



HAL
open science

Prioritizing the provision of urban ecosystem services in deprived areas, a question of environmental justice

Wissal Selmi, Slaheddine Selmi, Jacques Teller, Christiane Weber, Emmanuel Léo Riviere, David Nowak

► **To cite this version:**

Wissal Selmi, Slaheddine Selmi, Jacques Teller, Christiane Weber, Emmanuel Léo Riviere, et al.. Prioritizing the provision of urban ecosystem services in deprived areas, a question of environmental justice. *AMBIO: A Journal of Environment and Society*, 2021, 50 (5), pp.1035-1046. 10.1007/s13280-020-01438-1 . hal-03030612

HAL Id: hal-03030612

<https://hal.science/hal-03030612v1>

Submitted on 30 Nov 2020

HAL is a multi-disciplinary open access archive for the deposit and dissemination of scientific research documents, whether they are published or not. The documents may come from teaching and research institutions in France or abroad, or from public or private research centers.

L'archive ouverte pluridisciplinaire **HAL**, est destinée au dépôt et à la diffusion de documents scientifiques de niveau recherche, publiés ou non, émanant des établissements d'enseignement et de recherche français ou étrangers, des laboratoires publics ou privés.

Preprint, the final version, of the article that has been transferred to the journal's production team.

To cite this article: Selmi, W., Selmi, S., Teller, J. Weber, C., Rivière, N., Nowak, D. Prioritizing the provision of urban ecosystem services in deprived areas, a question of environmental justice. *Ambio* (2020). <https://doi.org/10.1007/s13280-020-01438-1>

Prioritizing the provision of urban ecosystem services in deprived areas, a question of environmental justice

Abstract

The distribution of urban ecosystem services (UES) is often uneven across socioeconomic groups, leading to environmental justice issues. Understanding the distribution of UES across a landscape can help managers ensure an equitable distribution of services. While many past studies have focused on the distribution of green spaces in relation to socioeconomic variables, this research analyzes the distribution of UES provided by these green spaces. This research quantified air pollution removal, atmospheric carbon reduction, and surface runoff mitigation provided by urban trees in Strasbourg city (France). The provision of these three UES was studied at the census block scale by creating an index of UES delivery, which was contrasted with a constructed social deprivation index. Our results show that there is no significant association between the delivery of UES and social deprivation. Some deprived populations benefit from high UES delivery. Results also suggest that mapping associations between UES delivery and social deprivation should be integrated with future development plans to enhance the equitable distribution of UES. This study provides insights into the French context where studies about the distribution of UES at a small-area level remain lacking.

Keywords: Deprivation, Environmental justice, Urban ecosystem services, Urban trees

1
2
3
4
5
6
7
8
9
10
11
12
13
14
15
16
17
18
19
20
21
22
23
24
25

1. Introduction

Numerous studies have stressed the importance of quantifying urban ecosystem services (UES) at different scales and integrating them into decision-making processes (Ernstson et al. 2010; Kremer et al. 2016; Yang et al. 2018). However, while the concept of UES represents a bridge between social and ecological systems within a city, little is known about the relationship between social systems and ecosystem services delivery at the local level (Amini Parsa et al. 2019). This issue is increasingly addressed within the environmental justice (EJ) framework, given the recognition of urban green spaces (UGS) and their UES as local solutions to both local (e.g. air pollution, intense precipitation events) and global environmental burdens (e.g. climate change) (Zhao and Sander. 2015; Amini Parsa et al. 2019; Escobedo et al. 2019) and the growing conviction that uneven exposure to environmental burdens and benefits among social groups affects city sustainability (Baró et al. 2019). The concept of EJ was mainly developed in the USA in the 1970s, with a focus on the relationship between socioeconomic groups and the distribution of environmental burdens and benefits, assuming that disadvantaged communities are most exposed to environmental risks (Greenberg and Cidon. 1997; Agyeman et al. 2002; Kruize et al. 2007; Brown et al. 2012; Raddatz and Mennis. 2013; Boyce et al. 2016). The application of EJ framework to UES focuses on their immediate distribution at the local scale (e.g. city-scale, block group, neighborhood) where ecosystem services delivery and benefits are realized (Baró et al. 2019) and hence can support policies of sustainability to maintain their long-term production in a fair and just manner (de Groot et al. 2010; Boone and Fragkias. 2013).

An important scientific corpus has been constructed around environmental justice linked to UGS (Grove et al. 2006; Heynen et al. 2006; Pham et al. 2012). For example, the distribution of green cover in Edinburgh, Glasgow, Leicester, Oxford and Sheffield (UK) and San Juan

26 (Puerto Rico) has been found to be linked with urban characteristics like housing type, detached
27 housing and age of construction (Tratalos et al. 2007; Martinuzzi et al. 2018). In Terre Haute
28 (IN, USA), positive correlations were found between leaf area, population density, and median
29 income (Jensen et al. 2005). However, knowledge about the distribution of UES from an EJ
30 perspective remains limited (Wilkerson et al. 2018; Baró et al. 2019). For instance, few studies
31 have investigated the link between income level and temperature reduction (Jenerette et al.
32 2011), and the variation of amenities according to economic status and ethnic origin (Landry
33 and Chakraborty. 2009). Dobbs et al. (2014) found that UES provision in Melbourne (Australia)
34 was positively related to the degree of human development, assessed as the ratio between
35 education level and income. In Bogota (Colombia), Escobedo et al. (2015) found that the
36 wealthiest population had the greatest access to particulate matter removal by trees. In Europe,
37 EJ applied to UES is an emerging subject that has been investigated only in few urban areas,
38 notably Berlin (Lakes et al. 2014), Barcelona (Baró et al. 2016; Baró et al. 2019) and Porto
39 (Graça et al. 2017).

40 In France, investigations of UES within the EJ framework are rather scarce. Cohen et
41 al. (2012) assessed indirectly UES distribution in Paris through perception and species
42 indicators. In 2013, the national program of ecosystem functions and services assessment
43 (“L’évaluation française des écosystèmes et des services écosystémiques”: EFESE) was
44 launched. It addressed the disparity in the distribution of green spaces in French cities by
45 referring to some indicators like accessibility and attendance (EFESE 2018). However, the
46 distribution of UES across a socioeconomic gradient has not been considered.

47 In line with such evidence, it is essential to investigate more case studies to draw a more
48 complete and general picture about the interactions between socioeconomic factors and
49 ecosystem services depending on local contexts. Providing empirical research about how
50 socioeconomic factors influence ecosystem services can enrich research in this field. It may

51 also facilitate orienting urban planning toward increased sustainability and equity. Hence, our
52 paper contributes to the literature of environmental justice related to UES by providing a case
53 study combining biophysical indicators that directly measure urban ecosystem services and
54 socioeconomic indicators in a context of French cities.

55 Therefore, the main goal of this study is to conduct a quantitative assessment of UES
56 from a distributive justice perspective in Strasbourg city (France) and analyze if there are social-
57 spatial inequalities related to their provision. Distributive justice of ES is defined as the equal
58 access to ES among different social groups (Sievers-Glotzbach 2013; Schüle et al. 2019).

59 We focused on ecosystem services provided by trees located in public green spaces and
60 along streets as these areas are managed via public funding. We aimed to analyze the
61 distribution of UES delivery across a deprivation gradient defined by both social and material
62 disadvantages of local communities (Havard et al. 2008). Our study had three specific
63 objectives: 1) identify and assess socioeconomic variables that define deprivation, 2) quantify
64 three UES (air purification, runoff mitigation, atmospheric carbon reduction) and combine them
65 into one UES delivery index, and 3) investigate the relationship between social deprivation and
66 UES delivery to highlight potential environmental inequalities within the city.

67

68 **2. Methods**

69 **2.1. Study area**

70 This study was conducted in Strasbourg city, in Northeastern France (48°35'N and
71 7°45'W). The city covers an area of 7830 ha and supports about 275 700 inhabitants with a
72 population density of 35 inhabitants per ha (INSEE, 2016). The city is covered by 400 ha of
73 parks and about 1600 ha of protected alluvial forests that are major carbon sinks in the city. The
74 historic core of the city is surrounded by the green belt which is 650 m wide and supports 370
75 ha of gardens, vacant areas, and green spaces. Strasbourg occupies the 7th place in terms of

76 number of m² of tree cover per inhabitant (32 m² /inhabitant) while the first place is occupied
77 by Montpellier city (43 m² of tree cover / inhabitant; about 5700 ha; 268 456 inhabitants.).

78 Strasbourg is projected to develop more than 1000 ha of green and vacant spaces
79 between 2017 and 2030. This development would lead to the release of carbon stored in soils
80 and vegetation and weaken strategies for mitigating climate change. Increasing cumulative
81 rainfall along with substantial impervious surfaces and a dense hydrographic network makes
82 the city vulnerable to flooding (ADEUS 2018). Air pollution is also an environmental burden
83 in Strasbourg, as NO₂ exposure is increasing and PM₁₀ exceeds prescribed daily limits (Atmo
84 Grand-Est 2018).

85 The city contains 14 neighborhoods subdivided into 116 census blocks called IRIS
86 (Fig.1a). To ensure relevant analysis of social and ecological mismatches in Strasbourg, IRISs
87 with a limited population (less than 250 inhabitants) were excluded. According to Havard et al.
88 (2008), the excluded blocks with very low population numbers do not present sufficient
89 socioeconomic information. In total, 94 IRISs were included in our study sample. The average
90 population per IRIS was 2619 inhabitants with a mean IRIS area of 59 ha (44 inhabitants per
91 ha).

92 **2.2.Quantification of social deprivation**

93 Studies of UES from an EJ perspective are often based on literature reviews to select
94 socioeconomic variables to define social deprivation (e.g. McPhearson et al. 2013; Dobbs et al.
95 2014; Baró et al. 2019). In this work, we instead built a composite deprivation index based on
96 quantitative data and statistical analyses.

97 Data on socioeconomic variables that are often used to reflect social deprivation within
98 UES environmental justice frameworks (e.g. Pham et al. 2012; Cohen et al. 2012; McPhearson
99 et al. 2013; Graça et al. 2018) were obtained from the 2013 national census (INSEE 2016).

100 From these original data, eight socioeconomic variables providing inter-related
101 indicators of social deprivation were determined: (1) Median household income (in euros), (2)
102 Density of household with standard of living below poverty line (= number of household with
103 standard of living below poverty line / IRIS area in ha), (3) Density of rent-controlled housing
104 (HLM) (= number of HLM / IRIS area in ha), (4) proportion of immigrants in the population
105 (= number of immigrants / total number of IRIS inhabitants), (5) ratio of house owners to
106 tenants (= number of owner-occupied primary residences / number of non-owner occupied
107 primary residences), (6) ratio of unemployed to active people (= number of unemployed people
108 aged 15 and older / number of employed people aged 15 and older), (7) ratio of blue collar to
109 white collar (number of blue collar workers aged 15 and older / number of white collar workers
110 aged 15 and older), and (8) ratio of households without car to households with cars (= number
111 of households without a car / number of households with a car).

112

113 **2.3. Quantification of UGS and UES**

114 Three UES provided by public urban trees were selected: atmospheric carbon reduction,
115 surface runoff mitigation, and air pollution removal. The three UES are considered as the most
116 important in terms of distributional EJ (Amini Parsa et al. 2019; Baró et al. 2019). While
117 production and benefits of surface runoff mitigation and air pollution removal are mostly local
118 effects, atmospheric carbon reduction affects global climate regulation (Fisher et al. 2009).
119 Numerous other local urban ecosystem services that impact human well-being are affected by
120 local leaf area and biomass, such as building energy use and thermal comfort (Graça et al.
121 2018). Although it affects global climate regulation, it is of interest to study the carbon storage
122 and sequestration by trees in urban areas since they are more vulnerable to climate change and
123 where adaptation strategies should be developed (Amini Parsa et al. 2019). Producing

124 information about the distribution of this UES may contribute to enhanced climate justice even
125 at the local scale (Srang-iam 2011).

126

127 The three UES were assessed using i-Tree Eco model that provides information about
128 forest structure and benefits (Nowak et al. 2008; www.itreetools.org). i-Tree Eco guidelines (i-
129 Tree Eco User's Manual 2013) were followed to define the sample size and field survey
130 protocol to collect municipal urban forest data. The sampling design and data collection were
131 carried out by delimiting and stratifying public UGS within land-use classes, assessing tree
132 cover within public UGS, generating field samples, and collecting field data

133 The city and its public green spaces were stratified into 8 land-use classes based on
134 1:10000 land-cover database of the Alsace region provided for 2008 by the "Cooperation for
135 Geographic Information System in Alsace" (CIGAL). Using the 2010 map of public green
136 spaces provided the City Council of Strasbourg (EMS), 228 circular 11 m-radius plots (0.04 ha)
137 were selected. Study plots were distributed randomly within public green spaces using random
138 plot generation functionality of Arcgis (by generating x and y coordinates) and following the
139 model documentation (see Selmi et al. 2016).

140 The field survey was conducted during the leaf-on season of 2013. At each plot, general
141 information (date, plot address, GPS coordinates, land use, tree and shrub cover, ground cover
142 and plantable space) were recorded, as well as individual tree data (species, diameter at breast
143 height (DBH), total tree height, crown width, height to base of live crown, crown light exposure,
144 crown dieback percent, percent crown missing).

145 Local hourly pollution data were obtained from the regional Air Quality Agency (Atmo-
146 Est). These data were measured at six monitoring stations located within the study area over a
147 one-year period (from July 2012 to June 2013). Concentrations of carbon monoxide (CO),
148 nitrogen dioxide (NO₂), sulfur dioxide (SO₂), ground-level ozone (O₃), particulate matter of

149 less than 2.5 μm ($\text{PM}_{2.5}$) and particulate matter between 2.5 μm and 10 μm ($\text{PM}_{10 \text{ coarse}}$) were
150 calculated as averages values from the six monitoring stations. Weather data (wind speed,
151 precipitation, temperature, etc.) were retrieved from “Météo France” (two stations within the
152 study area) and one station of the National Oceanic and Atmospheric Administration (NOAA)
153 located at Strasbourg city.

154 The collected field and environmental data were combined in i-Tree Eco v5 to assess
155 basic structural information (e.g., tree density, species composition) and three UES: air
156 purification (dry deposition indicator), atmospheric carbon reduction (carbon storage and
157 sequestration indicators) and surface runoff mitigation (avoided runoff indicator). The average
158 rates of all UES per tree cover (i.e., g m^{-2} or $\text{m}^3 \text{m}^{-2}$) were calculated for each land-use class.
159 Following UES environmental justice literature (Dobbs et al. 2014; McPhearson et al. 2013;
160 Baró et al. 2019), the aggregated index was calculated to quantify UES at each IRIS following
161 the formula:

162
$$\text{UES}_i \text{ (g) (IRIS scale)} = \sum (\text{tree cover per land-use class } j \text{ (m}^2) \times \text{Average of UES}_i \text{ per}$$

163
$$\text{tree cover per land-use class } j \text{ (g m}^{-2} \text{ or m}^3 \text{ m}^{-2}\text{)}).$$
 The IRIS value was then divided by the area
164 of the corresponding IRIS to obtain a relative measure of that UES metric (UES m^{-2}).

165

166 **2.4. Statistical analysis**

167 Separate Principal Component Analyses (PCA) were conducted on the three sets of
168 variables (socioeconomic, UGS, and UES) to summarize the original variables into a few
169 components. Only components with eigenvalues exceeding 1 were retained (Vyas and
170 Kumaranayake 2006). PCA provided three composite indices of social deprivation, UGS
171 parameters and, UES delivery (see results).

172

173 A classification was further conducted according to social deprivation and UES delivery
174 indices to identify vulnerable areas (i.e. areas with high deprivation and low UES delivery).
175 The three classes (high, moderate and low) were identified based on the mean of UES and
176 deprivation indices $\pm \frac{1}{2}$ standard deviation (Faburel et Guymard 2008). The combination of
177 deprivation and UES classifications provided nine categories that were mapped using
178 Geographic Information using Esri's ArcGIS for Desktop 10.5.

179 Since study units were locations, spatial autocorrelation, and hence statistical non-
180 independence among observations (Legendre and Legendre 1998; Rangel et al. 2006) could
181 occur. Spatial autocorrelation among indices of deprivation, UGS parameters, and UES
182 delivery was checked for using Moran's correlograms (Legendre and Legendre 1998). The
183 studied IRISs were organized into 10 equiprevalent classes of geographic distance, with a lag
184 ranging from 0.527 to 2.897 km and sample size of 872 pairs of IRISs per class. For each
185 distance class, a Moran's I autocorrelation coefficient was calculated and tested for significance
186 using 999 permutations. A correlogram was then obtained by plotting the Moran's I values
187 against the upper boundaries of the corresponding distance classes, and the significance of the
188 whole correlogram was assessed by applying the Bonferroni correction for multiple tests
189 (Legendre and Legendre 1998). These spatial autocorrelation analyses were carried out using
190 SAM software 4.0 (Rangel et al. 2006).

191 The significance of social deprivation as a predictor of green space cover and UES
192 delivery was investigated using simple linear regressions. These regressions were performed
193 using Ordinary Least Square (OLS) models assuming independence among model residuals.
194 Spatial models were also run that considered model residuals as correlated and their covariance
195 as a function of the geographic distance between pairs of sites; Generalized Least Square (GLS)
196 models (Cressie 1993; Rangel et al. 2006). The competing non-spatial and spatial models were
197 then compared according to their AICc (Akaike Information Criteria corrected for small sample

198 size) and R^2 . The best model was the one having the lowest AICc and highest R^2 values (Rangel
199 et al. 2006). These regression analyses were performed using SAM software 4.0 (Rangel et al.
200 2006).

201 **3. Results**

202 **3.1. General results about social deprivation and UES delivery**

203 The first component extracted from a PCA on the original socioeconomic variables has
204 an eigenvalue of 5.26 and explains 66% of the variance of the original data set, corresponding
205 to an increasing axis of social deprivation (Table 1). However, the second component has an
206 eigenvalue less than 1 (0.79) and is not retained.

207 Twenty-eight IRISs are characterized as having the best living condition, where only
208 12% of households live below the poverty line, 88% of the population has a job and only 4%
209 of houses are rent-controlled. Conversely, the high deprivation class contains 33 IRISs that are
210 characterized by a high average of households who live below the poverty line (38%) and a
211 high percentage of immigrants (47%).

212 Descriptive statistics of UGS and UES parameters for the 94 IRIS are shown in Table
213 1. Average green spaces cover and tree cover across all IRIS are 14.7% and 5.7 % respectively.
214 Average species number and tree density are 114 species and 23 trees ha⁻¹ respectively. The
215 conducted PCA reduces UGS variables into one component accounting for 75% of the variance
216 (eigenvalue = 3.02), while the second component has an eigenvalue of 0.73. The first
217 component is positively correlated with all original variables, thus providing a composite index
218 of UGS characteristics (Table 1).

219 UES provided by public urban trees in Strasbourg city during 2013 are estimated to
220 128000 t and 4000 kg C year⁻¹ of stored and sequestered carbon respectively, 88 t year⁻¹ of
221 removed air pollutants including 1.2 t year⁻¹ of CO, 13.8 t year⁻¹ of NO₂, 55.9 t year⁻¹ of O₃;
222 11.8 t year⁻¹ of PM₁₀, 4.5 t year⁻¹ of PM_{2.5} and 1.0 t year⁻¹ of SO₂ (Note: PM₁₀ estimates exclude

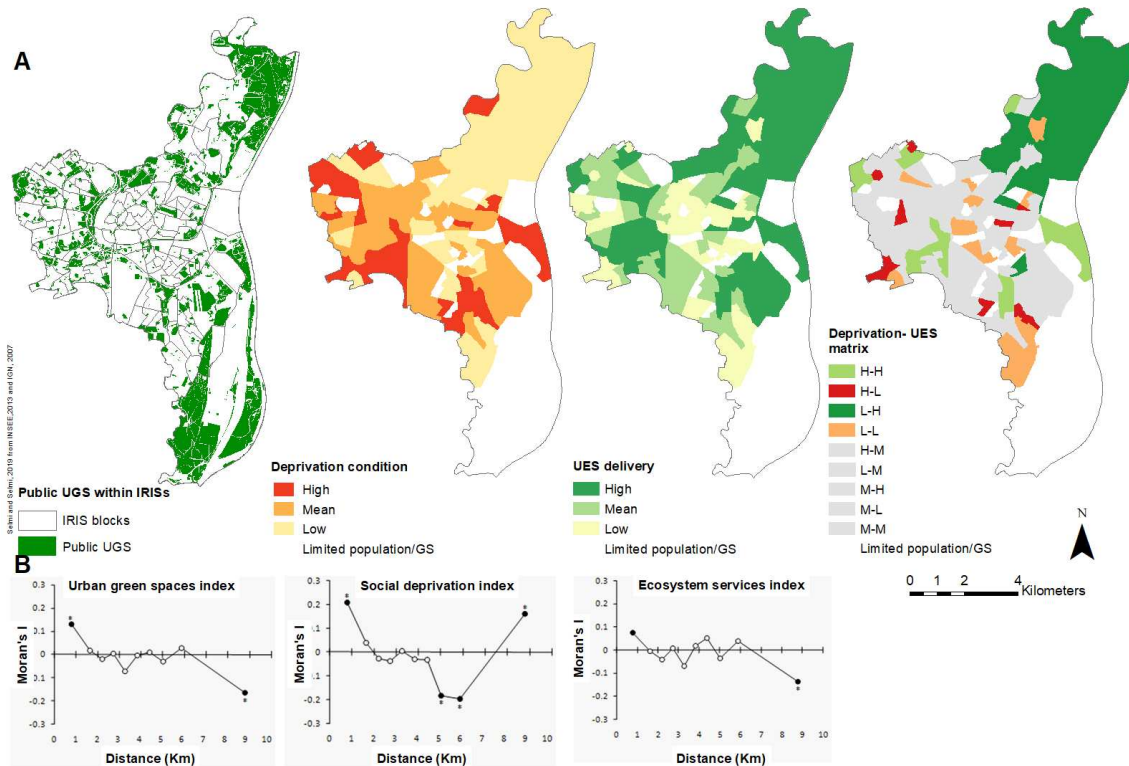
223 PM_{2.5} particles) and 24 160 m³ year⁻¹ of avoided runoff. A PCA summarizes the investigated
224 UES into a single composite index (first component) that explains 99% of the total variance
225 and has an eigenvalue of 8.92. In contrast, the second component has an eigenvalue of 0.07.
226 The composite index derived from this PCA is positively correlated with all original variables
227 (Table 1), indicating that high scores correspond to areas with increased delivery of UES.

228 **Table 1. List of assessed socioeconomic, green space and ecosystem service variables among IRISs, with descriptive statistics and correlations (Pearson**
 229 **correlation coefficient) with the indices derived from the Principal Component Analyses (sample size = 94 IRIS).**

Variables		Range	Mean \pm SD	Correlation test		Constructed index
				<i>r</i>	<i>P</i>	
Socio-economic variables	Median household income (euros)	10 847–38,290	19 228 \pm 5495	-0.799	< 0.0001	Deprivation index
	Density of household with standard of living below poverty line (number ha ⁻¹)	0–0.04	0.01 \pm 0.01	0.941	< 0.0001	
	Density of rent-controlled housing (HLM) (number ha ⁻¹)	0–89.61	10.89 \pm 16.01	0.730	< 0.0001	
	Proportion of immigrants in the population	0.05–0.40	0.20 \pm 0.08	0.808	< 0.0001	
	Ratio of house owners to tenants	0–3.99	0.53 \pm 0.59	-0.575	< 0.0001	
	Ratio of unemployed to active people	0.06–0.95	0.27 \pm 0.18	0.916	< 0.0001	
	Ratio of blue collar to white collar	0.05–60.68	4.38 \pm 9.19	0.765	< 0.0001	
	Ratio of households without car to households with cars	1.25–7.14	2.49 \pm 0.99	0.893	< 0.0001	
UGS variables	Cover of green areas (%)	0–68.30	16.80 \pm 15.30	0.917	< 0.0001	Urban vegetation index
	Number of species	17–671	114 \pm 108	0.602	< 0.0001	
	Tree density (trees ha ⁻¹)	0.25–182.29	23.22 \pm 27.77	0.941	< 0.0001	
	Tree cover (%)	0–44	5.70 \pm 7.50	0.962	< 0.0001	
Atmospheric carbon reduction	Carbon storage (kg ha ⁻¹)	0.32–36 058	7160 \pm 8195	0.994	< 0.0001	
	Carbon sequestration (kg ha ⁻¹ year ⁻¹)	0.01–1126	203 \pm 230	0.976	< 0.0001	
Surface runoff mitigation	Avoided runoff (m ³ ha ⁻¹)	0–9.76	1.54 \pm 1.85	0.994	< 0.0001	
UES variables	CO removal (g ha ⁻¹)	0–447	79 \pm 91	0.999	< 0.0001	Ecosystem services delivery index
	NO ₂ removal (g ha ⁻¹)	0–5146	912 \pm 1047	0.999	< 0.0001	
	O ₃ removal (g ha ⁻¹)	0–20774	3682 \pm 4226	0.999	< 0.0001	
	PM ₁₀ removal (g ha ⁻¹)	0–4374	775 \pm 890	0.999	< 0.0001	
	PM _{2.5} removal (g ha ⁻¹)	0–1676	297 \pm 341	0.999	< 0.0001	
	SO ₂ removal (g ha ⁻¹)	0–385	68 \pm 78	0.999	< 0.0001	

231 Twenty-four IRISs have a high level of UES delivery; 90% of tree cover and 82% of
232 green spaces are concentrated in these 24 IRISs. IRISs with high UES delivery produce about
233 88% of the total amount of UES in Strasbourg followed by IRISs with moderate delivery (10%)
234 and low delivery (2%). Relative UES delivery (normalized per area) is important in high
235 delivery areas except for air pollution removal, which is slightly higher in low delivery areas.
236 This difference is due to the variations of tree cover. For instance, in high delivery class
237 sequestered carbon is about 696 kg ha⁻¹ and removed pollution is about 6 g m⁻² (of tree cover).
238 In contrast, trees sequestered about 42 kg ha⁻¹ of carbon and removed about 7.5 g m⁻² of air
239 pollutants in low delivery class. However, the total amount of pollutant reduction is fifty times
240 greater in the high delivery class than the low class. This is due to the greater area and greater
241 percentage of tree cover (25 % of high delivery class is covered by trees while cover percentage
242 within low delivery class not exceed 2 %). Low-delivery areas are typically located in the
243 historic center and the Midwestern area of the city, which is dominated by industrial areas with
244 low tree cover (Fig. 1a).

245



246

247 **Figure 1. (A) Ecosystem services delivery, deprivation and combined (deprivation-UES) distribution maps**

248 **within Strasbourg. H-M through L-M were mapped with the same symbol as we want to highlight**

249 **extreme combinations. (B) Moran's I correlograms of the three indices in the 94 studied IRISs. Significant**

250 **Moran's I values at the 0.05 level are highlighted in bold. Asterisk-marked symbols represent Moran's I**

251 **values significant at the Bonferroni corrected level of $0.05/10 = 0.005$**

252 **3.2. Deprivation and UES delivery matrix**

253 Comparisons of deprivation condition with UES delivery (Fig. 1a, Table 2) show that 9

254 IRISs are vulnerable (high deprivation and low UES delivery: H-L) where 97% of its housing

255 is apartment-type and only 1% of the area is covered by trees. Eleven IRISs are in deprived

256 condition and moderate UES delivery (H-M). These two classes (H-L and H-M) are located in

257 central districts and the extreme western fringe of the city (Fig. 1A). Conversely, 8 IRISs have

258 a high quality of life with low deprivation and high UES delivery (L-H). They are located on

259 the northern side of the city where 31 % of the total area was covered by trees. Population across

260 the two classes is almost equally distributed with 10% and 9.8% of the total population living
 261 in vulnerable areas (H-L) and in areas with great quality of life (L-H) respectively.

262 Eight IRISs are considered as highly deprived and high producers of UES (H-H) with
 263 13% of the total population living in these 8 IRISs. 14 IRISs have a low deprivation condition
 264 and low UES production (L-L) where 13% of total inhabitants of the city is concentrated.

265 **Table 2. Distribution of IRIS across the deprivation-ES delivery matrix. Parentheses in the first**
 266 **row represent the percentage of IRISs and the parentheses in the second row represent the**
 267 **percentage of the population**

Deprivation ES	Low	Moderate	High	Total
Low	14 (32.6) (39.1)	20 (46.5) (31.6)	9 (20.9) (29.7)	43
Moderate	11 (40.7) (65.7)	5 (18.5) (13.8)	11 (40,7) (20.7)	27
High	8 (33.3) (34.7)	8 (33.3) (38.7)	8 (33.3) (26.6)	24
Total	33	33	28	94

268

269 **3.3. Spatial autocorrelation and relationships between social deprivation, green**
 270 **space and UES parameters**

271 Moran's I correlograms show that the social deprivation, UGS, and UES indices exhibit
 272 significant spatial autocorrelation (Fig. 1B). Of particular importance is the positive spatial
 273 autocorrelation in the first distance class, i.e. pairs of IRISs less than 0.527 km from each other
 274 (social deprivation index: $I = 0.207$, $P < 0.001$; UGS index: $I = 0.131$, $P < 0.001$; UES index:
 275 $I = 0.074$; $P = 0.043$). IRISs close to each other show more similar levels of social deprivation,
 276 UGS and UES delivery than predicted by a random distribution of these variables, reflecting
 277 the 'contagious' nature of these parameters in the studied city.

278

279 This significant spatial autocorrelation would suggest that nearby IRISs could not be
 280 considered as independent sampling units when investigating the relationships of social
 281 deprivation with UGS and UES parameters. This finding is supported by the results of

282 regression analyses, which show that models accounting for spatial covariance among
283 observations provide better fit to the data than standard models that assumed independence
284 among model errors, based on AICc and R² values (see below). These regression models show
285 negative but still non-significant associations of social deprivation with both UGS (non-spatial
286 model: AICc = 272, R² = 0.1%; $\beta \pm SE = -0.020 \pm 0.104$; $t = -0.193$, $P = 0.8470$; spatial model:
287 AICc = 237, R² = 33%; $\beta \pm SE = -0.107 \pm 0.103$; $t = -1.043$, $P = 0.3000$) and UES (non-spatial
288 model: AICc = 277, R² = 0.1%; $\beta \pm SE = -0.034 \pm 0.104$; $t = -0.327$, $P = 0.7450$; spatial model:
289 AICc = 247, R² = 26%; $\beta \pm SE = -0.099 \pm 0.114$; $t = -0.870$, $P = 0.3870$).

290

291 **4. Discussion**

292 We examined the distribution of UES delivery and whether it is associated with social
293 deprivation condition across census blocks in Strasbourg. Developing a spatial assessment
294 approach at this level is of interest because the majority of UES are produced at the local scale,
295 except for climate regulation which is provided at a larger scale (city, region, country) (Baró et
296 al. 2019). Thus, priorities of decision-making may change by acting on low provision areas and
297 protecting high provision ones. In agreement with Escobedo et al. (2011) and de Groot et al.
298 (2010), we argue that urban green patches are most affected by planning and management
299 decisions at the census block scale. However, the implication of different scales should be
300 considered because the relevance of UES could change among social groups and scales
301 depending on needs and local context. Also, managing urban vegetation and related ecosystem
302 services should be addressed in an integrative way by considering all impacts across temporal,
303 political, economic, and spatial scales (Escobedo et al. 2011, Andersson et al. 2019).

304 Unlike other studies carried out in Australia (Dobbs et al, 2014) and USA (Landry and
305 Chakraborty 2009), our results show no significant relationship between UES delivery and
306 social deprivation. This finding is in line with those previously reported in Barcelona (Baró et

307 al. 2019), where no association was found between ES distribution and household income. The
308 explanation of our results may rely on two assumptions. The first one is related to the fact that
309 our analysis was conducted on public green spaces where planting and managing decisions are
310 made regardless of the socioeconomic status of the neighborhood. This is in line with findings
311 of Kuras et al. (2020) that suggest that neutral relationship between socioeconomic status and
312 biodiversity, in general, occurs in public land uses. This raises the question whether local
313 policies are not interested in establishing a participatory approach or the residents do not feel
314 concerned by planting strategies of their neighborhood. More investigations are needed to
315 understand interactions between planners, managers and citizens and how they influence
316 greening decisions. The second potential explanation could be a possible difference in the speed
317 of change of green spaces compared to the socioeconomic situation of the population
318 concerned. Indeed, the spatial distribution of green spaces is not likely to evolve at the same
319 speed as the socioeconomic parameters of the population, as has previously been highlighted in
320 a New Zealand study on the relationship between green spaces and public health (Richardson
321 et al. 2010).

322 Nevertheless, there is one issue that raises particular concern related to the urban
323 mutation in Strasbourg at the mid and long terms, namely the gentrification (Gerber 1999;
324 Blanchard 2019). Municipal investments want to make the city uniform in terms of quality of
325 life. This uniformity can make it greener, but may increase property values which can displace
326 socioeconomically disadvantaged classes and exclude them from access to amenities
327 (Anguelovski et al. 2018). In New York (USA), the restoration of Prospect Park resulted in a
328 change of the socioeconomic status around the park toward socially advantaged population,
329 while in Barcelona, it was shown that gentrification took place in old industrial and desirable
330 neighborhoods, but not in the socially deprived neighborhood (Anguelovski et al. 2018).

331 Studying this process in Strasbourg will provide relevant information about the impact of
332 planning strategies (including greening strategies) on creating (or not) new inequities.

333

334 The trend of negative association between ES delivery and social deprivation in
335 Strasbourg city may be visible over a more extended period, with taking into account the
336 consequences of gentrification and interactions between the city and surrounding villages.
337 Further research should investigate shifts in historical expansion of the city, population
338 demography, socioeconomic status across neighborhoods to better understand the link between
339 UES delivery and deprivation conditions over time. This research could help substantiate the
340 idea of Pickett et al. (2008), who argued that the ecological dynamic within a city is slower than
341 the social dynamics. Thus, the ecological structure of a particular neighborhood may reflect a
342 previous social structure, not the existing one.

343 Our approach not only demonstrates the absence of a relationship between ES delivery
344 and social deprivation, but also detects their spatial distribution including vulnerable areas (i.e.
345 low ES provision and high social deprivation) and advantaged areas (i.e. high ES provision and
346 low social deprivation). In our case, it is also important to notice that descriptive analyses show
347 that the number of deprived neighborhoods with high UES is not negligible, which is consistent
348 with findings in other European cities like Porto (Graça et al. 2017), Barcelona (Baró et al.
349 2019) and Paris (Cohen et al. 2012). This neutral relationship may be explained by the planning
350 trends that facilitate greening actions and thus, UES provision across time (Kuras et al. 2020).
351 In Strasbourg, major urban projects were carried out in the urban fringe between 1950 and 1970
352 that represented the core of the social housing urban fabric of the city. At that time, those
353 neighborhoods were designed toward a predefined percentage of 10 to 40% of green spaces
354 (POS 1979), which explains the observed high level of UES delivery. Consequently, other
355 mechanisms like urban policies and landscape structure seem to have an impact on UES

356 delivery and should be studied to identify factors that spatially influence ecosystem services
357 distribution.

358 As demonstrated by other findings (Dobbs et al. 2014; Wilkerson et al. 2018), studying
359 the link between landscape structure and UES distribution gives insights on the consequences
360 of planning strategies on ecosystem services provision. These strategies, which have to be
361 sustainable, should include actions to improve the quantity, the distribution and the connection
362 between green spaces to equitably increase ecosystem services provision and hence to ensure a
363 balance between social and ecological health within the city.

364 UGS within socially-deprived neighborhoods where high population density occurs
365 may be faced with over-use. In addition to maintaining current UES potential, management
366 planning in those neighborhoods could focus on environmental quality drivers like recreational
367 facilities, diversified vegetation features, and maintenance types. Moreover, daily experiences
368 and perceptions of users could be taken into account to help green spaces services meet resident
369 needs (Hoffmann et al. 2017). Improved residents' awareness about nearby UGS and its
370 services could encourage support for its development and maintenance. It could also help build
371 community cohesion by recognizing local community power and expertise, which facilitates
372 the involvement of residents into the local decision-making process (Lovell and Taylor 2013).

373 High and moderate UES provisions were generally located within the green belt of the
374 city that is entirely included within the so-called "Urban Natural Park". This park resides within
375 the deprived districts "Elsau", "Koenigshoffen" and "Montagne verte" and surrounds the
376 central core of the city. One issue of concern is that 20% of the green belt could be built in the
377 future. Protecting this resource by reducing development can help preserve the current UES
378 flow. Enhancing the connectivity of green spaces within the green belt could also connect
379 neighborhoods of different socioeconomic levels, improve social mixing, and provide equitable
380 access to UES among different social categories. New urban projects in proposed development

381 areas should address the gentrification risk to ensure equal opportunities to a healthy
382 environment among various socioeconomic classes. Urban policies could go beyond
383 distributional justice to procedural justice to involve different social classes in making
384 connections among neighborhoods.

385 Spatial results of areas with high or low UES delivery could be crossed with the zoning
386 of the local urbanism plan (PLU) to better inform on the risk of environmental degradation if
387 future development occurs. For instance, few high UES provision patches were located within
388 areas assigned to future urbanization. In this case, strategies to maintain UES provision in these
389 areas could be investigated (Baró et al. 2016). Conversely, areas assigned to urbanization with
390 low UES provision could represent an opportunity to create sustainable urban projects that
391 support social mix, meet social needs, and enhance UES. According to existing literature,
392 addressing these issues is still challenging for urban renewal policies (Anguelovski and
393 Martinez Alier 2014; Chen and Yonh Hu 2015).

394 Studying UES from the EJ perspective may produce information stating an equitable
395 distribution of services provided by UGS. While approaches that can holistically quantify UES
396 are needed (Dobbs et al. 2014; Riley and Gardiner 2019), it may also be of interest to analyze
397 only the most easily measured variable, most relevant for local policies, or the most needed by
398 citizens and use it as a proxy for all UES only if a strong correlation is found between them,
399 which is our case. Although the correlation is demonstrated for the three quantified services,
400 the link between these services and others such as cultural services or temperature reduction
401 services should be explored to have a relatively global vision of the potential of vegetation to
402 produce benefits for the population. Furthermore, decision-makers could go beyond
403 characterizing the non-significant disparity in UES distribution and move into identifying
404 vulnerable neighborhoods and establishing priorities.

405 In sum, our study does not reveal a direct relationship between socioeconomic status
406 and the distribution of UES provision in Strasbourg city. To have a complete vision of UES
407 and meet local needs, studies could investigate the impact of socioeconomic factors on the
408 demand of UES and the relationship between supply and demand of UES (Baró et al. 2015;
409 Larondelle and Lauf 2016). Wilkerson et al. (2018) report that socially advantaged population
410 prioritizes recreational services instead of regulation services or production services. Given our
411 results, does social deprivation mean high demand for UES in Strasbourg city? How does
412 socioeconomic status influence perception and behaviors?

413 **5. Research limitations**

414 The estimation of UES production by i-Tree Eco was analyzed by land-use classes and
415 disaggregated into the IRIS level. This limitation could be overcome with an increased sampling
416 that would provide enough plot data to analyze results by IRIS. Multiple models run with locally
417 specific weather, and pollution data could increase local specificity of the analyses, but these
418 data are currently lacking.

419

420 This study focused on the IRIS level with varying area. Variables were converted to
421 density, proportion, and ratios for all IRISs to normalize the area. Dividing the city into equal
422 spatial units could be interesting to reduce the sensitivity of results to IRISs areas. A multi-scale
423 study could be more effective to understand different synergies between UES and to connect
424 local and regional actors.

425 Only tree cover and green spaces densities were analyzed. However, other factors could
426 influence the provision of UES such as age, diameter, green spaces area, and the total number
427 of trees (Dobbs et al. 2014). Further research is needed to understand better interactions between
428 structural variables, ES flow and socioeconomic status. Similarly, only regulation services were
429 examined in this study. These services mainly depend on tree cover as it presents the ecological

430 reservoir that ensures their production. However, the distribution of other categories of services
431 such as cultural services may not be associated with tree cover but rather with perception, use,
432 accessibility, etc. (Riley and Gardiner 2019).

433 In this study only public green spaces have been considered. However, private green
434 spaces, such as private gardens, cemeteries, allotment gardens and schools' gardens, may also
435 contribute to total UES provision in the city. Although these private green spaces are not easily
436 controlled by urban managers, their connection with public spaces to benefit from their
437 ecological potential would be an interesting path to explore through the environmental justice
438 framework. In addition, it would also be interesting to verify whether there is compensation
439 between the UES produced by private spaces and the UES produced by public green spaces
440 (i.e. verify whether the neighborhoods with low public green spaces and low ES provision have
441 more access to private gardens and their associated ES).

442

443 Finally, the main challenge of this study was the combination of socioeconomic and
444 ecological indicators. Composite indicators were created and classified into low, moderate, and
445 high clusters according to mean and standard deviation values. The aim was to simplify the
446 information, but it would be interesting to study the impact of each socioeconomic indicator on
447 each UES. Extensive research is also needed to define a mapping method that is transferable
448 and easy for decision-makers to use.

449

450 **6. Conclusion**

451 This study used a spatially explicit approach to visualize the distribution of UES across
452 socioeconomic conditions in Strasbourg, France. Using tools that connect UES to a social
453 context is useful to identify priority areas where enhancing UES and social justice are needed.
454 Urban policies could prioritize areas with high deprivation and low UES provision by defining

455 strategies that improve the quality and/or the number of green spaces, optimize UES and foster
456 the connection between population and local green spaces. Distributive environmental justice
457 studies could be followed by investigations of interactional and procedural justice that aim to
458 highlight the personal experiences of populations and their involvement in the decision-making
459 process. Comparing results with perceived access to UES according to the socioeconomic
460 condition of the population can help to identify the well-being demand in the city. A key
461 challenge of this work is to integrate outcomes of UES mapping into decision-making processes
462 that support future urban planning strategies and are consistent with social needs and
463 environmental restoration of the city. Considering environmental justice patterns associated
464 with UES in current urban policies may impact the interactions between society and semi-
465 natural ecosystems within the city in the future.

466 **Acknowledgements**

467 This work was supported by “Institution de la Recherche et de l’Enseignement Supérieur
468 Agricoles (IRESA-Tunisia), ‘Zone Atelier Environnementale Strasbourg’ (ZAEU) and
469 “Eurométropole de Strasbourg” (EMS).

470 **References**

- 471 ADEUS. 2018. Plan Climat Air Energie territorial de l’Eurométropole de Strasbourg.
472 Diagnostic et evolution de la situation Climat Air Energie, Strasbourg, France.
- 473 Agyeman, J., R. D. Bullard, and B. Evans. 2002. Exploring the Nexus: Bringing Together
474 Sustainability, Environmental Justice and Equity. *Space and Polity* 6: 77–90.
475 doi:10.1080/13562570220137907.
- 476 Amini Parsa, V., E. Salehi, A. R. Yavari, and P. M. van Bodegom. 2019. An improved method
477 for assessing mismatches between supply and demand in urban regulating ecosystem
478 services: A case study in Tabriz, Iran. Edited by Ali Kharrazi. *PLOS ONE* 14: e0220750.
479 doi:10.1371/journal.pone.0220750.

480 Andersson, E., J. Langemeyer, S. Borgström, T. McPhearson, D. Haase, J. Kronenberg, D. N.
481 Barton, M. Davis, et al. 2019. Enabling Green and Blue Infrastructure to Improve
482 Contributions to Human Well-Being and Equity in Urban Systems. *BioScience* 69: 566–
483 574. doi:10.1093/biosci/biz058.

484 Anguelovski, I., J. J. T. Connolly, L. Masip, and H. Pearsall. 2018. Assessing green
485 gentrification in historically disenfranchised neighborhoods: a longitudinal and spatial
486 analysis of Barcelona. *Urban Geography* 39: 458–491.
487 doi:10.1080/02723638.2017.1349987.

488 Anguelovski, I., and J. Martínez Alier. 2014. The ‘Environmentalism of the Poor’ revisited:
489 Territory and place in disconnected glocal struggles. *Ecological Economics* 102: 167–176.
490 doi:10.1016/j.ecolecon.2014.04.005.

491 Atmo GRAND-EST, 2018. Bilan de la qualité de l’air. 2018. Retrieved 10 March, 2020, from
492 http://www.atmo-grandest.eu/sites/prod/files/2019-12/BQA2018_indice2.pdf

493 Baró, F., D. Haase, E. Gómez-Baggethun, and N. Frantzeskaki. 2015. Mismatches between
494 ecosystem services supply and demand in urban areas: A quantitative assessment in five
495 European cities. *Ecological Indicators* 55: 146–158. doi:10.1016/j.ecolind.2015.03.013.

496 Baró, F., A. Calderón-Argelich, J. Langemeyer, and J. J. T. Connolly. 2019. Under one canopy?
497 Assessing the distributional environmental justice implications of street tree benefits in
498 Barcelona. *Environ Sci Policy* 102: 54–64. doi:10.1016/j.envsci.2019.08.016.

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

503 Blanchard, N. Etude du phénomène d’embourgeoisement à l’œuvre dans un quartier en
504 mutation à Strasbourg: 19.

505 Boone, C. G., and M. Fragkias, ed. 2013. *Urbanization and Sustainability*. Dordrecht: Springer
506 Netherlands. doi:10.1007/978-94-007-5666-3.

507 Boyce, J. K., K. Zwickl, and M. Ash. 2016. Measuring environmental inequality. *Ecological*
508 *Economics* 124: 114–123. doi:10.1016/j.ecolecon.2016.01.014.

509 Brown, A., A. Dayal, and C. Rumbaitis Del Rio. 2012. From practice to theory: emerging
510 lessons from Asia for building urban climate change resilience. *Environment and*
511 *Urbanization* 24: 531–556. doi:10.1177/0956247812456490.

512 Chen, W. Y., and F. Z. Y. Hu. 2015. Producing nature for public: Land-based urbanization and
513 provision of public green spaces in China. *Applied Geography* 58: 32–40.
514 doi:10.1016/j.apgeog.2015.01.007.

515 Cohen, M., R. Baudoin, M. Palibrk, N. Persyn, and C. Rhein. 2012. Urban biodiversity and
516 social inequalities in built-up cities: New evidences, next questions. The example of Paris,
517 France. *Landscape and Urban Planning* 106: 277–287.
518 doi:10.1016/j.landurbplan.2012.03.007.

519 Cressie, N.A.C. 1993. *Statistics for spatial data*. Wiley-Interscience Publications, New York.

520 de Groot, R. S., R. Alkemade, L. Braat, L. Hein, and L. Willemen. 2010. Challenges in
521 integrating the concept of ecosystem services and values in landscape planning,
522 management and decision making. *Ecological Complexity* 7: 260–272.
523 doi:10.1016/j.ecocom.2009.10.006.

524 Dobbs, C., D. Kendal, and C. R. Nitschke. 2014. Multiple ecosystem services and disservices
525 of the urban forest establishing their connections with landscape structure and
526 sociodemographics. *Ecological Indicators* 43: 44–55. doi:10.1016/j.ecolind.2014.02.007.

527 EFESE. 2018. Les écosystèmes urbains. Retrieved 5 April, 2019, from
528 [https://www.ecologie.gouv.fr/evaluation-francaise-desecosystemes-et-des-services-](https://www.ecologie.gouv.fr/evaluation-francaise-desecosystemes-et-des-services-ecosystemiques)
529 [ecosystemiques](https://www.ecologie.gouv.fr/evaluation-francaise-desecosystemes-et-des-services-ecosystemiques).

530 Ernstson, H., S. Barthel, E. Andersson, and S. T. Borgström. 2010. Scale-crossing brokers and
531 network governance of urban ecosystem services: The case of stockholm. *Ecology and*
532 *Society* 15. doi:10.5751/ES-03692-150428.

533 Escobedo, F. J., N. Clerici, C. L. Staudhammer, and G. T. Corzo. 2015. Socio-ecological
534 dynamics and inequality in Bogotá, Colombia's public urban forests and their ecosystem
535 services. *Urban Forestry and Urban Greening* 14: 1040–1053.
536 doi:10.1016/j.ufug.2015.09.011.

537 Escobedo, F. J., V. Giannico, C. Y. Jim, G. Sanesi, and R. Laforzezza. 2019. Urban forests,
538 ecosystem services, green infrastructure and nature-based solutions: Nexus or evolving
539 metaphors? *Urban Forestry & Urban Greening* 37. Green Infrastructures: Nature Based
540 Solutions for Sustainable and Resilient Cities: 3–12. doi:10.1016/j.ufug.2018.02.011.

541 Escobedo, F. J., T. Kroeger, and J. E. Wagner. 2011. Urban forests and pollution mitigation:
542 Analyzing ecosystem services and disservices. *Environmental Pollution* 159. Selected
543 Papers from the Conference Urban Environmental Pollution: Overcoming Obstacles to
544 Sustainability and Quality of Life (UEP2010), 20-23 June 2010, Boston, USA: 2078–
545 2087. doi:10.1016/j.envpol.2011.01.010.

546 Faburel, G., and S. Gueymard. 2008. Inégalités environnementales en région Île-de-France : le
547 rôle structurant des facteurs négatifs de l'environnement et des choix politiques afférents.
548 *Espace populations sociétés. Space populations societies:* 159–172.
549 doi:10.4000/eps.2477.

550 Fisher, B., R. K. Turner, and P. Morling. 2009. Defining and classifying ecosystem services for
551 decision making. *Ecological Economics* 68: 643–653.
552 doi:10.1016/j.ecolecon.2008.09.014.

553 Gerber, P. 1999. Processus de gentrification et demande sociale citadine. Exemple du centre-
554 ville de Strasbourg. *Revue Géographique de l'Est* 39.

555 Graça, M. S., J. F. Gonçalves, P. J. M. Alves, D. J. Nowak, R. Hoehn, A. Ellis, P. Farinha-
556 Marques, and M. Cunha. 2017. Assessing mismatches in ecosystem services proficiency
557 across the urban fabric of Porto (Portugal): The influence of structural and socioeconomic
558 variables. *Ecosystem Services* 23: 82–93. doi:10.1016/j.ecoser.2016.11.015.

559 Graça, M., P. Alves, J. Gonçalves, D. J. Nowak, R. Hoehn, P. Farinha-Marques, and M. Cunha.
560 2018. Assessing how green space types affect ecosystem services delivery in Porto,
561 Portugal. *Landscape and Urban Planning* 170: 195–208.
562 doi:10.1016/j.landurbplan.2017.10.007.

563 Greenberg, M., and M. Cidon. 1997. Broadening the definition of environmental equity: A
564 framework for states and local governments. *Population Research and Policy Review* 16:
565 397–413. doi:10.1023/A:1005702430607.

566 Grove, J. M., A. R. Troy, J. P. M. O’Neil-Dunne, W. R. Burch, M. L. Cadenasso, and S. T. A.
567 Pickett. 2006. Characterization of Households and its Implications for the Vegetation of
568 Urban Ecosystems. *Ecosystems* 9: 578–597. doi:10.1007/s10021-006-0116-z.

569 Havard, S., S. Deguen, J. Bodin, K. Louis, O. Laurent, and D. Bard. 2008. A small-area index
570 of socioeconomic deprivation to capture health inequalities in France. *Social Science &*
571 *Medicine* 67: 2007–2016. doi:10.1016/j.socscimed.2008.09.031.

572 Heynen, N., H. A. Perkins, and P. Roy. 2006. The Political Ecology of Uneven Urban Green
573 Space: The Impact of Political Economy on Race and Ethnicity in Producing
574 Environmental Inequality in Milwaukee. *Urban Affairs Review* 42: 3–25.
575 doi:10.1177/1078087406290729.

576 Hoffmann, E., H. Barros, and A. Ribeiro. 2017. Socioeconomic Inequalities in Green Space
577 Quality and Accessibility—Evidence from a Southern European City. *International*
578 *Journal of Environmental Research and Public Health* 14: 916.
579 doi:10.3390/ijerph14080916.

580 INSEE. 2016. Recensement de la population. 2013. Retrieved 1 January, 2019, from
581 <https://insee.fr/fr/accueil>

582 i-Tree Eco User's Manual. 2013. Retrieved 23 January, 2013, from
583 <https://www.itreetools.org/resources/manuals/EcoManualv5.pdf> :
584 <http://www.itreetools.org/resources/archives>.

585 Jenerette, G. D., S. L. Harlan, W. L. Stefanov, and C. A. Martin. 2011. Ecosystem services and
586 urban heat riskscape moderation: water, green spaces, and social inequality in Phoenix,
587 USA. *Ecol Appl* 21: 2637–2651.

588 Jensen, R., J. Gatrell, J. Boulton, and B. Harper. 2005. Using Remote Sensing and Geographic
589 Information Systems to Study Urban Quality of Life and Urban Forest Amenities. *Ecology*
590 *and Society* 9. doi:10.5751/ES-01201-090505.

591 Kremer, P., Z. Hamstead, D. Haase, T. McPhearson, N. Frantzeskaki, E. Andersson, N.
592 Kabisch, N. Larondelle, et al. 2016. Key insights for the future of urban ecosystem services
593 research. *Ecology and Society* 21.

594 Kruize, H., P. P. J. Driessen, P. Glasbergen, and K. van Egmond. 2007. Environmental Equity
595 and the Role of Public Policy: Experiences in the Rijnmond Region. *Environmental*
596 *Management* 40: 578–595. doi:10.1007/s00267-005-0378-9.

597 Kuras, E. R., P. S. Warren, J. A. Zinda, M. F. J. Aronson, S. Cilliers, M. A. Goddard, C. H.
598 Nilon, and R. Winkler. 2020. Urban socioeconomic inequality and biodiversity often
599 converge, but not always: A global meta-analysis. *Landscape and Urban Planning* 198:
600 103799. doi:10.1016/j.landurbplan.2020.103799.

601 Lakes, T., M. Brückner, and A. Krämer. 2014. Development of an environmental justice index
602 to determine socio-economic disparities of noise pollution and green space in residential
603 areas in {Berlin}. *Journal of Environmental Planning and Management* 57: 538–556.
604 doi:10.1080/09640568.2012.755461.

605 Landry, S. M., and J. Chakraborty. 2009. Street Trees and Equity: Evaluating the Spatial
606 Distribution of an Urban Amenity. *Environment and Planning A: Economy and Space* 41:
607 2651–2670. doi:10.1068/a41236.

608 Larondelle, N., and S. Lauf. 2016. Balancing demand and supply of multiple urban ecosystem
609 services on different spatial scales. *Ecosystem Services* 22: 18–31.
610 doi:10.1016/j.ecoser.2016.09.008.

611 Legendre, P., and L. Legendre. 1998. Numerical Ecology, Volume 24 - 2nd Edition.

612 Lovell, S. T., and J. R. Taylor. 2013. Supplying urban ecosystem services through
613 multifunctional green infrastructure in the United States. *Landscape Ecology* 28: 1447–
614 1463. doi:10.1007/s10980-013-9912-y.

615 Martinuzzi, S., O. M. Ramos-González, T. A. Muñoz-Erickson, D. H. Locke, A. E. Lugo, and
616 V. C. Radeloff. 2018. Vegetation cover in relation to socioeconomic factors in a tropical
617 city assessed from sub-meter resolution imagery. *Ecological Applications* 28: 681–693.
618 doi:10.1002/eap.1673.

619 McPhearson, T., P. Kremer, and Z. A. Hamstead. 2013. Mapping ecosystem services in New
620 York City: Applying a social–ecological approach in urban vacant land. *Ecosystem*
621 *Services* 5: 11–26. doi:10.1016/j.ecoser.2013.06.005.

622 Nowak, D.J., D.E. Crane, J.C. Stevens, R.E. Hoehn, J.T. Walton, and J. Bond. 2008. A ground-
623 based method of assessing urban forest structure and ecosystem services. *Arboriculture*
624 *and Urban Forestry* 34: 347–358.

625 Pham, T.-T.-H., P. Apparicio, A.-M. Séguin, S. Landry, and M. Gagnon. 2012. Spatial
626 distribution of vegetation in Montreal: An uneven distribution or environmental inequity?
627 *Landscape and Urban Planning* 107: 214–224. doi:10.1016/j.landurbplan.2012.06.002.

628 Pickett, S. T. A., M. L. Cadenasso, J. M. Grove, P. M. Groffman, L. E. Band, C. G. Boone, W.
629 R. Burch, C. S. B. Grimmond, et al. 2008. Beyond Urban Legends: An Emerging

630 Framework of Urban Ecology, as Illustrated by the Baltimore Ecosystem Study.
631 *BioScience* 58: 139–150. doi:10.1641/B580208.

632 Raddatz, L., and J. Mennis. 2013. Environmental Justice in Hamburg, Germany. *The*
633 *Professional Geographer* 65: 495–511. doi:10.1080/00330124.2012.700500.

634 Rangel, T. F. L. V. B., J. A. F. Diniz-Filho, and L. M. Bini. 2006. Towards an integrated
635 computational tool for spatial analysis in macroecology and biogeography. *Global*
636 *Ecology and Biogeography* 15: 321–327. doi:10.1111/j.1466-822X.2006.00237.x.

637 Richardson, E., J. Pearce, R. Mitchell, P. Day, and S. Kingham. 2010. The association between
638 green space and cause-specific mortality in urban New Zealand: an ecological analysis of
639 green space utility. *BMC Public Health* 10: 240. doi:10.1186/1471-2458-10-240.

640 Riley, C. B., and M. M. Gardiner. 2020. Examining the distributional equity of urban tree
641 canopy cover and ecosystem services across United {States} cities. *PLOS ONE* 15:
642 e0228499. doi:10.1371/journal.pone.0228499.

643 Schüle, S. A., L. K. Hilz, S. Dreger, and G. Bolte. 2019. Social Inequalities in Environmental
644 Resources of Green and Blue Spaces: A Review of Evidence in the WHO European
645 Region. *Int J Environ Res Public Health* 16. doi:10.3390/ijerph16071216.

646 Selmi, W., C. Weber, E. Rivière, N. Blond, L. Mehdi, and D. Nowak. 2016. Air pollution
647 removal by trees in public green spaces in Strasbourg city, France. *Urban Forestry and*
648 *Urban Greening* 17. doi:10.1016/j.ufug.2016.04.010.

649 Sievers-Glotzbach, S. 2013. Ecosystem Services and Distributive Justice: Considering Access
650 Rights to Ecosystem Services in Theories of Distributive Justice. *Ethics, Policy &*
651 *Environment* 16: 162–176. doi:10.1080/21550085.2013.801203.

652 Srang-iam, W. Planting Trees for Sustainability? A Climate Justice Perspective on Green
653 Agriculture in Thailand: 22.

- 654 Tratalos, J., R. A. Fuller, P. H. Warren, R. G. Davies, and K. J. Gaston. 2007. Urban form,
655 biodiversity potential and ecosystem services. *Landscape and Urban Planning* 83: 308–
656 317. doi:10.1016/j.landurbplan.2007.05.003.
- 657 Vyas, S., and L. Kumaranayake. 2006. Constructing socio-economic status indices: How to use
658 principal components analysis. *Health Policy and Planning* 21: 459–468.
- 659 Wilkerson, M. L., M. G. E. Mitchell, D. Shanahan, K. A. Wilson, C. D. Ives, C. E. Lovelock,
660 and J. R. Rhodes. 2018. The role of socio-economic factors in planning and managing
661 urban ecosystem services. *Ecosystem Services* 31: 102–110.
- 662 Yang, J., Y. Guan, J. (Cecilia) Xia, C. Jin, and X. Li. 2018. Spatiotemporal variation
663 characteristics of green space ecosystem service value at urban fringes: {A} case study on
664 Ganjingzi District in Dalian, China. *Science of The Total Environment* 639: 1453–1461.
665 doi:10.1016/j.scitotenv.2018.05.253.
- 666 Zhao, C., and H. A. Sander. 2015. Quantifying and Mapping the Supply of and Demand for
667 Carbon Storage and Sequestration Service from Urban Trees. *PLoS One* 10.
668 doi:10.1371/journal.pone.0136392.