Cooling cities through urban green infrastructure: a health impact assessment of European cities

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Summary

Background **High ambient temperatures are associated with many health effects, including premature mortality. The combination of global warming due to climate change and the expansion of the global built environment mean that the intensification of urban heat islands (UHIs) is expected, accompanied by adverse effects on population health. Urban green infrastructure can reduce local temperatures. We aimed to estimate the mortality burden that could be attributed to UHIs and the mortality burden that would be prevented by increasing urban tree coverage in 93 European cities.**

Methods **We did a quantitative health impact assessment for summer (June 1–Aug 31), 2015, of the effect of UHIs on all-cause mortality for adults aged 20 years or older in 93 European cities. We also estimated the temperature reductions that would result from increasing tree coverage to 30% for each city and estimated the number of deaths that could be potentially prevented as a result. We did all analyses at a high-resolution grid-cell level (250×250 m). We propagated uncertainties in input analyses by using Monte Carlo simulations to obtain point estimates and 95% CIs. We also did sensitivity analyses to test the robustness of our estimates.**

Findings **The population-weighted mean city temperature increase due to UHI effects was 1·5°C (SD 0·5; range 0·5–3·0). Overall, 6700 (95% CI 5254–8162) premature deaths could be attributable to the effects of UHIs (corresponding to around 4·33% [95% CI 3·37–5·28] of all summer deaths). We estimated that increasing tree coverage to 30% would cool cities by a mean of 0·4°C (SD 0·2; range 0·0–1·3). We also estimated that 2644 (95% CI 2444–2824) premature deaths could be prevented by increasing city tree coverage to 30%, corresponding to 1·84% (1·69–1·97) of all summer deaths.**

Interpretation **Our results showed the deleterious effects of UHIs on mortality and highlighted the health benefits of increasing tree coverage to cool urban environments, which would also result in more sustainable and climate-resilient cities.**

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Introduction

Many epidemiological studies have shown how extreme temperature affects human health and mortality. Exposure to high ambient temperatures has been associated with premature mortality,[1,2] cardiorespiratory morbidity,[3,4] hospital admissions,[5] and children's mortality and hospital admissions.[6] Temperature and mortality are related not only during periods of extreme heat, such as heat waves, but also when temperatures are moderately warm.[2,7] Small temperature increases at mild or moderate temperatures tend to occur more frequently, and therefore can have substantial health effects.[2,8,9]

The term urban heat island (UHI) refers to the higher temperatures of cities compared with surrounding areas. UHIs are one of the most striking climatic manifestations of urbanisation.[10] UHIs result from the anthropogenic modification of natural landscapes, such as changes in patterns of vegetation and bodies of water through fragmentation and conversion into impermeable surfaces.[11] Increased absorption and trapping of solar radiation in built-up urban fabrics, increasing population density, and the absence of green areas are the main factors that have been associated with the formation of UHIs.[12] UHIs might intensify the adverse effects of high temperatures on health, thereby increasing health risks in vulnerable populations.[13] In a study[14] done in the West Midlands, UK, UHIs were estimated to have contributed around 50% of the total heat-related mortality during a 2003 heat wave. Another study,[15] in which heat-related mortality was compared between the central and outer districts of Ho Chi Minh City, suggested that the attributable fraction resulting from the UHI was 0·42%.

Previous studies[1–3] have shown non-linear—typically U-shaped or J-shaped—associations between temperature and mortality. These associations vary substantially between populations due to differences in susceptibility, age distribution, access to resources, adaptability, and local public policies (eg, extreme-heat-warning systems,

See Online for appendix 1

Research in context

Evidence before this study

We did two different literature searches in PubMed, Scopus, and Google Scholar with no restrictions on language or publication date up to Feb 10, 2022. In the first, we searched for estimates of the effect of urban heat islands on health, and in the second we searched for health effects that could be avoided by increasing urban green infrastructure. Appendix 1 (p 4)contains a full list of search terms. In both cases, we considered only studies done in European cities. Our searches identified only a few relevant studies done in a small number of European cities. We found a large body of evidence based on time-series studies of the effects of suboptimal temperatures on mortality, but only a few studies of the mortality fraction attributable to urban heat islands (all of which focused on heat-wave events). We also found a few studies in which the potential of urban green interventions to prevent mortality was assessed, but again these studies focused on extreme-heat events.

Added value of this study

To our knowledge, this is the first study to estimate the mortality burden attributable to the effect of urban heat islands and the mortality that could be prevented by increasing tree coverage in European cities. Our health impact assessment covered 93 European cities at high resolution (250 × 250 m grid cell size)—an unprecedented magnitude. The spatially explicit analysis of urban heat exposure and its interaction with urban vegetation will inform future realistic city-specific scenarios that could help to mitigate adverse heat-related health effects.

Implications of all the available evidence

Our results showed that substantial mortality can be attributed to the effects of urban heat islands in European cities. Importantly, these effects could be considerably mitigated by increasing tree coverage to provide cooling in urban environments. This evidence will be valuable to policy makers aiming to introduce targeted green interventions to maximise population health benefits and promote sustainable, climate-resilient cities.

health-care system preparedness).[1] The modelling of such complex patterns requires a sophisticated statistical approach and the collection of large amounts of historical data.[2] Masselot[16] has provided mortality risk estimates for 801 European cities by age group that take into account a large list of city-level socioeconomic, climatic, and environmental characteristics. These estimates enable health impact assessments to be done of the effects of potential temperature variations—by using a comparative risk assessment approach, for example.

Comparative risk assessments enable assessment of the potential effects on population health that could result from shifting baseline exposure to an alternative, counterfactual exposure level.[17] This approach serves as a decision-making framework that provides robust and usable evidence about the implications of health-promoting scenarios that could be achieved through specific urban-planning strategies.[18] It can be applied at high level of spatial resolution and therefore can capture spatial variability, which means it can factor in important environmental-justice and health-equity implications.

Some planning and design strategies can mitigate urban heat: the introduction of green roofs or facades,[19–22] enhancement of the reflective properties (ie, albedo) of buildings by using light colours for roof and wall surfaces,[20,23] replacement of impervious surfaces with permeable or vegetated areas,[24–26] and increasing tree coverage.[27–30] Planting urban trees offers an important opportunity to mitigate high temperatures and, compared with other strategies, is relatively simple and cost-effective to implement.[28] Marando and colleagues[27] estimated the cooling capacity of trees in more than 600 European cities. The authors simulated the temperature difference between a baseline and a no-vegetation scenario by extrapolating the role of trees in mitigation of UHIs in different contexts. In their analysis, urban trees cooled European cities by an average of 1·1°C and up to 2·9°C. An evidence-based guideline[31] has recommended a goal of 30% tree coverage per neighbourhood for cooling, improving the microclimate, mitigating air and noise pollution, and improving mental and physical health. Many cities have already set a target of 30% tree coverage.[32–36] Furthermore, previous epidemiological studies have suggested health benefits associated with tree coverage of at least 30%, including reduced odds of incident psychological distress[37] and non-communicable diseases[38] such as diabetes, hypertension, and cardiovascular disease.

In view of global warming, increased urban sprawl, and increased development of natural lands, intensification of UHIs is expected.[6,39,40] Although the benefits of global mitigation strategies have been well discussed, the health benefits of improving local climate via urban planning are still unknown. Furthermore, compared with global efforts, some local actions to improve urban climate would be politically easier to implement and have short-term and medium-term benefits (eg, promotion of physical activity, improved perceived health, improved mental health, mitigation of air and noise pollution, heat mitigation, potential social cohesion).[41]

In this study we aimed to estimate the annual summer mortality burden that could be attributed to UHIs and the mortality burden that could be prevented by increasing tree coverage to 30%. Our ultimate goal is to inform local policy makers and decision makers about the benefits of strategically integrating urban green infrastructure into urban planning to promote sustainable, resilient, and healthy urban environments and contribute to climate change adaptation and mitigation.

Methods

Study design and data sources

We did a quantitative health impact assessment of 93 European cities. Cities and their boundaries were defined according to the Urban Audit 2018 dataset of Eurostat, a database that includes data for all European cities with more than 50 000 inhabitants (appendix 1 p 5). We selected the cities on the basis of the availability of Urban Climate (UrbClim) model temperature data.[42] The dataset includes 100 cities, six of which were not included in the Urban Audit dataset—specifically Belgrade, Novi Sad, Podgorica, Sarajevo, Skopje, and Tirana. We also excluded Reykjavík because of a lack of mortality risk estimates. Therefore, our analysis included the remaining 93 cities. Because the City of London is more of an economic centre rather than a residential area (ie, only 8200 inhabitants lived there as of 2015), we decided to include Greater London instead (appendix 1 p 5) to increase the coverage in terms of city size and population.

We retrieved demographic data by following the procedures described in previous health impact assessments of European cities.[43–45] Briefly, we retrieved total population counts for each city from the Global Human Settlement Layer (GHSL) for 2015,[46] which was the latest available population layer in a high resolution (ie, 250×250 m). To better represent population distribution, we used land-use data from the 2012 European Urban Atlas[47] to exclude non-residential areas (eg, industrial zones, port areas, bodies of water, airports, parks) from the baseline GHSL dataset. We reallocated the population from the removed grid cells among the dataset according to the GHSL population distribution to maintain total city population counts (appendix 1 pp 6, 8; appendix 2). We retrieved population age distribution data for 2015 from Eurostat at the Nomenclature of Territorial Units for Statistics (NUTS) level 3 (corresponding to metropolitan regions). We calculated the proportion of the population in each 5-year age group by NUTS3 and then estimated the population distribution by age group. We aggregated the groups as 20–44 years, 45–64 years, 65–74 years, 75–84 years, and 85 years or older to fit them with exposure–response functions (ERFs; appendix 1 p 7).

We retrieved weekly all-cause mortality counts by age group for 2015 from Eurostat (available for 81 cities at NUTS3 level). We estimated the daily mortality per age group per city, assuming the same distribution as the NUTS3 and a homogeneous distribution of deaths over the same week, and applied the rates to each grid cell. For cities without weekly deaths counts (appendix 1 p 7; appendix 2), we retrieved annual city-specific all-cause mortality counts for 2015 from Eurostat. We estimated mortality per age group and applied these estimates to each grid cell. We also retrieved monthly country mortality counts and estimated the proportion of deaths per month. We assumed a homogeneous distribution of deaths over the same month and estimated the daily deaths per grid cell.

The estimated daily mortality counts were correlated strongly between the two methods for the 81 cities for which data were available (Pearson correlation 0·98). However, annual city-specific mortality counts were overestimated by 17%. Therefore, we calibrated the dataset (appendix 1 p 8; appendix 2).

For **Eurostat** see https://ec.europa.eu/eurostat/data/database

We defined the baseline exposure to heat scenario as the daily mean temperature for the corresponding baseline 2015 tree coverage of each city. We retrieved daily mean temperatures from the UrbClim model for 93 cities (100×100 m resolution),[42] which combines large-scale meteorological data for surface, sea, precipitation, soil, and vertical profile and includes a description of the terrain with information about land use, vegetation (eg, the normalised difference vegetation index), and soil sealing. Temperature series were created by averaging the 100 m grid cells with centroids within the spatial boundaries of each 250 m grid cell.

Procedures

Our quantitative health impact assessment was done at the 250×250 m grid cell level for 2015 according to a comparative risk assessment approach.[43–45] We defined the summer as lasting from June 1 to Aug 31 in line with previous seasonality studies of temperature-attributable mortality.[48] In terms of temperature, 2015 was judged typical of the current European climate (appendix 1 p 9). We did two main analyses. In the first, we estimated the effect of exposure to UHIs on mortality by comparing the baseline temperature exposure with a counterfactual (non-realistic) exposure without an UHI effect. In the second analysis, we estimated the effect on mortality of increasing tree coverage to 30% as recommended (and the associated potential temperature reductions).

See **Online** for appendix 2

We retrieved city-specific and age group-specific ERFs from Masselot's work.[16] We estimated daily baseline temperature exposure levels and calculated the population attributable fraction for each daily mean and age group. We estimated the attributable premature mortality burden by combining the population attributable fraction and daily all-cause mortality data (appendix 1 p 10). We repeated the same procedure for each of the counterfactual scenarios and calculated the difference from the baseline scenario. The obtained result was the premature mortality burden attributed to shifting baseline exposure to the specific counterfactual exposure level scenario (ie, the effect of UHI or 30% tree coverage; appendix 1 p 11).

We summed the results by city and estimated the preventable age-standardised mortality per 100 000 population on the basis of the European standard population.[49] We also estimated the proportion of preventable all-cause deaths for both the year and specifically during the summer. Additionally, we calculated years of life lost as a result of these premature deaths (appendix 1 pp 10–11).

Potential sources of uncertainty considered in our main analyses were the ERFs and errors in UrbClim temperature data, the temperature adjustment model, the UHI data, and the cooling model. We constructed uncertainty distributions for each parameter and estimated the point estimates and 95% CIs, performing 500 Monte Carlo iterations by sampling from the built uncertainty range and considering all potential uncertainties simultaneously to calculate cumulative uncertainty. Finally, we ran Pearson correlations assessing the association between the outcomes from the UHI scenario and the 30% tree coverage scenario.

We retrieved the ERFs quantifying the association between temperature exposure and all-cause mortality by city and age group from Masselot's study,[16] which includes a comprehensive list of city-level characteristics (making the ERFs the best evidence available). Given that the risk estimates in Masselot's study were based on the ERA5-LAND temperature dataset (a re-analysis dataset that combines model data with observational data) with a resolution of approximately 9 km (therefore covering rural areas), we expected that the ERF temperature range would be lower than the UrbClim temperature. We thus applied a city-specific correction to the UrbClim dataset (appendix 1 pp 13–17).

We retrieved mean daytime and night-time summer data for UHIs (at 100×100 m resolution) from the Copernicus (the EU's Earth observation programme) UrbClim model application,[42] in which the effect of UHIs was estimated as the difference between the mean rural temperature (ie, represented by the rural classes of CORINE, an inventory of land cover in 44 classes) and each of the urban grid cells. We estimated 24-h daily mean UHI effect at 250×250 m resolution by averaging the daytime and night-time UHI data with centroids within the spatial boundaries of each 250 m grid cell (appendix 1 pp 18–19). We averaged the daytime and night-time data because the available ERFs considered 24 h of exposure to a daily mean temperature. For grids with negative values, we considered a null UHI effect (appendix 1 p 20).

We estimated the decrease in temperature—ie, cooling effect—associated with increasing tree coverage to 30% at a grid cell level. The Copernicus high-resolution layer Forest defines tree coverage as the vertical projection of tree crowns to a horizontal Earth's surface.[50] For each city, we analysed the feasibility of achieving this counterfactual by estimating the proportion of open space where trees could potentially be planted according to the corresponding land use. Overall, the mean difference between available open space and the 30% target at a grid-cell level was 2·9% (range 0·1–7·7), suggesting that 30% is reasonable target for European cities (appendix 1 pp 21–24). As an additional analysis, we constructed two other scenarios, one with a more attainable goal of 25% tree coverage (based on translations of WHO recommendations on access to green spaces in previous studies[51,52]) and the other with a more ambitious goal of 40% coverage (based on previous research[29] suggesting that 40% tree coverage was associated with significantly reduced daytime air temperatures).

We used linear regressions to build our models, following the approaches of Marando and colleagues[27] and Heris and colleagues,[53] who used machine learning techniques to establish that the best fitted models were linear regressions. Briefly, we retrieved Landsat-8 images[54] (30×30 m resolution) and estimated the median summer land surface temperature for each grid cell. Then, for each city, we developed a linear regression model with an ordinary least square algorithm trained by the land surface temperature dataset, Copernicus tree coverage data,[50] (appendix 1 p 21; appendix 2), and data for the amount of water evaporated from trees (500×500 m resolution), which is the sum of transpiration and vaporisation of intercepted rainfall from vegetation (based on the Penman-Monteith-Leuning canopy conductance model[55,56]), to estimate the effect of trees on surface temperature reduction at a grid-cell level:

$$\text{Land surface temperature} = \beta_{0e1} + \beta_{1e1}(\text{tree coverage}) + \beta_{2e1}(\text{water evaporated})$$

We then built a second ordinary least squares model, which was trained with an air temperature dataset for predicting the maximum air temperature as a function of land surface temperature and latitude. European weather stations had insufficient coverage to provide air temperature data, so we used a US air temperature dataset (appendix 1 p 24):

$$\text{Maximum air temperature} = \beta_{0e2} + \beta_{1e2}(\text{land surface temperature}) + \beta_{2e2}(\text{latitude})$$

We validated the model through a linear regression between the predicted values and the UrbClim values, with an adjusted R^2 equal to 0·66 and a percentage root mean square error of 2·03 (appendix 1 p 25). To estimate the land surface temperature corresponding to 30%, 40%, and 25% tree coverage, we estimated for each city the mean amount of water evaporated from trees considering the grid cells with 28–32%, 38–44%, and 23–27% tree coverage, respectively. We considered a range of values from plus or minus 2°C to avoid low counts.

Finally, we set the counterfactuals as 30%, 40%, and 25% tree coverage and estimated the respective land surface temperatures, which we then used to calculate the associated maximum air temperatures. We then calculated the difference between the baseline and counterfactual maximum air temperatures. This difference was the cooling that would be obtained at the different levels of tree coverage at grid-cell level. We used this temperature reduction as our counterfactual in the health impact assessment. The 16% of grids with negative cooling values were set to null (appendix 1 p 25). In addition, 3·6% of grid cells, covering 3·4% of the total population, were excluded from the analysis because of missing values for any of the parameters required for running the model. The error of the model has been estimated by calculating the propagated error of the two regressions for each city, as described by Marando and colleagues (appendix 1 p 26).[27] The mean city R^2 was 0·41 (range 0·07–0·79).

Sensitivity analyses

We did sensitivity analyses to assess the effects of changes in the input variables for the health impact assessment on the magnitude of our mortality estimates. We analysed for both scenarios (ie, the UHI effect and 30% tree coverage) the effects of using Martínez-Solanas and colleagues' ERFs,[57] which were available for 147 European regions (NUTS2) covering 66 cities. For the UHI scenario, we assessed the effects of using the adjusted and non-adjusted annual city-specific mortality datasets, the effect of using the mean grid summer UHI and the mean city summer UHI effect. For the 30% tree coverage scenario, we assessed the effects of using the mean city cooling. In addition, we did a sensitivity analysis of the cooling model by changing the estimated amount of water evaporated from trees at 30% coverage. We ran linear regression by city between tree coverage and evaporation data and predicted the amount of water evaporated from trees when coverage was 30%. We also ran regressions between tree coverage and evaporation data grouped by biome, because the amount of water

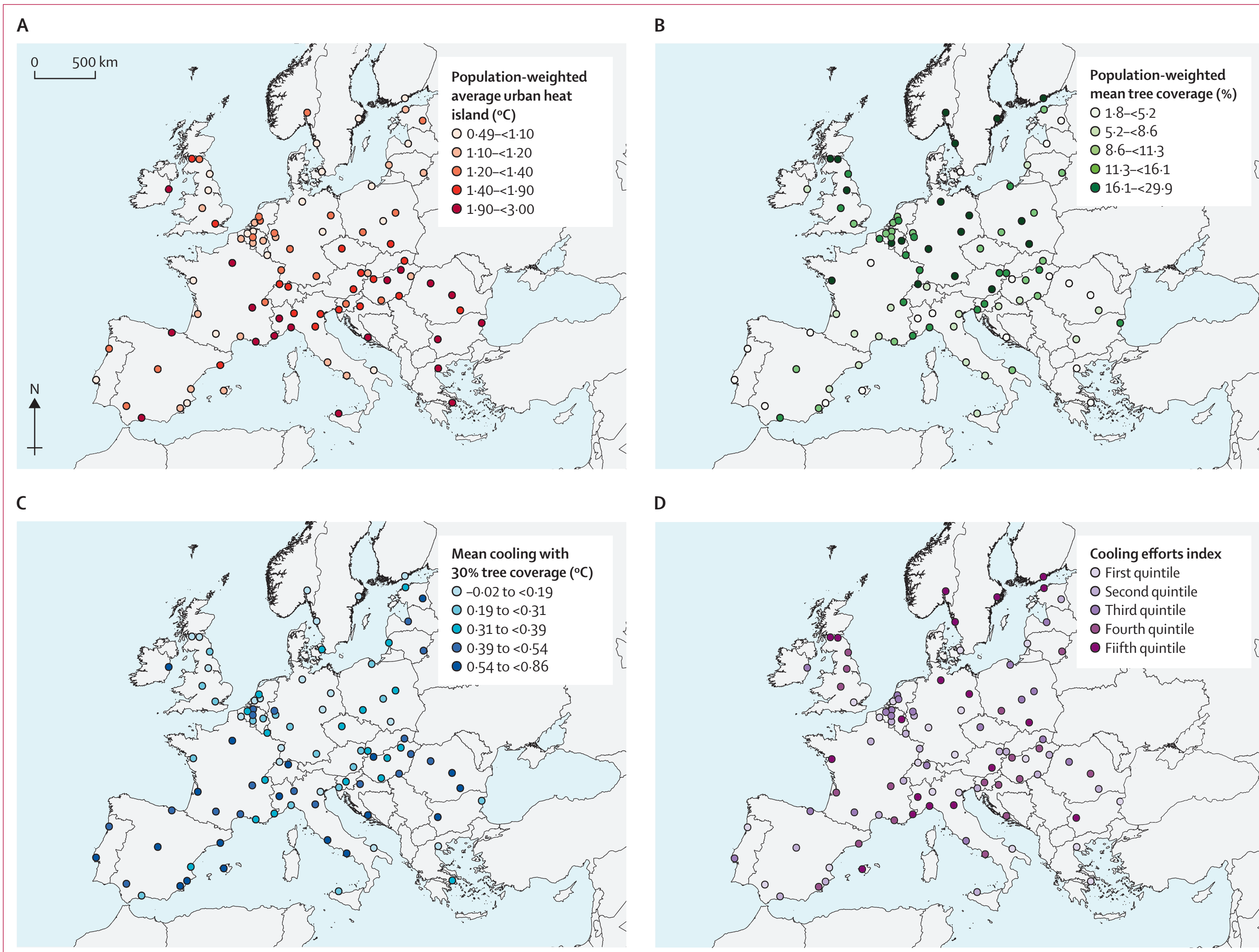

Figure 1: **Population-weighted mean urban heat island effect (A) and tree coverage (B), potential cooling capacity of 30% tree coverage (C), and cooling efforts index (D) in European cities**

evaporated from trees is associated with regional vegetation and climate.[55,58] In this way, we increased the counts and avoided poor adjustments. We assessed the effects on mean city cooling and on 30% tree coverage, and the corresponding effect on mortality (appendix 1 pp 27–34).

To understand the uncertainty contribution of each parameter to our CIs, we did an uncertainty analysis for six cities for both health impact assessment scenarios. We ran 500 Monte Carlo simulations that treated each parameter's uncertainty separately. We selected two cities with high mortality effects (ie, Barcelona and Budapest), two with moderate effects (ie, Munich and Lodz), and two with low effects (ie, Riga and Rotterdam; appendix 1 pp 35, 36).

We created an indicator of the tree coverage increment efforts needed to cool down cities, which is the ratio between the cooling effect of tree coverage at 30% and the mean increase in tree coverage needed to reach the target of 30% coverage, hereafter referred to as the cooling effort index. It can be interpreted as the cooling that would result per 1% increment of tree coverage.

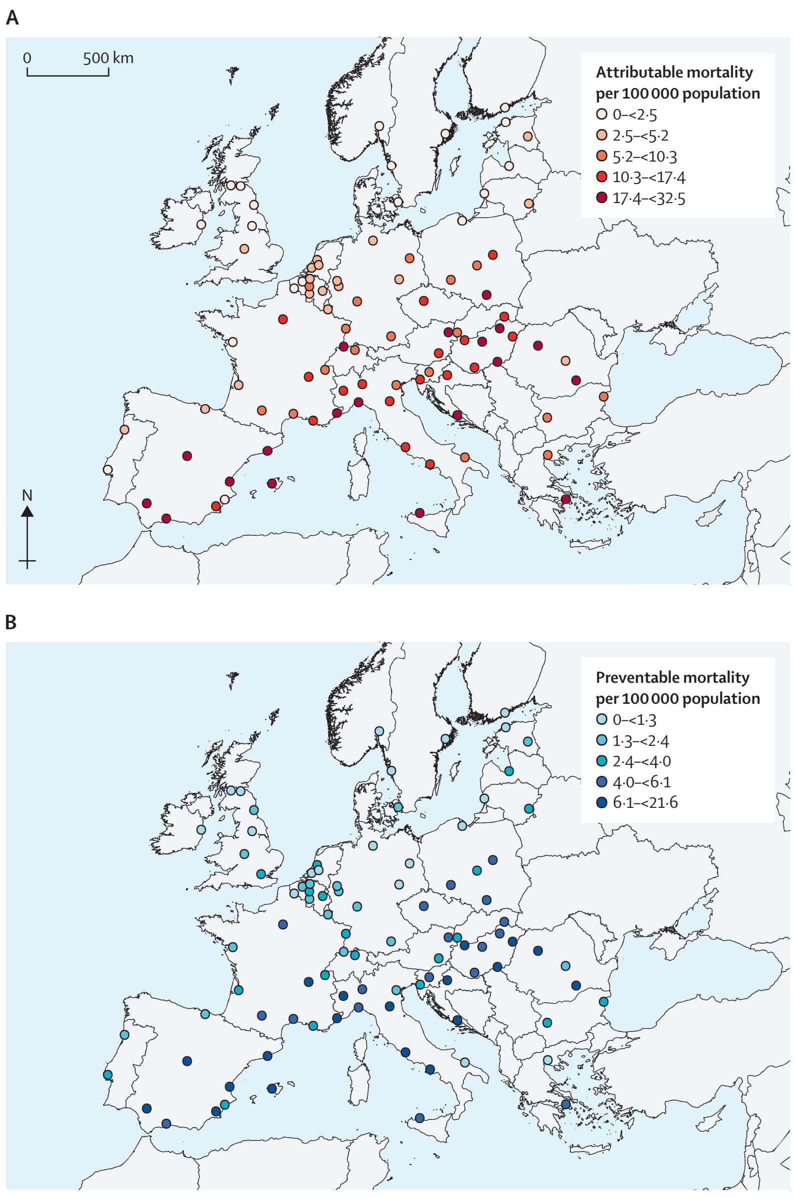

Figure 2: **Mean standardised mortality attributable to urban heat island effects (A) and mean standardised preventable mortality assocaited with increasing tree coverage to 30% (B) per 100 000 inhabitants in European cities**

Role of the funding source

The funders of the study had no role in study design, data collection, data analysis, data interpretation, or writing of the report.

Results

Overall, 57 896 852 inhabitants aged 20 years or older resided in the 93 studied cities in 2015. Population density ranged from 4274 inhabitants per km² (Murcia) to 21 462 per km² (Paris; appendix 2). 555 215 deaths from all causes were reported in 2015, 128 269 (23·1%) of which occurred in summer. Overall, mean summer temperatures ranged from 14·2°C (SD 2·3) in Glasgow to 29·7°C (3·3) in Seville, with mean maximum temperatures ranging from 22·7°C (SD 2·5) in Tallinn to 36·8°C (3·3) in Seville. The mean daily UHI effect during the summer was 1·3°C (0·5), with a daytime mean of 0·6°C (0·4) and a night-time mean of 1·9°C (0·8). The population-weighted city-mean daily UHI effect in summer was 1·5°C (0·5; city range 0·5—3·0; figure 1A), with the highest summer mean grid-cell value recorded in Cluj-Napoca (4·1°C; appendix 2).

The mean city tree coverage was 14·9% (SD 13·9); range 2·1–34·6), whereas the city population-weighted mean was 10·9% (6·1; 1·8–29·9; figure 1B). We estimated that increasing tree coverage to 30% at 250 m resolution would result in a mean city cooling of 0·4°C (0·2; 0·0–1·3; figure 1C), with a maximum grid-cell value of 5·9°C (appendix 2). Increasing the tree coverage to 30% at a grid-cell level would lead to an average increase of 17·7% (range 3·8–28·8; appendix 2).

Across all examined cities, 57 089 394 (78%) of 73 082 044 people (ie, the total population) lived in areas with a mean summer UHI effect greater than 1°C, and 14 491 628 (20%) lived with a mean summer UHI effect greater than 2°C. Overall, 6700 (95% CI 5254–8162) premature deaths could be attributed to UHI effects during the summer months, corresponding to 4·33% (3·37–5·28) of summer mortality (figure 2A; table 1). 2644 (95% CI 2444–2824) premature deaths could be prevented by increasing tree coverage to 30%, which equates to a 1·84% (1·69–1·97) reduction in summer mortality (figure 2B; table 2). Thus roughly 39·5% of the deaths attributable to UHI effects could be prevented by increasing tree coverage to 30%.

Attributable mortality burden varied greatly among cities: per 100 000 age-standardised inhabitants, UHI was associated with no premature deaths in Gothenburg, compared with 32 in Cluj-Napoca (figure 2A; table 3). The overall mean was roughly ten deaths per 100 000 (table 1). Similarly, we estimated that whereas increasing tree coverage to 30% would prevent no deaths per 100 000 age-standardised inhabitants in Oslo, it could prevent 22 in Palma de Mallorca (figure 2B; table 4).

Overall, the cities with the highest mortality attributable to UHI effects were in southern and eastern Europe, particularly in Spain, Italy, Hungary, Croatia, and Romania, whereas those with the lowest mortality attributable to UHI effects were mainly located in northern Europe including Sweden, Estonia, UK, and northern France (figure 2A; table 3). A similar pattern was noted for mortality that could be prevented by increasing tree coverage (figure 2B, table 4). The number of deaths attributable to UHI effects and the number of preventable deaths associated with increasing tree coverage to 30% were strongly linearly correlated (r=0·89). The number of deaths attributable to UHI effects was also correlated with preventable mortality (r=0·75), the proportion of annual preventable mortality (r=0·73), and the preventable years of life lost (r=0·89) (appendix 1 p 12).

Sensitivity analyses suggested that the largest variations in the final estimates of UHI effects were due to use of the non-adjusted city-specific annual mortality dataset

	Summer attributable deaths (95% CI)	Summer attributable age-standardised mortality per 100 000 inhabitants (95% CI)	Contribution of summer attributable deaths to deaths (95% CI)	Effect of summer attributable deaths on annual deaths (95% CI)	Years of life lost per 100 000 inhabitants (95% CI)	Change (%)
Main analysis (daily UHI effect per grid cell)	6700 (5254–8162)	9·91 (7·71–12·07)	4·33% (3·37–5·28)	0·90% (0·67–1·11)	166·42 (128·47–201·98)	Reference
Mean summer UHI effect per grid cell	6854 (6196–7494)	10·10 (9·08–11·00)	4·42% (3·98–4·82)	0·90% (0·76–0·99)	169·78 (148·98–185·44)	2%
Mean UHI effect per city	5478 (0–11 742)	8·08 (0·00–17·45)	3·51% (0·00–7·68)	0·72% (0·00–1·66)	135·90 (0·00–288·61)	–18%
Adjusted annual city mortality dataset	6933 (5434–8483)	10·09 (7·80–12·33)	4·46% (3·43–5·48)	0·93% (0·68–1·15)	142·68 (111·95–171·82)	3%
Non-adjusted annual city mortality dataset	8061 (6319–9864)	11·73 (9·08–14·33)	5·19% (3·99–6·37)	0·86% (0·65–1·04)	165·91 (130·18–199·79)	20%
Martinez-Solanas et al (2021) as exposure–response function*	4401 (3779–5056)	10·18 (8·75–11·65)	4·86% (4·18–5·56)	1·17% (0·99–1·37)	185·76 (159·65–211·70)	–17%

The exposure–response function for these analyses is based on work by Masselot,[16] unless otherwise specified. UHI=urban heat island. *For the 66 cities covered in this paper.

Table 1: **Sensitivity analyses of the health impact assessment of UHI effects**

	Preventable summer deaths (95% CI)	Summer age-standardised mortality prevented per 100 000 inhabitants (95% CI)	Contribution of preventable summer deaths to deaths (95% CI)	Effect of summer preventable deaths on annual deaths (95% CI)	Years of life gained per 100 000 inhabitants (95% CI)	Change (%)
Main analysis (daily effect per grid cell)	2644 (2444–2824)	4·17 (3·83–4·49)	1·84% (1·69–1·97)	0·37% (0·32–0·41)	69·85 (62·36–75·67)	Reference
Mean cooling effect per city	2148 (792–3472)	3·21 (0·77–5·54)	1·42% (0·38–2·43)	0·29% (0·06–0·50)	54·06 (15·05–91·53)	–19%
Martinez-Solanas et al (2021) as exposure–response function*	1694 (1580–1811)	3·96 (3·68–4·23)	1·87% (1·74–2·00)	0·46% (0·42–0·50)	72·49 (67·56–77·59)	–21%
Linear regression by city for different estimated water evaporated from trees	2667 (2466–2861)	4·19 (3·86–4·51)	1·84% (1·70–1·98)	0·37% (0·32–0·40)	70·24 (62·91–76·05)	1%
Linear regression by biome for different estimated water evaporated from trees	2687 (2477–2888)	4·21 (3·85–4·54)	1·87% (1·72–2·02)	0·38% (0·32–0·41)	71·17 (63·30–76·90)	3%
Additional analyses						
Counterfactual scenario with 25% tree coverage	2092 (1933–2241)	3·32 (3·05–3·58)	1·46% (1·34–1·57)	0·29% (0·25–0·32)	55·62 (49·46–60·18)	–21%
Counterfactual scenario with 40% tree coverage	3727 (3462–3992)	5·83 (5·38–6·26)	2·58% (2·38–2·76)	0·51% (0·44–0·56)	97·85 (87·77–105·99)	41%

The exposure–response function for these analyses is based on work by Masselot,[16] unless otherwise specified. *For the 66 cities covered in this paper.

Table 2: **Sensitivity analyses of health impact assessment of tree coverage effects**

	Mean summer temperature (°C)	Mean urban heat island effect (°C)	Population-weighted mean urban heat island effect (°C)	Proportion of population exposed to more than 1°C from urban heat islands	Summer attributable deaths (95% CI)	Attributable age-standardised mortality per 100 000 inhabitants (95% CI)	Effect of summer attributable deaths on annual deaths (95% CI)
Stockholm	16·68	0·34	0·49	0·11	0·00 (–10·00 to 8·72)	0·00 (–1·73 to 1·48)	0·00% (–0·84 to 0·73)
Gothenburg	15·93	0·44	0·63	6·84	0·00 (–4·03 to 2·69)	0·00 (–0·88 to 0·59)	0·00% (–0·47 to 0·32)
Newcastle	15·13	0·72	0·78	23·54	0·89 (–2·51 to 4·72)	0·38 (–1·11 to 2·05)	0·16% (–0·46 to 0·86)
Leeds	15·08	0·42	0·63	14·29	3·32 (–6·23 to 13·92)	0·63 (–1·31 to 2·76)	0·28% (–0·53 to 1·19)
Tallinn	16·38	0·95	1·11	75·19	2·13 (–0·56 to 4·30)	0·73 (–0·17 to 1·44)	0·29% (–0·08 to 0·60)
Cluj-Napoca	23·09	2·43	3·00	95·67	71·12 (65·49 to 77·05)	32·49 (29·89 to 35·14)	10·36% (9·54 to 11·23)
Málaga	27·75	1·91	2·42	98·76	112·69 (100·53 to 124·59)	27·29 (24·32 to 30·20)	12·39% (11·05 to 13·70)
Barcelona	25·82	1·09	1·47	76·70	362·96 (312·73 to 405·94)	26·69 (22·91 to 30·02)	14·82% (12·77 to 16·58)
Budapest	24·82	1·60	1·90	93·95	378·10 (316·06 to 425·43)	25·71 (21·34 to 28·92)	8·77% (7·33 to 9·86)
Palma de Mallorca	27·06	0·88	1·17	73·21	69·50 (57·37 to 81·00)	23·87 (19·57 to 27·94)	11·99% (9·90 to 13·97)

The ten cities associated with the lowest and highest impacts on attributable mortality are displayed.

Table 3: **Main health impact assessment results for the effect of urban heat islands in ten European cities**

	Tree coverage (%)	Population-weighted tree coverage (%)	Tree coverage increment (%)	Mean cooling (°C)	Maximum cooling (°C)	Summer preventable deaths (95% CI)	Annual preventable age-standardised mortality per 100 000 inhabitants (95% CI)	Effect of summer preventable deaths on annual deaths (95% CI)
Oslo	34·62	29·42	3·76	0·10	0·81	0·01 (–0·56 to 0·67)	0·00 (–0·15 to 0·17)	0·00% (–0·07 to 0·09)
Bari	15·83	8·99	14·08	–0·02	0·47	0·26 (0·01 to 0·45)	0·09 (0·01 to 0·16)	0·05% (0·00 to 0·09)
Glasgow	19·02	17·29	11·97	0·04	0·24	0·61 (0·42 to 0·77)	0·15 (0·11 to 0·19)	0·05% (0·03 to 0·06)
Lille	12·97	15·26	16·11	0·01	0·22	0·90 (0·72 to 1·08)	0·17 (0·14 to 0·20)	0·07% (0·06 to 0·09)
Edinburgh	25·36	25·48	5·40	0·02	0·33	0·62 (0·43 to 0·80)	0·18 (0·12 to 0·23)	0·08% (0·05 to 0·10)
Palma de Mallorca	8·03	5·15	23·03	0·68	1·04	62·56 (61·31 to 63·72)	21·60 (21·19 to 22·00)	1·95% (1·91 to 1·99)
Barcelona	8·41	5·39	23·31	0·70	0·89	214·52 (205·60 to 220·98)	15·84 (15·16 to 16·33)	1·69% (1·62 to 1·74)
Split	5·40	1·79	25·93	0·79	1·04	14·72 (13·95 to 15·38)	12·44 (11·80 to 12·99)	0·71% (0·67 to 0·74)
Naples	13·05	6·37	19·67	0·64	1·00	75·77 (72·14 to 79·34)	11·28 (10·72 to 11·81)	0·98% (0·93 to 1·02)
Murcia	10·31	8·85	20·83	0·66	1·25	29·85 (29·04 to 30·60)	10·60 (10·31 to 10·86)	0·96% (0·93 to 0·98)

The ten cities associated with the lowest and highest impacts on preventable mortality are displayed.

Table 4: **Main health impact assessment results for the 30% tree coverage scenario in ten European cities**

(which was associated with an increase in summer attributable deaths of roughly 20%), using the mean city UHI effect (associated with an 18% decrease in summer attributable deaths), and using Martínez-Solanas and colleagues' work as the ERF source (associated with a 17% decrease; table 1).

The ambitious 40% tree coverage counterfactual scenario would lead to a mean city cooling of 0·5°C (SD 0·3), resulting in a 41% increase in the number of deaths that could be prevented compared with the 30% coverage scenario, whereas the 25% tree coverage counterfactual would lead to mean city cooling of 0·3°C (SD 0·2) and a 21% decrease in the number of deaths that would be prevented (table 2). Table 2 details the results of other sensitivity analyses. Overall, the results of these analyses strongly correlated with our main findings, suggesting that our results were highly robust (appendix 1 pp 27–34).

Uncertainty analyses of UHI scenarios suggested that UHIs were the primary contributors of uncertainty, followed by baseline temperature, ERF, and the temperature adjustment to the ERA5 dataset. For the 30% tree coverage scenario, baseline temperature was the primary source of uncertainty, followed by ERFs, the cooling model, and temperature adjustment to ERA5 (appendix 1 pp 35, 36).

Cities with high cooling effort index scores were mainly located in northern Europe (eg, Oslo, Edinburgh, Gothenburg, Tallinn) but also included Sofia, Liège, Krakow, Graz, Nantes, and some cities in northern Italy (eg, Turin, Bologna, Genoa; figure 1). Cities with low scores were mostly in southern Europe (eg, Athens, Thessaloniki, Bari, Varna, Valencia, Porto), but were also dispersed across central Europe (eg, Zurich, Padua, Milan, Leipzig, Munich; figure 1).

Discussion

To our knowledge, our study is the first to estimate the mortality burden attributable to the effect of UHIs and mortality that could be prevented by increasing tree coverage in European cities. Our results show that 6700 deaths could be attributed to the effect of UHIs in 2015, and that 2644 (40%) of these deaths could have been avoided if there was 30% tree coverage at a grid-cell level.

Our results align with those of previous studies in which the cooling obtained from urban green infrastructure strategies was estimated. Kalkstein and Sheridan[59] estimated that a 10% increase in tree coverage could reduce urban temperatures in Philadelphia (PA, USA) by 0·22°C. In a study[22] of New York (NY, USA), it was estimated that a potential reduction of 0·6°C at 1500 h could result if 31% of the city area were covered with trees and green roofs. In addition, a 2021 systematic review[60] of cooling modelling showed that street trees can reduce urban air temperature by an average of 0·3°C for each 10% increase in coverage. We estimated that a mean increase in tree coverage of 17·7% (ie, to tree coverage of 30%) would cool European cities by 0·4°C. In 2021, Marando and colleagues[27] estimated that temperatures could be reduced by 1°C in European functional urban areas if tree coverage was increased by 16%. Our estimates are notably lower than Marando and colleagues', even though we used similar methods. The difference can probably be explained by the area of scope of the study. Our model is based on the city level, whereas theirs focused on functional urban areas, which are constituted by a core city and its commuting zone (often including green areas, such as peri-urban forests). Focusing on cities has two main consequences, particularly with regard to the amount of water evaporated from trees. First, because this layer has coarse spatial resolution (500×500 m) it might not capture spatial heterogeneity well at city level, especially scattered trees. Second, trees in highly urbanised settings have different rates of transpiration from those in peri-urban areas,[61] which might explain the smaller reduction in temperature calculated in our study. Urban trees are often exposed to harsh conditions (eg, paved soils, air pollution) that can limit transpiration—and, therefore, their cooling capacity.[62] However, it should be noted that the cooling effect of street trees, although small, is important for alleviation of the UHI effect in highly urbanised areas.[63]

Most of the cities with high UHI effects were also among the most densely populated (eg, Paris, Thessaloniki, Athens, Bilbao, Brussels), with population densities of 7272–21 462 inhabitants per km². The association between population density and UHI effects has been well described in previous studies.[10,12] Furthermore, these cities also had low tree coverage, which suggests an opportunity to improve urban microclimates by increasing urban tree coverage. However, the formation of UHIs is complex and is associated with many factors. Furthermore, various drivers of UHI effects function differently during the day and the night. Although vegetation is the dominant factor influencing the intensity of UHI effects during the day, the urban canyon (ie, the geometry formed by a city street and its flanking buildings) more strongly determines UHI effects at night.[64] Additionally, the night-time intensity of the UHI effect is on average three times the daytime intensity (ie, 0·6°C and 1·9°C, respectively). Therefore, urban green infrastructure strategies need to be accompanied by other interventions—especially those that reduce night-time UHI effects—to achieve health benefits, such as changing ground surface materials (eg, from asphalt to granite) and structural interventions that change the sky view factor (ie, the fraction of visible sky relative to street geometry and building density).[64] Our results, together with those of other studies,[65,66] suggest that increases in tree coverage should be combined with other interventions to produce larger temperature reductions, thereby having greater beneficial effects on health—particularly for cities with low cooling capacity, where increasing tree coverage would not reduce the temperature substantially.

Just as the characteristics of UHIs are specific to each city, so too is the tree coverage cooling capacity of each city. Our cooling estimates were affected not only by tree coverage cooling capacity, but also by baseline tree coverage. If the cooling capacity was already high and baseline tree coverage already close to 30%, the potential for reducing temperatures through urban green infrastructure was low. In turn, if both the vegetation cooling capacity and tree coverage were low, the potential for cooling might be higher than expected. To improve the interpretability of our results, we built the cooling efforts index. Notably, most cities with high scores on the index had low mortality attributable to UHI effects (eg, Glasgow, Edinburgh, Oslo, Gothenburg, Tallinn, Helsinki). Conversely, several Mediterranean cities with low cooling index scores had greater mortality attributable to UHI effects (eg, Athens, Valencia, Sevilla, Palermo, Málaga, Madrid). These findings imply that greater efforts are required for these cities to achieve temperature reduction because of the combination of low baseline tree coverage and low tree coverage cooling capacity.

Some of the cities in semi-arid conditions had low or even negative UHI effects. However, these effects are not the result of effective urban planning. In dry regions, rural land surfaces can be warmer than urban areas, particularly if the vegetation is not irrigated.[67,68] Droughts can limit evapotranspiration.[61] In such regions, urban centres with tall buildings can provide shading, further reducing the temperature compared with nearby rural areas.[69] Despite relatively low-intensity UHI effects, some cities (eg, Palma de Mallorca, Alicante, Porto, Rome, and Naples) had associated high attributable mortality. One possible explanation is the already-high baseline temperature in these cities, resulting in a baseline increased risk for the population, combined with the specific association between exposure to heat and mortality. For this reason, UHI should not be used as an indicator of excess heat in these cases. However, actions to mitigate general high temperatures would still be needed to reduce the associated mortality. In these settings, urban green infrastructure can have an increased cooling effect if urban irrigation is used.[55,70] Therefore, tree coverage cooling capacity could be increased and would constitute a partial solution for excessive heat. A drawback of this approach is that urban irrigation might cause water scarcity, which could be exacerbated as a result of climate change.[71]

Cost is also a consideration, given that tree maintenance is more expensive in dry conditions.[72] Therefore, local policy makers and decision makers need to consider the complete range of costs and benefits. Despite the overall positive balance of evidence from individual studies assessing the cost–benefit ratio of urban trees, there is no general consensus because of high variation in values, methodological differences, and the low number of studies.[73] Economic valuation is important to justify investment in urban tree planting, and more studies are needed on this topic.[73] Furthermore, economic valuation should also incorporate health and social effects into the decision-making framework, which would probably increase the economic benefits associated with urban trees.

Urban trees provide substantial public health and environmental benefits. However, some factors should be considered to maximise their potential. First, the population-weighted mean city tree coverage was 22% lower than the unweighted mean coverage, meaning that the most populated areas have less tree coverage. In addition, previous studies[74] have shown that urban trees are often unevenly distributed across the population, and that socioeconomically disadvantaged groups might be deprived of environmental benefits, constituting a form of environmental injustice. This is a reason why the intervention is proposed at a small scale (ie, on a girid-cell level), which enabled us to consider urban tree distribution in addition to total coverage. Nonetheless, we acknowledge that it is not always possible to meet the target at the grid-cell level, and depending on the urban design, the scale of the intervention should vary. Second, planting groups of trees in green areas (eg, parks, squares, community gardens) or in central tree-lined gardens with permeable surfaces rather than isolated street trees might result in synergic positive effects, improving not only the trees' cooling capacity but also the quality and aesthetic of green spaces, thereby maximising the population health benefits.[75]

Our sensitivity analyses showed that the effect sizes for both UHI effects and tree coverage were greater when used Martínez-Solanas and colleagues' ERFs,[56] which were modelled on the basis of a broader level of aggregation (ie, NUTS3), the entire population, and a different temperature dataset. We used age-specific and city-specific ERFs in our main analysis,[16] which better reflect the population's adaptability to ambient temperature. This is particularly important in view of evidence showing differential susceptibility associated with different age groups (ie, older adults and children have increased risk of dying or becoming ill at increased temperatures).[9] In addition, the ERFs also account for some socioeconomic variables, which is crucial considering that vulnerable subpopulations are at increased risk of adverse health effects due to high temperatures.[9] Nonetheless, we should note that we applied the same ERF across each entire city, whereas socioeconomic inequalities are often highly pronounced within each city population.[76]

Using the city mean substantially reduced the attributable mortality estimates, a reduction that did not occur when we used the summer grid-cell mean UHI. This finding shows that ignoring the spatial variability of UHIs leads to an underestimation of the real effects, given that often the most densely populated areas also have the highest-intensity UHI effects,[10] which is also reflected by the mean 41% increase in UHI effects when weighted by population. A similar outcome was obtained when we used mean city cooling instead of the grid-cell level, emphasising the importance of accounting for cooling spatial heterogeneity. In such a context, our analysis aims to provide spatial information about the areas that would benefit the most from targeted green interventions to reduce temperatures and ameliorate the living conditions of urban dwellers.

The results of our analysis of alternative scenarios with both a higher and a lower target tree coverage proportion suggested a linear association tree coverage and cooling effects, which facilitates planning of urban green infrastructure, considering that the feasibility of the intervention can thus be adapted to each local setting. For cities with little available open public space, the 30% tree coverage target will be very challenging to achieve. Tree planting programmes will need to target privately owned industrial, commercial, or institutional spaces. We encourage city planners to choose a 30% tree coverage target, but a 25% target could be set for compact cities facing space difficulties. This more achievable target could be combined with other strategies beyond tree planting, such as installing green roofs to reduce local temperature.

The main strengths of our study include the use of a fine spatial scale of 250 m covering 93 European cities, enabling the generation of high-resolution maps that can be used for identifying where interventions are most urgently needed, the use of city-specific and age-specific ERFs, the analysis of effects attributable to UHIs done on a daily basis, and the building of a realistic city-specific counterfactual scenario that can partly mitigate UHI effects. Another strength is the substantial number of sensitivity analyses we did, and the high correlation between these results and those of our main analyses, which suggests that our results are robust.

Nonetheless, our study also has several limitations that need to be addressed. First, population data were only available for 2015, which is why we could not do the analysis for a more recent year. Also, mortality data were available at NUTS3 level and on a weekly basis, and age structure data were available at a city level, which made the analysis less sensitive to within-city variability and also ignored potential weekend effects (ie, greater mortality at the weekend than during the week).[77] Furthermore, we were not able to build the uncertainty ranges for both population counts and mortality because of a lack of reported errors in the published data, resulting in narrower CIs. However, we were still able to consider the exposure spatial variability and uncertainty in both main analyses.

We acknowledge that our study applies specifically to summer, 2015, meaning that the exact mortality estimations are only attributable for the reference year. However, similar or greater mortality effect could be expected given that 2015 had summer temperatures similar to other years and that global warming and the intensification of UHIs might increase effects on health due to heat stress.[40,56] Our ultimate goal was to generate a broad idea of the health benefits that could be achieved through investing in urban green infrastructure.

We based our analyses on resident population exposure and did not consider commuters for work or study, which could have led to misclassification of exposure. Nonetheless, as we showed, night-time UHI effects are considerably greater than daytime effects, and therefore we postulate that this limitation might not have substantially affected our mortality estimates.

Several limitations are associated with the cooling model we applied. First, we used a US dataset to build a predictive model of the relationship between surface temperature and air temperature in European cities. A European dataset would have been more appropriate, but the US one was the best option available in view of the insufficient coverage of the European weather station network and the wide range of variables covered by the dataset. Furthermore, our model proved reliable for comparisons of the estimated average temperature with the UrbClim temperature. A second limitation is the weak adjustment the cooling model had for some cities, which could also reflect the weak association between tree coverage and ambient temperature. However, at the same time the model enabled us to predict air temperature reductions in a simple, straightforward, and scalable manner across a wide spatial area. Additionally, tree coverage cooling capacity might depend on other variables that were not considered in the model, such as type of trees planted (eg, in terms of leaf size and shape,[78,79] height, and crown width[80]). We also acknowledge that we did not account for the uncertainties associated with each model input, specifically the data for the amount of water evaporated by trees, which were obtained from another model.[54,55] A further source of uncertainties was the estimation of the amount of water evaporated from trees in grid cells with 30% tree coverage. Although probably none of the methods used can accurately estimate evapotranspiration when tree coverage is 30%, we did sensitivity analyses that detected no significant differences between the methods used in its estimation. Despite the limitations of the cooling model, our coarse-grained approach provides a first-order guideline on expected cooling effects that is valid across the European region and that can be adjusted to specific city-settings.

We focused on analysing the effect on health of high temperature, yet we need to note the potential role that UHIs have in mitigation of low temperatures,[81] particularly in view of the greater health effects of cold compared with heat in the European region.[2,16,57] Nonetheless, under global warming both maximum monthly temperatures and average monthly temperatures are projected to rise. Therefore, health effects attributable to heat are projected to exceed those attributable to cold in the future under high-emission scenarios.[57]

Finally, despite achieving a small temperature reduction with our proposed urban green infrastructure intervention, the cooling produced could prevent a considerable number of premature deaths. Here, we estimated only the preventable mortality associated with temperature reduction, but the full extent of the health benefits associated with urban greening should not be assessed on the basis of air cooling alone. A previous health impact assessment study by Pereira Barboza and colleagues[44] estimated that 20 deaths per 100 000 inhabitants could be prevented annually if European cities complied with WHO recommendations with regard to access to green space (ie, all people should live within 300 m of a green space, with the normalised difference vegetation index used as a proxy of greenness). Although we used a different indicator, undoubtedly, our study and Pereira Barboza and colleagues' one complement each other and suggest an urgent need to green cities for health. Urban greening also mitigates air and noise pollution,[82–84] provides biodiversity, promotes physical activity,[75] and has direct effects on physical and mental health.[75,85] Further studies of all the benefits of incorporating urban green infrastructure in urban areas are necessary to show the full potential to improve environmental quality and make cities healthier, more sustainable, and more resilient in the face of climate change.

Our results showed large effects on mortality associated with UHI effects in European cities, which could be partly reduced by increasing tree coverage. We encourage city planners and decision makers to incorporate urban green infrastructure adapted to local settings in combination with other interventions to maximise health benefits and promote more sustainable cities.

Contributors

MN conceived the study. TI and MC designed the study and collected data. TI did the data analysis. TI, MC, FM, EPB, SK, MQ-Z, and MN interpreted the data. NM and MT provided input on the health impact assessment methods. FM, MH, and MC developed the cooling model. PM and AG provided the age group-specific ERFs. JU and MQ-Z contributed to statistical analysis and data management. TI wrote the Article, which was reviewed by all authors. TI, SK, MC, and EPB had access to and verified all study data. All authors provided feedback on study design, data analysis, and interpretation of results.

Declaration of interests

We declare no competing interests.

Data sharing

All the data in this study are routinely collected and contain no information about specific people. Our data are available upon request to the corresponding author, subject to the agreement of the trial steering group.

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