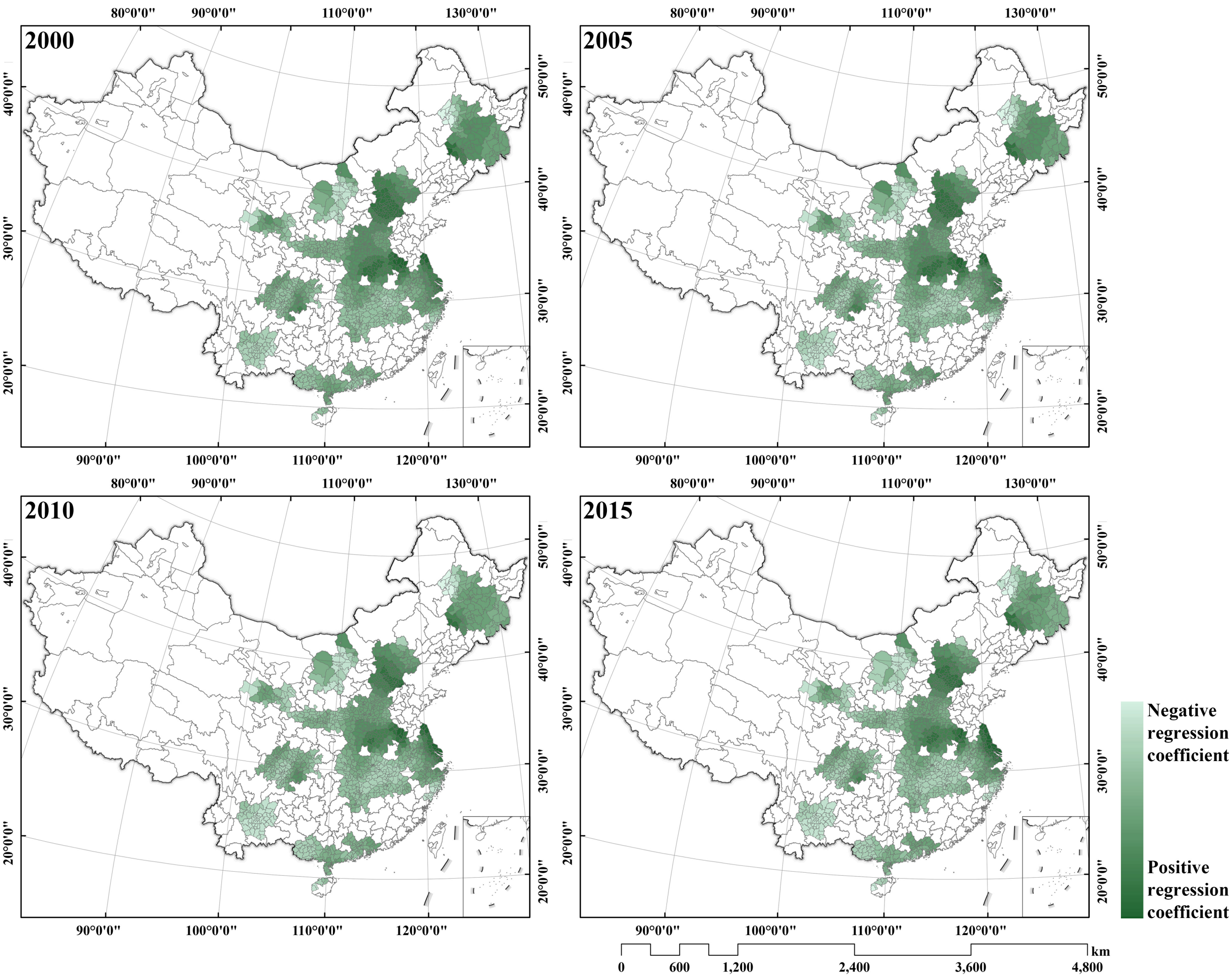

FIGURE 10 | Spatial distributions of the forest-area coefficients of the GWR models in typical urban agglomerations in China.

same time (e.g., BG). Their core cities are developing in a single mode, making them highly impacted by socioeconomic factors. In addition, geographical conditions, such as widely varying elevations and climate features, foster higher biodiversity and subsequently affect the internal ecosystem service values. These urban agglomerations should address the industrial characteristics and structures of core and node cities by coordinating the ecosystem and land-use resources and mitigating the negative impacts of industrial development. For instance, the BG urban agglomeration is underdeveloped, and the core city of Nanning is still in the process of urban expansion. However, the BG is located among complicated geographical conditions where the wide range of elevations and rich biodiversity are strongly sensitive to urban expansion. Therefore, the priority of the Beibu Gulf urban agglomeration is to arrive at a rational urban plan to highlight the preservation of the ecosystem and carefully select regional locations for intensive economic development.

Some urban agglomerations were in the early stage of development with less intensive construction activities when the policy was proposed. This type of urban agglomeration is more policy-oriented, in which the differentiation degree between core city and node city is not obvious. The characteristics of socioeconomic and industrial development structure are unclear, and the urban land use agglomeration degree is not strong due to the influence of national macro-strategic layout on the ecological service function of urban agglomeration. In YC, for example, Lanzhou and Xining cities have a similar development stage. In addition, due to the limited traffic condition, the cities are less connected, showing less identifiable urban agglomeration structures. Therefore, YC should first consider how to strengthen connections between cities, in which the government should take the lead while considering how to strengthen control of the natural environment and living environment simultaneously.

CONCLUSION

With the growing level of urbanization, urban agglomerations are becoming a major form of urban development and should

TABLE 7 | Summary of driving factors on ecosystem services in each urban agglomeration.

NAME	Environmental factors			Socioeconomic factors		
	Elevation	Precipitation	Proportion of woodland	GDP per unit area	Population density	Proportion of built-up area
MRTR	++(+)	++(−)	++(+)	−(−)	−(+)	−(−)
BTH	++(+)	++(+)	+++(+)	−(−)	−(+)	−(−)
HC	+++(+)	+++(+)	++(+)	−(−)	−(−)	−(−)
CC	++(+)	+++(−)	++(+)	−(−)	−(+)	−(−)
YRD	+++(+)	+++(−)	+++(+)	−(−)	−(+)	−(+)
CP	++(+)	+(+)	+++(−)	−(−)	−(−)	−(−)
BG	+++(+)	+(+)	+(+)	−(−)	−(−)	−(−)
GP	+(+)	++(−)	++(+)	−(−)	−(−)	−(−)
HBOY	+(−)	+(−)	+(+)	+++(−)	+++(−)	+(−)
LX	+(+)	+++(−)	+(+)	−(−)	−(−)	−(−)
GHM	+++(+)	+(−)	+++(+)	−(−)	−(−)	−(−)
YC	−(+)	−(+)	+(+)	−(−)	−(+)	−(−)

The icon outside of the bracket refers to the strength of driving factors, +++ refers to a high positive impact, ++ refers to a moderate positive impact, and + refers to a weak positive impact; − refers to a high negative impact, − refers to a moderate negative impact, and—refers to a weak negative impact. The icons in the bracket refer to the trend of impact level during the study period. (+) indicates a growing strength, and (−) indicates a declining strength.

be considered in national-level spatial planning. Understanding the spatiotemporal dynamics of ecosystem services and the driving factors is necessary to provide decision makers with reference information for promoting explicit location strategies and guidelines for urban development. Using multiple data sources, this study quantified the values of ecosystem services in 12 typical urban agglomerations in China from 2000 to 2015. Furthermore, the potential driving factors are identified through GWR and OLS.

(1) The ecosystem service values were heterogeneous among individual urban agglomerations due to the internal variances in natural conditions and urbanization patterns. Spatial discrepancies also existed across different urban agglomerations in China because of the different locations and contexts. Most southern coastal urban agglomerations had higher ecosystem service values than those in the hinterland. From 2000 to 2015, the overall ecosystem service values declined in more than 70% of urban agglomerations, with reduction rates ranging from 0.02 to 4.27%. Such reductions were more common in the central areas of urban agglomerations.

(2) Elevation, precipitation, and fraction of woodland had positive impacts on ecosystem service values in urban agglomerations, while GDP, population, and proportion of built-up area had negative effects. The impacts of elevation and fraction of woodland were gradually increasing, while the effects of GDP, population, and proportion of built-up area were declining and transitioned to positive in the most recent year. The effects of precipitation varied each year.

(3) The driving factors impacted the ecosystem services of different urban agglomerations in different ways. Although the MRTR, BTH, and CP were impacted mostly by natural factors, one or two socioeconomic factors influenced the ecosystem service values. The ecosystem service values in the HBOY and BG were largely driven by socioeconomic factors, but the latter was also driven by elevation. The GHM, CC, HC, and GP had ecosystem services affected by both socioeconomic and natural factors. In the LX and YC, the development of ecosystem services was largely affected by policies, with minor impacts from socioeconomic and natural factors.

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

MS and CL: conceptualization, methodology, writing—review, and editing. CL: data curation and funding acquisition. MS: formal analysis, software, and writing—original draft. CL and FL: project administration and supervision. MS, LW, and CL: validation. MS and LW: visualization. All authors have read and agreed to the published version of the manuscript.

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Connecting Biodiversity With Mental Health and Wellbeing — A Review of Methods and Disciplinary Perspectives

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Biodiversity conservation and mental health and wellbeing are of increasing global concern, with growing relevance to planning and policy. A growing body of literature exploring the relationships between biodiversity and mental health and wellbeing—based on early research conducted largely from social science perspectives—suggests that particular qualities within natural environments confer particular benefits. Results so far have been inconclusive and inconsistent, contributing to an incohesive body of evidence. While past reviews have focused on reporting variations in results, the present study builds on early reviews by exploring variations from the perspective of author disciplines and the use of different guiding theories, and variables used to measure biodiversity, mental health and wellbeing. This aims to address a research gap in understanding whether research in this topic has become more interdisciplinary or has employed more consistent study designs, which were highlighted as priorities in past reviews, but the progress of which has not yet been explored in depth. We found that research connecting biodiversity and mental health and wellbeing has become only marginally more interdisciplinary in recent years, and there is still a large inconsistency in the use of guiding theories, variables and overall study designs. The variation in disciplinary perspectives and methods reflects a growing interest in this field and the variety of ways researchers are trying to understand and test the complex relationships between biodiversity and mental health and wellbeing. Our study shows that there are unique perspectives that different disciplines can contribute to this body of research and continuing to increase collaboration between disciplines with the use of consistent mixed methods approaches in future may contribute to a more cohesive body of evidence. We provide a framework to conceptualize recommendations for future research, highlighting the importance of interdisciplinary collaboration at multiple scales, and importantly focusing on more specific, mechanistic studies to inform decision-making that provides co-benefits for biodiversity and mental health and wellbeing.

Keywords: subjective wellbeing, green prescription, research methods, systematic literature search, species richness, urban greenspace, interdisciplinary science

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INTRODUCTION

Increasing global urbanization presents significant challenges for both human mental health and biodiversity (Marselle et al., 2020). In recent decades, there has been immense research interest in demonstrating the importance of natural environments to human health and wellbeing (Beute et al., 2020; Felappi et al., 2020; Kosanic and Petzold, 2020), which has produced strong evidence that nature promotes various dimensions of human health and wellbeing. An initial search on the Web of Science for research connecting nature and health and wellbeing (for the years 2006–2021) returns approximately 800 results. In early research, measures of greenspace as they relate to health and wellbeing considered greenspace as a relatively homogenous "green" treatment, such as assessing the amount of greenness, or merely presence or absence. The relationships were predominantly explored in the context of aesthetic preference (Gobster et al., 2007), drawing upon psychoevolutionary theories such as savannah theory, which describe our natural affinity for relatively homogenous greenspaces (Hartig et al., 2011). Attention Restoration Theory (ART) is another theory that guided early work in this space, and which has continued to feature strongly in subsequent research. This theory is based on the ability of a landscape to renew personal adaptive resources and cognitive abilities to meet the demands of everyday life (Kaplan and Kaplan, 1989). As such, early work in this field has been driven from a social science perspective.

More recently, researchers have been interested in understanding the mechanisms by which correlations between health and wellbeing, and different greenspace characteristics arise. Studies that connect nature and health have highlighted that particular qualities within a natural space might provide particular benefits (Thompson Coon et al., 2011; Van den Berg et al., 2015; Reining et al., 2020). Thus, while earlier work looked at the quantity of green within an environment, more recent work seeks to explore more specific qualities of those environments. From this, a body of research has emerged which examines health and wellbeing benefits in the context of the ecological characteristics of particular spaces, such as naturalness or ecological integrity (Reining et al., 2020). Measures of biodiversity within a space, such as species richness, can provide an indication of the ecological functioning of that space. Additionally, more recent studies have begun to distinguish different aspects of human health and wellbeing, such as understanding whether landscapes with ecological benefit can also provide mental health and wellbeing benefits. This moves beyond the measures of greenness to uncover complexities and inform ways to approach the development of synergistic scenarios for biodiversity conservation and human health and wellbeing (Giusti and Samuelsson, 2020). The relevant definitions of biodiversity, mental health and psychological wellbeing for this research are as follows:

Biodiversity: *the variability among all living organisms from all sources and the ecological complexes of which they are a part; this includes diversity within species, between species and of ecosystems* (United Nations Convention on Biological Diversity, 2006).

Mental health: *a state of wellbeing in which an individual realizes their own abilities, can cope with the normal stresses of life, can work productively and is able to make a contribution to their community* (World Health Organisation, 2018).

Psychological wellbeing: *a combination of positive affective states such as happiness (the hedonic perspective) and functioning with optimal effectiveness in individual and social life (the eudaimonic perspective)* (Winefield et al., 2012).

More in depth studies of the biological quality of greenspace are essential for conservation purposes, and benefits to human mental health and psychological wellbeing are key for understanding how to best design, manage and conserve landscapes for both objectives. Understanding synergies between biodiversity and human wellbeing is particularly important within urban landscapes, because planners and policy makers around the globe are seeking to incorporate ecological restoration or design within the fabric urban environments (Fisher et al., 2021). Simultaneously, those involved in decision making are becoming increasingly aware of the importance of understanding landscapes in the context of social values, and the health and wellbeing benefits that natural spaces can provide (Jorgensen and Gobster, 2010).

Given the importance of mental health and psychological wellbeing to people, there is a growing interest in their presence as a societal aspiration with increasing relevance to decision making. There is therefore a need to understand which social, environmental and economic conditions can provide optimal population level wellbeing (Balvanera et al., 2016; Mavoa et al., 2019). However, significant challenges still exist in connecting research fields of biodiversity and health and wellbeing, particularly with regard to the lack of consistent metrics and outcomes explored (Jorgensen and Gobster, 2010).

Numerous studies have demonstrated that biodiversity can enrich the appreciation of natural spaces (Collar, 2003). Measures of biodiversity such as species richness or abundance have been shown to contribute to wellbeing; however, trends are inconsistent and inconclusive, and complicated largely by the use of different wellbeing and biodiversity variables (Dallimer et al., 2012; Fisher et al., 2021). While it has been shown that people generally perceived natural settings with high levels of complexity to be favorable (Kaplan and Kaplan, 1989; Southon et al., 2017), different people have vastly different preferences for landscapes and there are many environmental, social and personal factors that influence the benefits obtained from natural spaces. While some studies have found that objectively measured biodiversity is positively associated with wellbeing, others have found either weak positive, no correlation or inverse effects. For example, while Fuller et al. (2007) and Wolf et al. (2017) found that higher plant and bird diversity correlated positively with psychological wellbeing, Dallimer et al. (2012) and Methorst et al. (2020) found that psychological wellbeing decreased with higher plant diversity, while no correlations were found for measures such as butterfly diversity.

In **Figure 1** we show the growth in research connecting biodiversity to health and wellbeing, since 2006. An early work of this topic by Lovell et al. (2014) reviewed just eight studies,

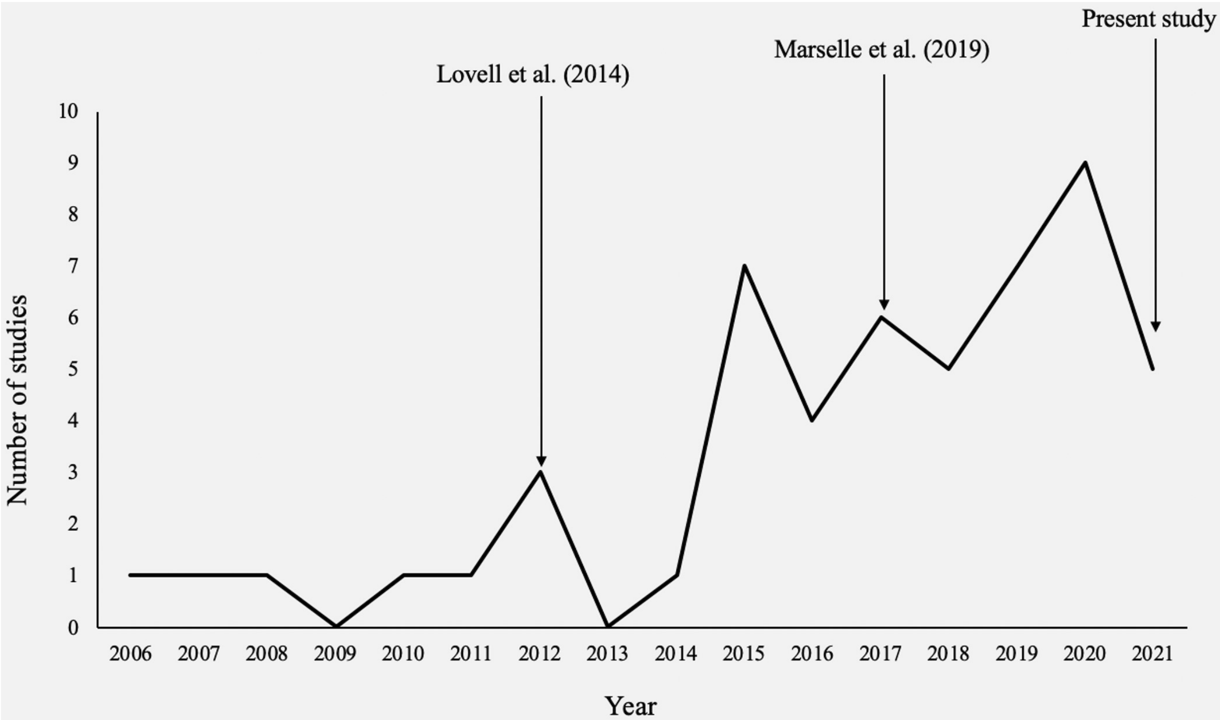

FIGURE 1 | Growth in research connecting biodiversity to mental health, between the years 2006–2021. Captions across the top with arrows indicate the years that have been included in the reviews conducted by Lovell et al. (2014) and Marselle et al. (2019) upon which this current study builds.

which explicitly explored links between biodiversity indicators and mental health. A more recent review conducted by Marselle et al. (2019) explored a further 16 studies. In the present study, we build upon this body of literature, and focus on the epistemological framing, disciplinary approaches and specific methodologies used to date. Earlier research connecting health and biodiversity called for interdisciplinary approaches that integrate multiple perspectives to guide planning and design (Chiang et al., 2017). Furthermore, previous reviews (Aerts et al., 2018; Collins et al., 2020; Methorst et al., 2020) have focused largely on reporting the variability in results. Reviews such as Marselle et al. (2019) highlight the need for more interdisciplinary work and robust experimental designs with consistent use of metrics, to provide for a more replicable and cohesive body of evidence of biodiversity-mental health relationships (Lovell et al., 2014; Chiang et al., 2017; Giusti and Samuelsson, 2020). There has, however, been little exploration of how different academic disciplines have contributed to the body of work so far, and how the use of different metrics has been evolving.

In order to understand whether there has been progress in terms of interdisciplinary work or consistent use of metrics, we aim to compare the 24 earlier studies reported by Marselle et al. (2019) with more recent works that have been published since 2018, which explicitly examine relationships between biodiversity and mental health and/or psychological wellbeing. In doing so, we seek to address a research gap in our understanding of how different disciplines have contributed to this body of research, and how different variables have been incorporated into study designs and may be contributing to inconsistencies in results. In order to address this research gap, we investigate the following research questions:

(1) which academic disciplines have contributed to research connecting biodiversity and mental health and psychological wellbeing, and have recent studies become more interdisciplinary than earlier studies? And (2) are the guiding theories and methods used in recent studies connecting biodiversity and mental health and psychological wellbeing similar or different from earlier studies?

By answering these questions, we seek to contribute to the current state of research linking biodiversity with health and wellbeing and how it is evolving based on early recommendations for interdisciplinary work and consistent use of methods. This may shed light on the understanding of how different academic disciplines have contributed to the body of research and highlight any robust elements of study designs to inform future research.

METHODS

Systematic Search Process

The systematic search conducted by Marselle et al. (2019) and previously by Lovell et al. (2014) was replicated for this study using the Web of Science, to capture more recent studies conducted following the Marselle et al. (2019) review. From this, it was determined that 28 recent studies met the inclusion criteria for full-text review. Data extraction methods used by Marselle et al. (2019) have been replicated and built upon. In doing this, together with determining the author disciplines as with earlier studies, changes in disciplinarity and use of variables in recent works could be compared.

Using the Web of Science, journal articles which had published original work between the years 2017–2021 were searched for. The search terms used and process of identifying papers appropriate for full-text review are summarized in **Table 1** and **Figure 2**. Although the year 2017 was included in the review by Marselle et al. (2019), this year was included in the present study in order to capture any potential studies that were missed in that review. The search terms (**Table 1**) and inclusion and exclusion criteria were adapted from Marselle et al. (2019), with the aim of being able to build upon and draw comparisons with the evidence base they obtained in their review.

Inclusion and Exclusion Criteria

The following inclusion criteria were used during the systematic search process:

TABLE 1 | Search terms used in the systematic search of the Web of Science.

Search terms	Number of studies
Biodiversity and “mental health”	86
Biodiversity and “mental wellbeing”	5
Biodiversity and “mental well-being”	7
Biodiversity and “psychological wellbeing”	5
Biodiversity and “subjective wellbeing”	1
Species diversity and “mental health”	16
Species richness and “mental health	8
Biodiversity and “psychological restoration”	13
Biodiversity and “perceived restorativeness”	12
Species richness and “psychological restoration”	2
Biodiversity and “attention restoration”	10
Total number of studies	165

With each search, only original, peer reviewed journal articles were searched for, between the years 2017–2021.

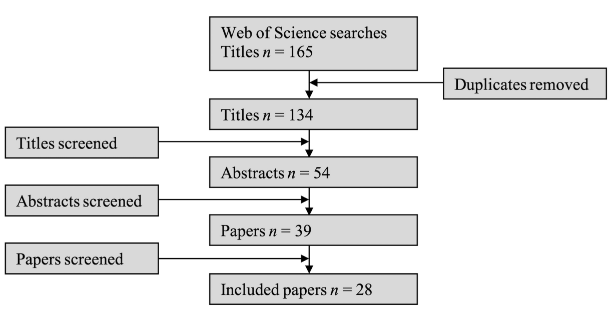

FIGURE 2 | Flow chart representing the systematic search and subsequent processes resulting in the inclusion of 28 additional papers, added to the body of 24 papers previously reviewed by Marselle et al. (2019).

1. Any peer reviewed study published between 01/01/2017 and 05/04/2021.
2. Any recognized and reliable study design with any population group, from any country—English language only.
3. An explicit exploration of one or more biodiversity variables such as species richness, or a setting protected for its biodiversity.
4. An explicit exploration of one or more mental health/wellbeing related outcome, including both objective and subjective measure of mental health or wellbeing.

Studies were excluded if they did not directly assess (i) biodiversity and (ii) mental health or mental wellbeing related outcome measures. Studies assessing preference, or the amount of greenspace without specifically measuring its biodiversity were excluded. Studies not reporting primary research such as review papers, were also excluded. In contrast to Marselle et al. (2019), studies assessing physiological stress measures were included in this search. It is acknowledged that including studies written only in English may have reduced the full scope of research that has been conducted on this topic.

Author Disciplines

This study builds upon the work of Marselle et al. (2019). Using the 24 studies they reviewed and the additional 28 recent studies found, the authorship of each of these studies has been reviewed. A novel approach was adopted to determine author disciplines for this research. To do this, Google Scholar author searches were used to determine author affiliations, “key words” and the topics on which they had previously written. Author “key words” were determined to be reflective of the focus discipline of researchers, as the particular school or organization that they belonged to did not always reflect their specific research interests or expertise. The discipline of the author was defined as the research specialization as self-described by the author on their Google Scholar profile. Where this information was not available, the school and home institution to which the author belonged was recorded from the relevant journal article webpage. This enabled the disciplinary framing of each study to be determined (acknowledging that this is a novel and rudimentary approach). For each of the 52 studies included in this review, each author was recorded and included in the analysis.

Key Elements of Study Design

Based on potential sources of inconsistency as described in previous works, key elements of study designs that may be contributing to inconsistent findings were extracted from each study:

- Guiding psychological theory.
- Biodiversity variables.
- Mental health and wellbeing variables.
- Type of study: correlational or mechanistic.

As seen in Marselle et al. (2019), biodiversity variables were assigned to one of three levels: (i) ecosystem/habitat, which includes variables such as landscape heterogeneity and structural diversity, (ii) species communities, which includes variables such as species richness, perceived species richness and the abundance of a specific taxonomic group, and (iii) single species level. Mental health and wellbeing variables were recorded as described in each study. Mental wellbeing variables were separated out and included attention restoration, recovery from stress, emotion/mood, and quality of life/life satisfaction, while "mental health" as a variable was recorded individually. In this study, a correlational type of study refers to an observational study that explores relationships between two or more variables, and may have obtained data using cross-sectional, longitudinal, cohort or retrospective record methods MacDonald et al. (2015). A mechanistic study refers to a study that directly explores the mechanisms of action of a particular experimental intervention.

Narrative Synthesis

A standardized spreadsheet was used to record all data, and a process of narrative synthesis, as developed by Popay et al. (2006), was used to interpret and compare the data from the two time periods. This type of synthesis has been used in previous reviews, including Marselle et al. (2019), and is a form of vote-counting synthesis that is useful when dealing with variables and data that are heterogenous, and where other forms of quantitative or statistical analyses are not suitable. This involves extracting key themes from the literature in order to compare and highlight points of difference.

RESULTS

Author Disciplines

Similar results emerge for disciplinarity from the two time periods, with environmental psychology continuing to be the most prominent author discipline. There has been a slight increase in interdisciplinary work in recent years, shown by a greater number of studies including three disciplinary perspectives (**Table 2**). The majority of studies, however, continue to include collaboration from just two disciplines. This is not necessarily surprising considering the two fields that this research combines. Again, unsurprisingly, the most prominent collaboration occurs between the disciplines of environmental psychology, and ecology or biological sciences. These results do, however, show that there is quite a diversity of disciplines that have contributed to the research so far. In recent works there has also been a greater contribution from the perspectives

TABLE 2 | Summary of results for author disciplines, comparing level of interdisciplinarity between early studies (2006–2017) and recent studies (2018–2021).

Number of disciplines	1	2	3	4
Study (2006–2017)	Johansson et al. (2014) (EP) Jones (2017) (EE) Duarte-Tagles et al. (2015) (PH) Saw et al. (2015) (BC)	Annerstedt et al. (2012) (EP, PH) van den Bosch et al. (2015) (EP, PH) Chiang et al. (2017) (EP, LA) Cracknell et al. (2016) (EP, MB) Marselle et al. (2015) (EP, PH) Marselle et al. (2016) (EP, PH) Carrus et al. (2015) (EP, EAS) Cox et al. (2017) (E, BC) de Jong et al. (2012) (EP, EOM) Fuller et al. (2007) (EP, BC) Grahn and Stigsdotter (2010) (EP, LA) Hoyle et al. (2017) (H, E) Luck et al. (2011) (EP, E) King et al. (2017) (BC, E, EAS)	Chang et al. (2016) (LA, H, E) Dallimer et al. (2012) (EP, BC, E) Huby et al. (2006) (M, E, BC) Wolf et al. (2017) (EP, EE, BC)	Björk et al. (2008) (EP, EOM, EAS, G) Wheeler et al. (2015) (EP, PH, E, M)
Total number of studies	4	14	4	2
Study (2018–2021)	Coldwell and Evans (2018) (BC) Meyer-Grandbastien et al. (2020) (E) Simkin et al. (2021) (EP) Reining et al. (2020) (G) Rantakokko et al. (2018) (HS)	De Bell et al. (2020) (G, HS) Fisher et al. (2021) (EP, BC) Harvey et al. (2020) (EP, BC) Giusti and Samuelsson (2020) (M, ES) Kortmann et al. (2021) (E, BC) Nghiem et al. (2021) (EP, BC) Young et al. (2020) (E, EP) Southon et al. (2018) (BC, H) Hussain et al. (2019) (BC, PH) Raymond et al. (2019) (EE, EAS) Methorst et al. (2021) (E, EE) Skevington et al. (2019) (EP, BC) Lindemann-Matthies and Matthies (2018) (E, BC)	Cameron et al. (2020) (EP, LA, M) Schebella et al. (2019) (EP, PH, M) Schebella et al. (2020) (EP, M, PH) Marselle et al. (2020) (EP, E, M) Mavoa et al. (2019) (E, G, M) Methorst et al. (2021) (EP, EE, EAS) Wyles et al. (2019) (EP, EE, MB)	Mears et al. (2019) (LA, M, E, PH) Wood et al. (2018) (EP, E, BC, PH)
Total number of studies	5	13	7	2

The disciplines contributing to each study are abbreviated in brackets beside each citation, with the key to disciplines below the table.
Key: EP, environmental psychology/psychology; EE, environmental economics; BC, biological and conservation sciences; M, mapping/mathematics and statistics; E, ecology; MB, marine biodiversity; PH, public health; HS, health sciences; EAS, environmental and agricultural sciences/engineering; G, geography; LA, landscape architecture; H, horticulture.

of landscape architecture, horticulture, health sciences and environmental economics. This may have implications for subsequent study designs and the application of findings.

Guiding Theory

Attention Restoration Theory (ART) and Stress Recovery Theory (SRT) are the most prominent guiding theories for studies from both time periods (**Figure 3**). These theories are similarly based on the notion that natural elements within landscapes provide the opportunity for mental relief from the many things that demand attention in everyday life and enable recovery from stress or the use of active attention through passive engagement with the environment (Kaplan and Kaplan, 1989).

The use of these theories reflects the short-term nature of most of the studies, which examine what may be referred to as "momentary wellbeing" (Fisher et al., 2021). Another theory that has been used in both early and recent works is the Biophilia Hypothesis (**Figure 3**), which suggests that humans have an innate tendency to seek connections with nature and other forms of life (Kellert, 1995). In recent work, new theories have been used such as Cultural Ecosystem Services, which describes the non-material benefits offered by natural environments, and which has importance for environmental economics and decision making (Milcu et al., 2013).

The most surprising finding here is that the majority of studies in both earlier and recent years do not refer to a guiding theory at all (**Figure 3**). This is reflected in the type of correlational study design used in most of the research as opposed to experimental, mechanistic studies. This may also have flow on effects in subsequent aspects of study designs and interpretation of findings. There is no notable correlation between author disciplines and the use of guiding theoretical frameworks. There are many studies with the perspective of environmental psychology without being guided by ART or other theory (e.g., Wheeler et al., 2015; Schebella et al., 2019), and conversely, there are numerous studies without this psychological perspective but which utilize a guiding psychological theory (e.g., Hoyle et al., 2017; Meyer-Grandbastien et al., 2020).

Biodiversity and Mental Health and Psychological Wellbeing Variables

There is a significant variation in the biodiversity and health and wellbeing variables used across the 52 studies reviewed (**Figures 4**, **5**). So far, there are no two studies which used the same study design or variables. Furthermore, it is important to note that many studies utilize multiple variables of both biodiversity and mental health.

The relative frequency with which particular metrics of biodiversity have been utilized are summarized in **Figure 4**. Comparing the two sets of studies [Early (E) and Recent (R)], it can be seen that species richness, measuring biodiversity at a species communities' level, is the most prominent metric used to evaluate biodiversity in the studies in both time periods, however, the variables and instruments used across studies are highly heterogeneous. The ecosystem/habitat level of biodiversity is also studied frequently across both time periods. In recent works, there is a greater representation of perceived species richness

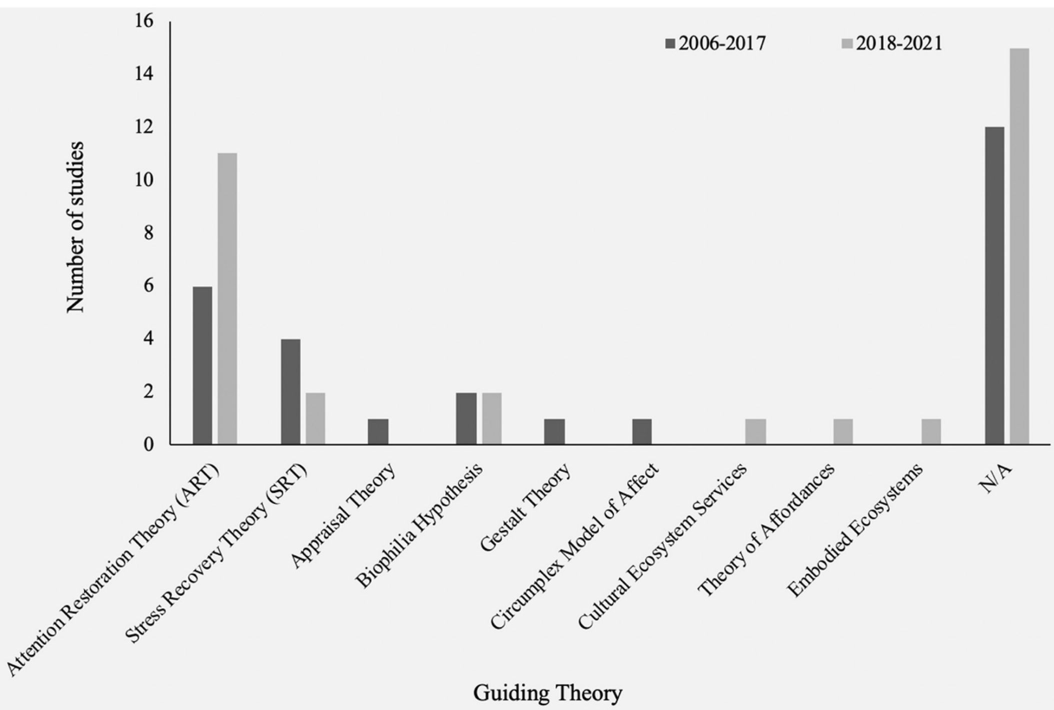

FIGURE 3 | Summary of the guiding theories used in studies from 2006–2017, compared to those between 2018–2021. The number of studies may exceed the total number of studies reviewed (52) as some studies utilized more than one guiding theory.

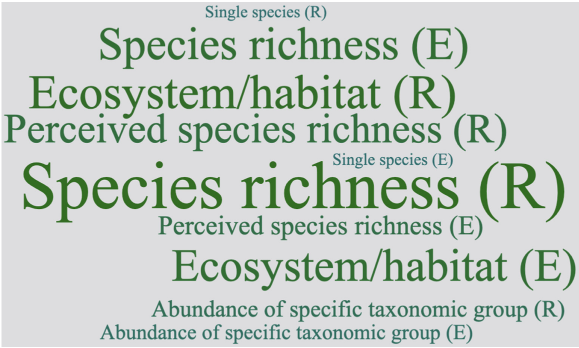

FIGURE 4 | Visual summary of results for biodiversity variables used in early studies (E) compared to recent studies (R). The three biodiversity levels represented are ecosystem/habitat, species communities (which includes species richness, perceived species richness, abundance of specific taxonomic group) and single species. The size of each variable represents the relative frequency with which it occurred in the body of literature, noting that some studies included more than one biodiversity variable (produced with worditout.com). A numerical representation of this data may be found in **Supplementary Appendix 1**.

FIGURE 5 | Visual summary of results for the mental health variables used in early studies (E) compared to recent studies (R). The size of each variable represents the relative frequency with which it occurred in the body of literature, noting that some studies included more than one mental health and/or wellbeing variable (produced with worditout.com). a numerical representation of this data may be found in **Supplementary Appendix 1**.

as a biodiversity variable. As reflected in the results for guiding theory (**Figure 3**), attention restoration continues to be the most prominent variable by which mental health is measured. This is followed by the use of the broader "mental health" variable, for both time periods. As with the biodiversity variables, many studies measured multiple different outcomes of mental health and/or wellbeing.

There is such a diversity of both mental health and biodiversity indicators utilized across studies that it remains difficult to compare or draw any correlation between discipline and use of variables. In recent work we do see a move to diversify the range of biodiversity variables tested, moving beyond just simply vegetation indices to include other taxa such as birds and insects (Wood et al., 2018; Cameron et al., 2020). We also see a large number of studies that include *perceived* biodiversity metrics, which have been shown to be of great importance to obtaining mental health and wellbeing benefits from nature (White et al., 2017; Nghiem et al., 2021).

Throughout the two time periods, correlational study designs continue to pervade, as reflected by the frequent use of ecosystem/habitat level of biodiversity, and broader "mental health" as a variable (**Figures 4**, **5**). There have yet only been five experimental studies conducted within the recent body of literature, and only one experimental study conducted in early literature. Within recent literature, newly emerging experimental study designs include those with the planting of specific experimental plots (Lindemann-Matthies and Matthies, 2018; Southon et al., 2018), which have been conducted from horticultural and/or landscape architectural perspective. Other emerging methodologies include measuring physiological stress responses in a quasi-experimental environment *in situ* (Lindemann-Matthies and Matthies, 2018; Hussain et al., 2019) or in response to the presentation of photographs or short videos (Schebella et al., 2020).

DISCUSSION

The diversity of methodologies, guiding frameworks and findings we found in our review are reflective of how interdisciplinary this topic area is and that researchers are only just beginning to develop ways to explore and understand the relationships between biodiversity and human mental health and psychological wellbeing. It is clear from the literature that exploring biodiversity-mental health relationships remains complex due to an underdeveloped yet growing understanding of the underlying mechanisms, and the large number of potential variables to be tested. More recent studies in particular (e.g., Southon et al., 2018; Schebella et al., 2020; Nghiem et al., 2021) highlight the importance of perceived biodiversity for psychological benefits and combining these with technology that measures physiological responses to biodiversity could continue to contribute to an understanding of the mechanisms underlying these relationships.

It is not surprising given the two disciplinary fields this research combines, that contribution from two disciplinary perspectives continues to be most prevalent in recent works. However, there are a number of unique contributions that can be made from additional disciplines outside of environmental psychology and ecology/biology. Increased representation of disciplines such as urban ecology, horticulture and landscape architecture in more recent studies (**Table 1**) indicates growing interest in using findings from this research to guide the design of urban landscapes, for which biodiversity conservation/restoration is becoming an increasingly prevalent objective.

The type of passive engagement with nature as described by the Attention Restoration Theory and the Stress Recovery Theory has been demonstrated by research to be the main way that we obtain psychological benefits from natural environments (Berto, 2014), so it is not surprising that these theories continue to feature heavily within the literature. However, given the psychological underpinnings of this research, a surprisingly low proportion of studies from both time blocks utilize guiding theoretical frameworks. This may have thus flow on effects within subsequent study designs, contributing to the inconsistency in methodologies and results. The prominent use of Attention Restoration Theory as a guiding framework reflects the short-term nature of many of these studies and the uncertainty still surrounding the mechanisms of biodiversity-mental health relationships.

A large proportion of studies explore relationships at the ecosystem/habitat level, as metrics such as habitat structural heterogeneity have been suggested to be the lens through which we perceive biodiversity within a landscape (Fuller et al., 2007; Beninde et al., 2015; Mears et al., 2019). The metrics of mental health, as compared to metrics of "momentary wellbeing" such as attention restoration or affect (Fisher et al., 2021) that consider the mechanisms by which health benefits arise, tend to show weak associations (Mears et al., 2019). Using these broader scale variables of health may therefore be contributing to inconsistent results. Additionally, there are a range of potential moderators and mediators that may be contributing to variations in the relationships between biodiversity and mental health, such as biodiversity knowledge and connection to nature (Van den Berg et al., 2015; Coldwell and Evans, 2018). Only two studies have investigated the role of a specific species on mental health and wellbeing and it will be important for future research to consider this scale to understand the role of specific species in contributing to mental wellbeing, and to understand whether particular species or combinations of species are more important than others, which could also be important in the context of particular endangered species or communities (Aerts et al., 2018; Mavoa et al., 2019).

The move toward the use of perceived biodiversity in recent literature is important as people's perception of a given environment, which can differ greatly from the actual environment, can have a strong impact on the wellbeing benefits derived (Coldwell and Evans, 2018; Nghiem et al., 2021). Perceived metrics such as species richness may have even greater influence than objectively measured biodiversity. Reining et al. (2020) highlight the fact that these studies have been conducted in distinctly different environments such as meadows and coastal areas, making it even more difficult to replicate or apply findings. Few studies focus on the diverse environments that might exist within one protected area, for example. Studying particular protected areas or particular species or communities therein may be of great importance in the context of particular locally endangered species or communities.

Findings from the present study are consistent with the mixed and inconclusive evidence reported in previous reviews (Aerts et al., 2018; Marselle et al., 2019), which describe the limited and conflicting evidence base for relationships between biodiversity and mental health. Inconsistencies in methods and results also highlights difficulties in studying these relationships within existing paradigmatic frameworks (Aerts et al., 2018; Giusti and Samuelsson, 2020). One way of addressing this in future could be to consider how these findings fit within alternative frameworks such as sustainable social-ecological systems, and the way interactions between mental health and biodiversity may help to understand and promote the restoration of the relationships between human and ecological functioning (Giusti and Samuelsson, 2020).

Recommendations for Future Research

It is clear that researchers are only just beginning to understand the nuances of relationships between biodiversity and mental health and psychological wellbeing, and the variation in methodologies and results reflects the growing interest in this area, and the many perspectives from which people are trying to understand and test these relationships. Although there is yet little consistency in terms of disciplinarity and methodologies, there are some key elements of study designs that can be informed by the unique perspective of particular disciplines. For example, the use of physiological measurements as informed by health sciences indicates a growing interest in understanding the underlying mechanisms at play, and the use of experimental planting plots from the perspective of landscape architecture or horticulture indicates an interest in using these findings to apply to landscape design. The importance of contribution from perspectives of psychology and ecology is clear for this type of research, however, the contribution of additional disciplines would be of great benefit for both understanding the underlying mechanisms of these relationships and applying them to decision-making or design.

To assist with bringing in a wider range of perspectives, we have created a visualization of how different disciplines might contribute to the body of research at various scales (**Figure 6**). At the top of this inverted triangle there are correlational studies, which so far have tended to work at large geographic scales, obtaining for example, census data for mental health variables and satellite derived biodiversity data. These operate at the larger ecosystem/habitat scales of biodiversity. As we move to the bottom of the triangle, there are the more experimental, mechanistic studies, which study biodiversity at the community or single species level, and which measure mental health at physiological or attention restoration level. These tend to be on much smaller geographic scales, in for example a specific greenspace or planting plot, with an accordingly small sample size. These are the findings that would be more specific than those at the top of the triangle, as indicated by the arrows on the right-hand side.

Beside the triangle on the left-hand side, some of the key research disciplines which may make the greatest impact at various scales are displayed. Environmental psychology and ecology are displayed across the whole length of the triangle as these will be of greatest importance at every level of this research. Collaboration between psychology and ecology could greatly benefit from additional perspectives, for different outcomes and applications of findings and to develop a more

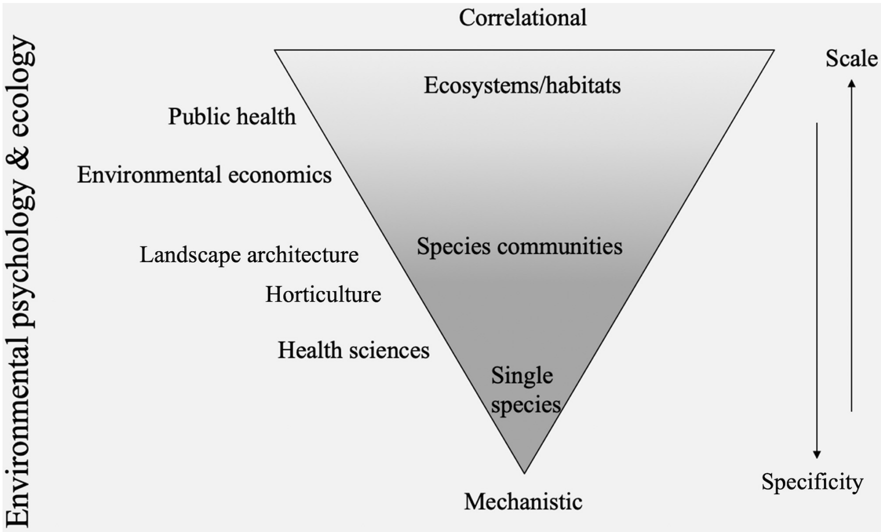

FIGURE 6 | Visual conceptualization of recommendations for future research exploring relationships between biodiversity and mental health and wellbeing.

mechanistic understanding of these relationships. In addition to the importance of foundations in psychology and ecology, perspectives such as public health and environmental economics could assist in framing the research from the perspective of decision making. For example, it could be important from the perspective of environmental economics and decision making to consider the concept of Cultural Ecosystem Services (including psychological restoration, improved physiological health, better social relations and spiritual development) (Kosanic and Petzold, 2020). A challenge exists in ascertaining whether cultural ecosystem benefits are sensitive to variations in biodiversity (King et al., 2017). Cultural Ecosystem Services could be an important concept for guiding decision making based on findings from research in this topic and warrants further investigation, as so far only one study has framed their research and findings in the context of this concept. Toward the mechanistic end of the research, perspectives of landscape architecture, horticulture, medical and health sciences may contribute important elements for experimental study designs, such as the use of physiological measurements in response to biodiversity, or the establishment of specific planting plots in which the ecology is well-documented. From a medical perspective for example, demonstrating stronger mechanistic links between biodiversity and mental health and wellbeing variables could provide important evidence to support healthcare approaches that formalize the role of nature, through concepts such as "green prescription" (Ulmer et al., 2016).

It is important to note that the categories shown in **Figure 6** are not clear-cut, and that important interactions would occur between various aspects of studies at different scales. However, overall, it will be more consistent work in these lower, more mechanistic areas of the inverted triangle, that will help to understand the underlying mechanisms and relationships on local scales that can account for variables such as sociodemographic factors and the importance of place-based data (Harvey et al., 2020). Mixed methods approaches that combine perspectives from multiple disciplines, with clear theoretical foundations and use of robust experimental study designs would enable a clearer progression of findings in this field. It is clear that a longer period of time, including the use of longitudinal research may be needed for these relationships to be understood, particularly in how biodiverse environments might contribute to mental health over time, for example beginning in childhood (Harvey et al., 2020). There is also a need to understand the full range of potential mediators and moderators of these relationships, such as biodiversity knowledge, connectedness to nature and socioeconomic factors, as these have been shown to influence the way benefits are obtained from biodiverse environments (Schebella et al., 2020). For example, psychological benefits have been reported as being greatest for those with lower socio-economic status, lower education level or low biodiversity knowledge in multiple studies (Hoyle et al., 2017; Coldwell and Evans, 2018; Marselle et al., 2020). Further investigation of the relative benefits of biodiversity for a greater diversity of populations and socioeconomic factors could be of great importance in the context of environmental equity. Similarly, it would be valuable to see a greater representation of non-western populations such as those represented in Fisher et al. (2021), Chiang et al. (2017), Chang et al. (2016), and Nghiem et al. (2021). We acknowledge, however, that the inclusion of English language only studies in our review may have precluded representation of such populations within the body of literature.

In order to progress toward transdisciplinary research within this field, there are several theoretical frameworks that future researchers might consider as a means of framing studies in a more holistic manner. The Actor Network Theory for example, provides a means by which researchers may frame relationships between biodiversity and health in the context of social-ecological system resilience (Horgan and Dimitrijević, 2018), which may be of great importance for applying findings to urban planning. Given the importance of perceived biodiversity as a metric conferring health and wellbeing benefits, we believe that tools such as the "Place Standard," which invite community participation in assessing greenspace qualities, may provide important information for decision-makers in understanding how communities relate to biodiversity through concepts such as place attachment and sense of place (Hasler, 2018; Colley and Craig, 2019). These concepts, which describe cognitive-emotional connections between people and their valued places have been demonstrated to mediate the wellbeing benefits obtained from greenspace (Scannell and Gifford, 2017; Basu et al., 2020; Han et al., 2021) and could therefore be important for future research to consider. However, by drawing upon a common set of metrics used to represent biodiversity, it will become easier to identify consistent pathways through which biodiversity contributes to human mental health, regardless of the frameworks being used. The metrics identified by this study make an excellent starting point for future studies looking to quantify biodiversity, particularly when they are combined with the conceptual framework we present in **Figure 6**.

One limitation of our study is the novel approach used to determine authorship, which may not have captured the true research specialization or interests of each author. However, our method does provide a useful indication of the disciplinary framing of the paper, and this brings a unique perspective that hasn't previously been examined. The 3 years gap between the two time periods studied is also a potential limitation, as a longer period of time may be required in order to draw meaningful conclusions as to whether research in this field is progressing toward a cohesive body of evidence. This may have also largely contributed to the lack of differences found between time periods for the variables examined. Furthermore, we acknowledge that the wide variation in terminology used by different disciplines contributing to this body of research may have precluded several studies from being included in our analysis. For example, a mechanistic study of frontal alpha asymmetry as a direct neurological response to urban green spaces was not included in our study as they used a contemplative landscape model to represent variation between greenspaces, rather than a direct measure of real or perceived biodiversity (e.g., Olszewska-Guizzo et al., 2020). Bringing together these diverse fields and framings is one of the key challenges that need to be addressed. The simplest solution is to ensure that biodiversity is explicitly included in the title, abstract or keywords of studies that include it as a consideration; and that a simplified search term such as "human health" or "wellbeing" is included in the keywords for studies with more specific measures, as demonstrated by Olszewska-Guizzo et al. (2020). This will be particularly important for studies that employ additional frameworks and tools such as the Actor Network Theory or Place Standard mentioned above, as these approaches do not always explicitly consider biodiversity, and therefore could potentially be overlooked in the absence of an additional signal.

The range of contributing disciplines and use of variables to the body of research reviewed in this study reflects a growing interest in this topic, and the complexity of relationships between biodiversity and mental health and wellbeing. The variation in uses of guiding theory and variables also reflects the many different ways by which researchers are trying to understand and test these relationships. The use of mixed methods approaches within recent works demonstrates a growing interest in understanding the underlying mechanisms (physiology) and how these relationships can inform greenspace design (experimental planting plots). This study has shown that while there is still great inconsistency and lack of coherence in study designs even in recent works, it is clear that different disciplines have unique perspectives to offer, and that continued interdisciplinary collaboration within mechanistic studies could contribute to a cohesive body of evidence, to inform strategies or policies that aim for win-win scenarios for both biodiversity conservation and mental health and wellbeing.

AUTHOR CONTRIBUTIONS

MH, AH, LM, and KL: conceptualization, development of methodology, and reviewing and editing draft manuscript. MH: investigation, compiling and curating data, analysis, and writing initial draft. All authors have read and agreed to the published version of the manuscript.

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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.2022.865727/full#supplementary-material

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