GreenCityLabHuế

Deliverable D3

Land-use change and nature-based solutions impact assessment

Impact assessment, supply and demand modelling

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The joint research project GreenCityLabHuế – Strengthening climate resilience of urban regions in Central Vietnam through nature-based solutions for heat adaptation and air quality improvement, funded by Federal Ministry of Education and Research (BMBF) as part of the funding measure „Sustainable Development of Urban Regions" within the framework of the Strategy „Research for Sustainability" (FONA), started its Research and Development (R&D) phase in April 2021, following completion of the preceding definition phase. In the R&D-phase the project aims to strengthen the climate resilience of the city of Huế (Thừa Thiên Huế Province, Central Vietnam) through nature-based solutions (NBS) with a focus on heat adaptation and air quality improvement. To this end, a multidisciplinary research and experimental space will be created to develop, test, visualise, discuss, and implement ideas and concepts on the restoration and expansion of green-blue infrastructure (GBI), and thus for the promotion and implementation of NBS, in the urban area of Huế. In cooperation with stakeholders from science, politics, administration, and civil society, the international project consortium of Independent Institute for Environmental Issues (UfU), Humboldt-Universität zu Berlin (HUB), Mientrung Institute for Scientific Research (MISR), Thừa Thiên Huế Institute for Development Studies (HuếIDS), and the Faculty of Architecture of the University of Sciences/Huế University (HUSC) will generate joint knowledge for stakeholders and decision-makers on NBS, resulting in a city-wide vision – a strategy containing guiding principles and best-practice recommendations for a greener, more resilient, and sustainable urban development of Huế, including proposals for specific measures of GBI implementation.

Project website: www.greencitylabHuế.com

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Reference

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EXECUTIVE SUMMARY

This report elaborates a methodology for the assessment of potential impacts of green-blue infrastructure (GBI) interventions on ecosystem service supply and demand. The work described herein is based on the scenarios developed in WP1. In assessing supplied ecosystem services, and through an ordination of associated ecosystem service potentials, the importance of GBI interventions for delivering benefits to citizens of Hue is illustrated, e.g., regarding the regulation of air quality. For example, it is estimated that depending on the underlying scenario, Hue's GBI may filtrate up to about 670 tons of pollutants each year. The modelling of ecosystem service supply is subsequently contrasted with the spatially explicit modelling of population for both, the 2019 status-quo and 2030. This is referred to as the demand side, as Hue's citizens are the potential beneficiaries of the benefits provided by Hue's GBI. Additionally, scenario-specific impacts on green space availability are estimated. Here, green space availability subsumes both the accessibility to green spaces, i.e., proximity to urban green spaces and/or other elements of the GBI, as well as their provision, i.e., available area per capita.

KEYWORDS

Impact assessment; ecosystem service supply; ecosystem service ordination; ecosystem service valuation; population modelling; dasymetric mapping; spatial disaggregation; green space availability; green space accessibility; green space provision

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1 Introduction

The work conducted in work package WP2 is concerned with the assessment of potential impacts of green-blue interventions on ecosystem service supply at the city-level (case study level). The analysis is based on the scenarios implemented in WP1, Task 1.1. A closer evaluation of service-providing areas (i.e., supply-side modelling, with particular focus on urban green spaces) as well as of the spatial distribution of citizens as the beneficiaries of ecosystem service provisioning (demand-side modelling) is part of the work conducted in WP2.

The methodological steps undertaken in the work described hereafter are depicted in Figure 1. As shown, demand-side modelling includes, first, the spatially explicit modelling of the 2019 population (status-quo), based on available statistical data. Second, 2030 total population is estimated at the case study area level, and subsequently, third, the spatial distribution of estimated 2030 population is modelled for the case study area. Demand-side modelling aims at the quantitative estimation and spatial identification of possible beneficiaries or users of Hue's GBI. Supply-side modelling contrasts this demand-side estimation. The estimation of supply is concerned with the assessment of possible impacts of planned and/or proposed interventions on ecosystem service provisioning and thus, the delivery of associated benefits. It also includes an assessment of green space availability, here subsuming green space accessibility and green space provision.

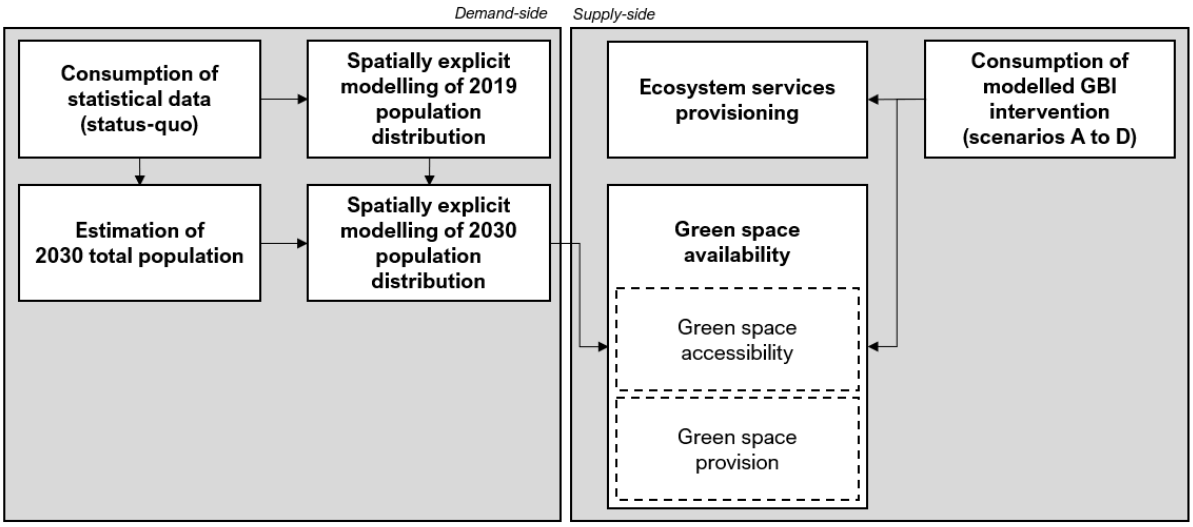

Figure 1. Methodological steps undertaken in Task 2.1 for the assessment of impacts of GBI planned (under baseline scenario A) and/or proposed interventions (under scenarios B, C, and D) at city-level.

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2 Demand-side modelling

Demand-side modelling seeks to estimate and identify potential beneficiaries of ecosystem service provisioning, and simultaneously, potential users of Hue's GBI, in a spatially explicit manner. As shown in Figure 1, demand-side modelling subsumes the following methodological steps: (i) based on statistical data retrieved for the status-quo year 2019, population is modelled in a spatially explicit manner for this time step; (ii) based on information by the General Statistics Office of Vietnam (2020b), population for the year 2030 is estimated at the case study level; and (iii) based on the estimated 2030 population, population is also modelled spatially explicitly for the year 2030.

2.1 Spatially explicit modelling of 2019 population

For the status-quo 2019, various demographic data is available (General Statistics Office of Vietnam, 2020a), including tabulated total population count aggregated at the level of various administrative units. However, no spatially explicit data could be identified. Consequently, spatial distribution of the population for the status-quo must be modelled. To support this modelling, aggregate data has been retrieved by consortium partners for the smallest spatial-administrative units (source zones) available. This refers to ward-level data within the Hue city itself, and to the level of communes within the remaining case study area (Figure 2).

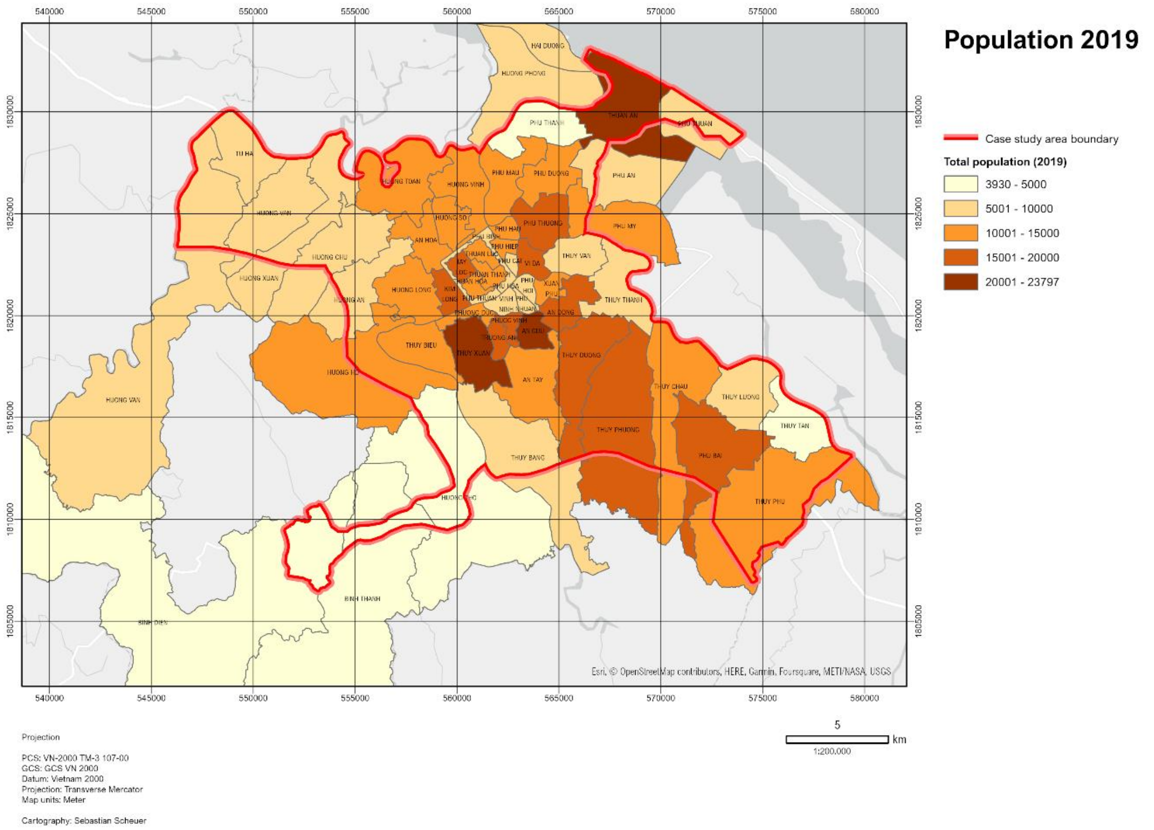

Figure 2. Choropleth map of the total population at the level of administrative units for the status-quo year 2019. Looking at the map, it becomes evident that in certain administrative units, the case study area covers only parts of the respective area. This is particularly the case for communes to the west and south of Hue.

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The spatial disaggregation of 2019 population is subsequently conducted through dasymetric mapping (Mennis, 2009; Mennis & Hultgren, 2006). Dasymetric mapping estimates, for a given spatial feature, i.e., a spatially homogeneous unit t as target zone, a disaggregated property $\hat{y}_t$ (here: population), as a function of a relative density $\widehat{D}_c$ (here: corresponding to population density of an ancillary land-use class *c*) and feature area A_t, standardized at the level of administrative units *m* (as source zone) for which the property is given in aggregate form y_s:

$$\hat{y}_t = y_s \left(\frac{A_t \widehat{D}_c}{\sum_{t=1}^{m} (A_t \widehat{D}_c)} \right) \tag{1}$$

The steps conducted to estimate disaggregated population counts are detailed below, and include: (i) determination of relevant ancillary classes (land-uses) for disaggregation; (ii) identification of spatially homogeneous units (target zones) within each administrative unit (source zone) of interest; (iii) estimation of relative densities for each ancillary class of interest; and (iv) estimation of population for each target zone, based on Eq. 1.

As described above, the estimation of disaggregated population for 2019 is based on total population provided by consortium partners for the administrative units within the case study area. To support the aforementioned steps, first, data that has been provided as multiple spreadsheets were systematically restructured and compiled as a database (Microsoft Access). Hence, total 2019 population of each relevant administrative unit (source zone), i.e., ward or commune, was compiled, and assigned an administrative code as primary key (table *PopulationAndDensity_2019*). Each source zone was subsequently mapped to its higher-level administrative unit, i.e., district (table *CommuneToDistrictMapping*). Moreover, in the provided data, population for each source zone has been categorized as either urban population or rural population (table *PopulationTypesPerAdm*). Second, the compiled database was added as data source in ArcGIS Pro to make direct use of the compiled data.

2.1.1 Determination of relevant ancillary classes (land-uses) for disaggregation

Ancillary classes considered for population disaggregation include primarily residential land-uses, i.e., urban residential land and rural residential land. All other land-use classes are not considered for the disaggregation.

2.1.2 Identification of spatially homogeneous units (target zones) within each administrative unit (source zone) of interest

For disaggregation, spatially homogeneous units are required as target zones for disaggregation. Here, spatially homogeneous unit refers to each target zone having only a single, specific land-use, i.e., being either of type rural or urban residential land. As the estimation of disaggregated population for these target zones is based on the aggregated population count per source zone, all target zones *m* (spatially homogeneous units) must be associated with their source zone *s*. To do so, the integrated land-use layer prepared in WP1 (cf. Scheuer et al., 2022), including land-use 2019, was spatially intersected with the relevant administrative units (wards and communes).

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2.1.3 Estimation of relative densities for each ancillary class of interest

Subsequently, for each land-use as ancillary class, a respective relative density $\widehat{D}_c$ needs to be determined. However, for all land-uses not considered for disaggregation, relative density is equal to 0. For the relevant ancillary classes urban residential land and rural residential land, relative densities are subsequently estimated based on Eq. 2, and using the layer representing spatially homogeneous units as prepared previously:

$$\widehat{D}_c = \frac{\sum_{s=1}^{m} y_s}{\sum_{s=1}^{m} A'_s} \quad (2)$$

where y_s … total population of a source zone *s*; A'_s … total area of land-use (ancillary class) *c* in source zone *s*; and $\widehat{D}_c$ … relative density of land-use (ancillary class) *c*.

The steps conducted are then as follows: First, for each relevant residential land-use as ancillary class, total area is determined. However, in certain cases as shown in Figure 2, the case study area of interest covers only corresponding source zones only partially. Therefore, to obtain a better estimate of relative density, for each source zone, the total area of urban and/or residential land located within and located outside the case study area is determined per source zone (tables *UrbanLandUsePerAdmInsideArea*, *UrbanLandUsePerAdmOutsideArea,* *RuralLandUsePerAdmInsideArea* and *RuralLandUsePerAdmOutsideArea* based on a spatial intersect of 2019 land-use and case study area, and query *LandUseTypesAggregation* based on aforementioned tables).

Second, aggregated population of a given source zone is associated with a specific residential land-use (either rural or urban). To do so, based on table *LandUseTypesAggregation*, those source zones where selected that are fully located within the case study area.

Then, third, from the selected source zones: (i) the population total $\sum_{s=1}^{m} y_s$ of the source zones *s* exclusively with urban residential land (i.e., 100% of A'_s in the m source zones corresponds to urban residential land) is used to determine relative density of urban residential land; this relative density is equal to 0.0097573236794289; and (ii) the population total $\sum_{s=1}^{m} y_s$ of the source zones *s* exclusively with rural residential land (i.e., 100% of A'_s in the m source zones *m* corresponds to rural residential land) is used to determine relative density of rural residential land; this relative density is equal to 0.0059656351895196.

2.1.4 Estimation of population for each target zone

Finally, disaggregated population is estimated based on Eq. 1. Based on the previously identified target zones and relative densities, this estimation is straightforward.

Disaggregated population for the status-quo (2019) is depicted in Figure 3. The total population for the case study area in 2019 is, according to this estimation, equal to 621,227 inhabitants. This corresponds to about 55% of the 2019 province's total population. In the case study area, about 497,054 inhabitants are considered urban residents, and 124,173 inhabitants are considered rural residents. Re-aggregating disaggregated population at district level, modelled population for Hue is equal to 351,622; for Huong Thuy, 103,789; for Huong Tra, 85,775; and for Phu Vang, 80,041 inhabitants. In comparison to provided data as a reference, differences result from inaccuracies in provided spatial data or, e.g., rounding errors.

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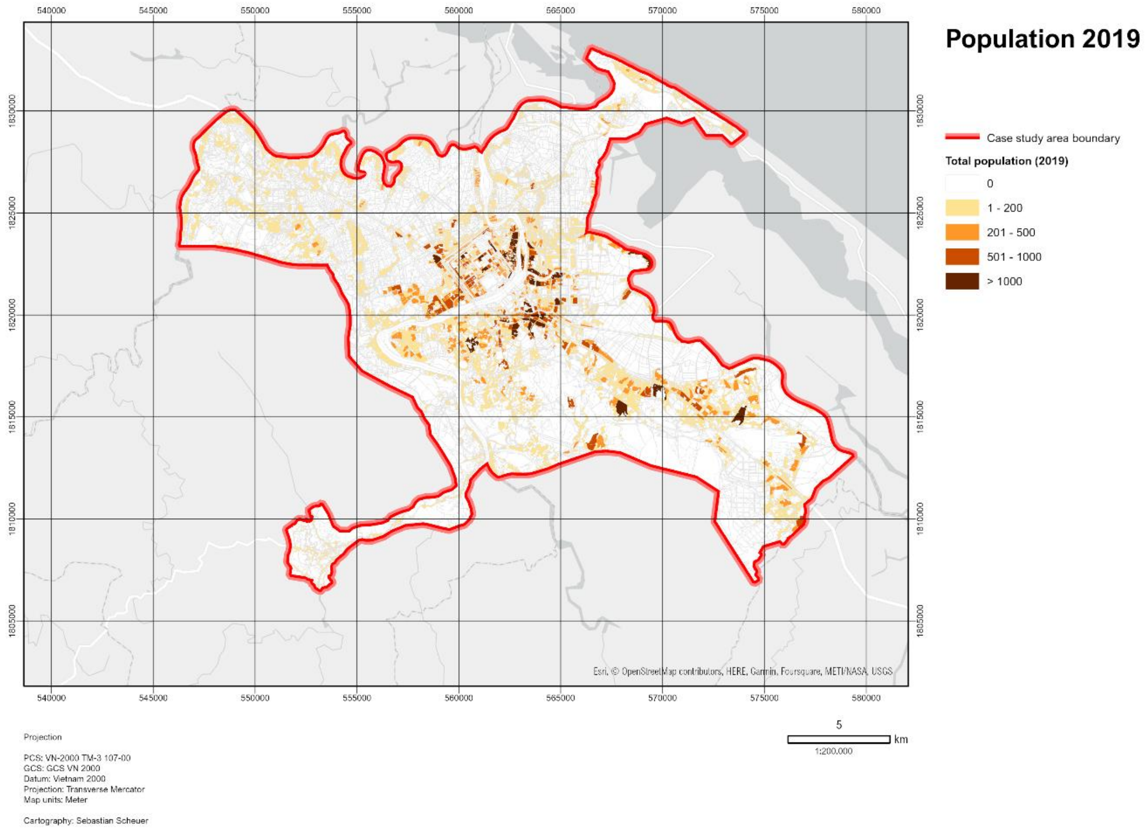

Figure 3. Spatially disaggregated population for the status-quo year 2019.

2.2 Estimation of total population for 2030

Neither for the immediate case study area, nor for all relevant administrative units at the level of wards and communes, could specific data on projected total population for the year 2030 be identified. Consequently, this data had to be estimated. Here, it is suggested to do so based on population estimates for the period 2019 to 2069 by the General Statistics Office of Vietnam that is available for the provincial level (General Statistics Office of Vietnam, 2020b). The provincial level is the next highest administrative level to districts, and therefore the most immediate level at which relevant data is available. From these provincial population projections, the total population for each year from 2020 to 2050, and the corresponding year-on-year change in population are determined for the Thua Thien Huế province (Figure 4).

As shown in Figure 4, the projections by the General Statistics Office of Vietnam (2020b) estimate an overall increase of population for the province level until 2045[1]. Looking at the year-on-year change in population, the population change rate is estimated to peak in 2025, and to decline thereafter for most of the period of interest until 2030. For the period 2019 to 2045, the average change in population is estimated at 0.29% p.a. In comparison to that, for the period 2009 to 2019, this rate is estimated at 0.37% p.a.

[1] The period 2045 to 2069 is present in the reference, but was not considered in this analysis.

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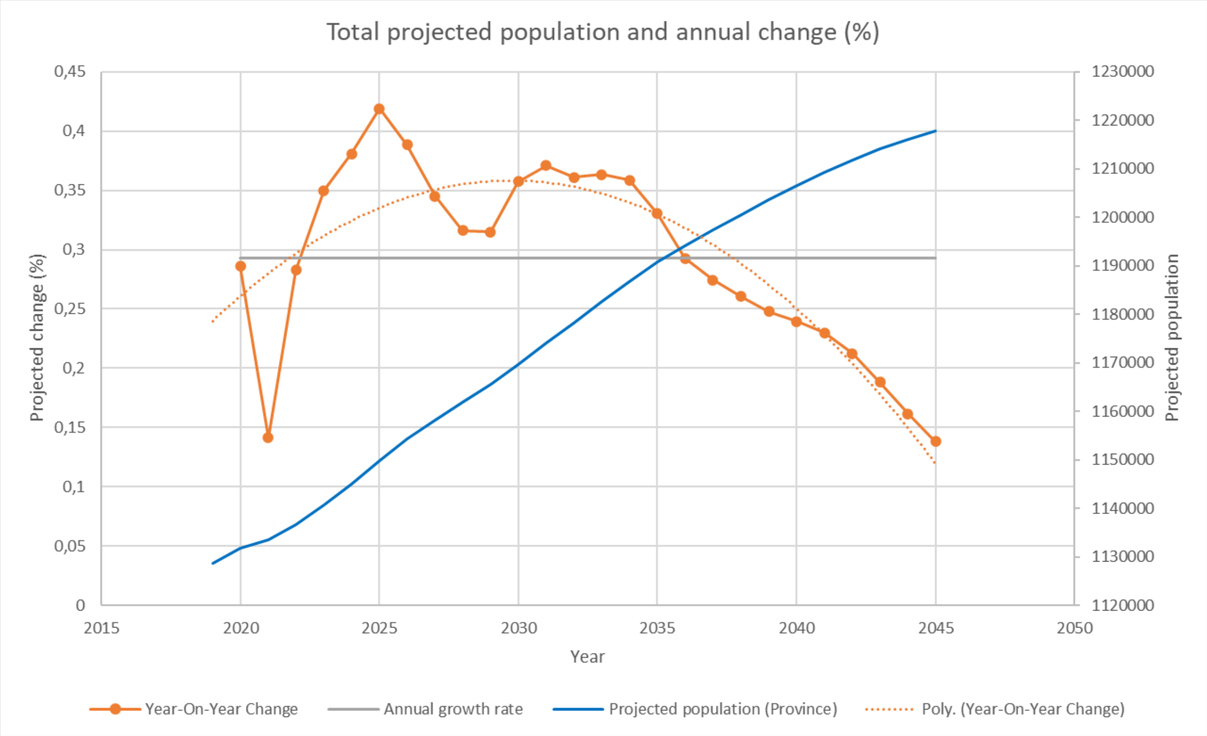

Figure 4. Projected total population and estimated year-on-year population change rates, including polynomial fit, at province level.

Moreover, also as shown in Figure 4, a polynomial trend was fitted to the estimated province-level year-on-year change (Eq. 3). This trend estimates the population change rate for a given year as a function of time (here: year):

$$r_t = -0{,}001026114\, t^2 + 4{,}16549825645918\, t - 4227{,}09028791644 \qquad (3)$$

where r_t … population change rate in year *t*, and *t* … given year in the assessed period of time.

Subsequently, the polynomial fit is transferred to the case study area (Figure 5), and adjusted to the known change rates. Then, based on projected year-on-year change, 2030 population is estimated.

This trend transfer is done in a two-pronged approach. First, the modelled fit is applied to each relevant district individually, i.e., the polynomial fit is adjusted to the known annual population change rates for the reference period 2009-2019 of each district (scenario P1). These reference change rates were estimated at 0.467% p.a. for Hue; 1.780% p.a. for Huong Thuy, -0.048% p.a. for Huong Tra, and 1.070% p.a. for Phu Vang. Estimated average annual population change rates for the period 2019-2045 are then modelled as 0.52% p.a. for the city of Hue, 1.82% p.a. for Huong Thuy, 0.005% p.a. for Huong Tra, and 1.122% p.a. for Phu Vang. The total population in 2030 at the case study level is so determined at 677,709 inhabitants (cf., Figure 6 and Table 1). Under this scenario P1, total population of the case study area is estimated to increase by 56,482 inhabitants (+9.1%) from 2019 to 2030, and the estimated total population in the case study area accounts for about 57.9% of the estimated population of the Thua Thien Huế province in 2030.

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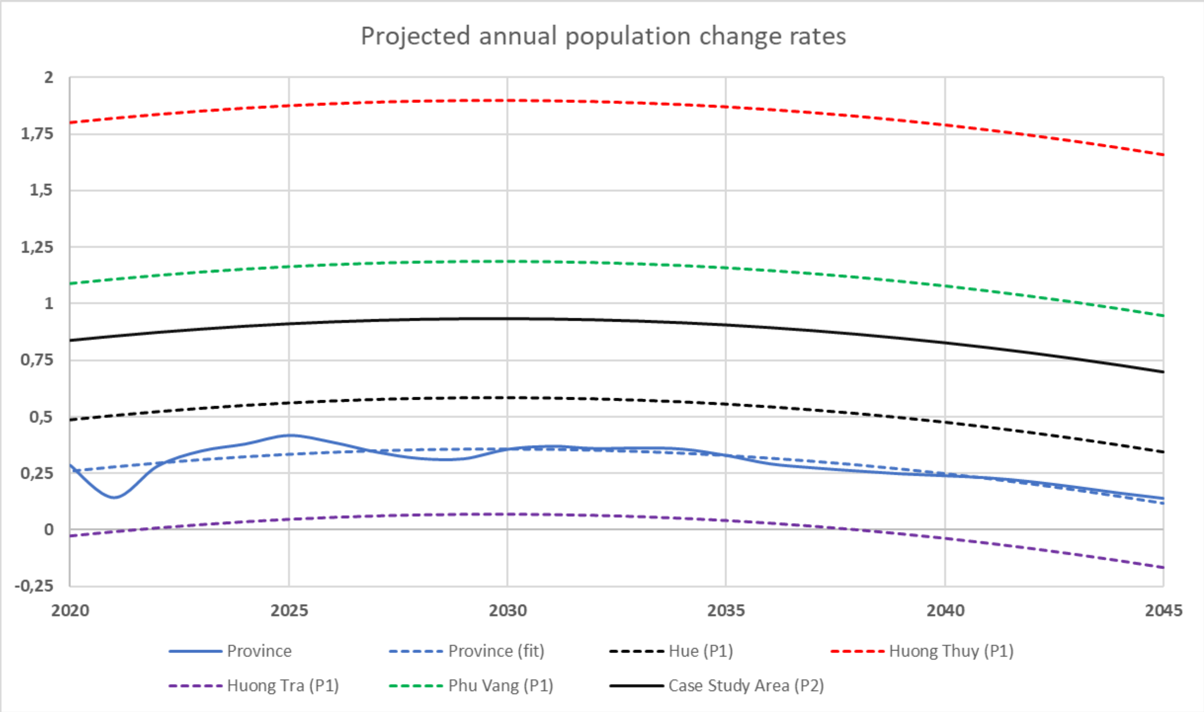

Figure 5. Population change rates at district level (scenario P1) and at case study area level (scenario P2), obtained through transfer of province-level polynomial fit to project known reference average change rates 2009-2019. Additionally, population change rate at province level is shown for reference.

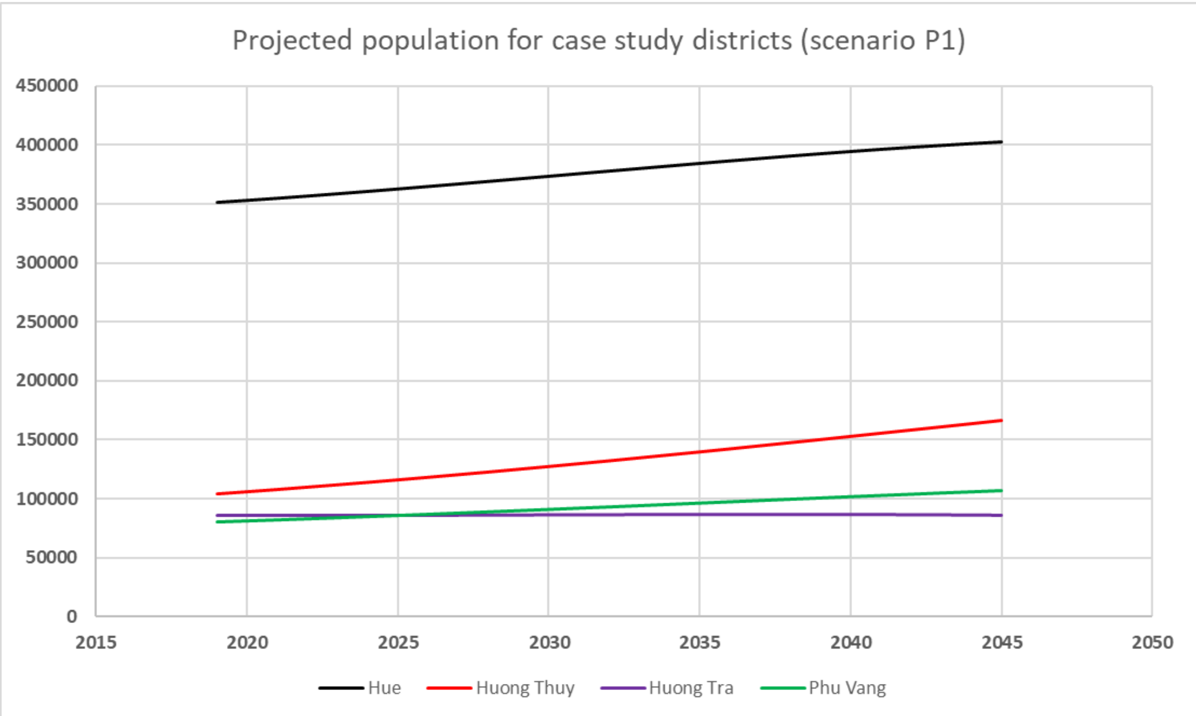

Figure 6. Projected population at district level under scenario P1.

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However, it needs to be acknowledged that clearly, there are numerous uncertainties involved in so-doing. This does not only refer to use of the modelled polynomial fit per se, but is also related to the application of a single fit to multiple, heterogeneous spatial units. To minimize the latter uncertainty at least in part, a second population projection (scenario P2) estimates total population in 2030 based on the annual average population change rate averaged for the whole case study area, i.e., the mean of individual reference change rates. This mean reference change rate 2009-2019 is approximately 0.8175% p.a. Subsequently, the derived polynomial fit is adjusted to this reference value. So-doing approximates the annual average population change rate with 0.87% for the period 2019-2045. This is significantly higher than the estimated annual average change rate at province level, being 0.29%. Subsequently, total population at the case study area level is estimated for 2030 at 685,766 inhabitants (+8057 inhabitants compared to the scenario P1 (cf. Figure 7 and Table 1). Hence, under this scenario P2, population of the case study area would increase by 64,539 inhabitants (+10.3%) from 2019 to 2030, and the share of province-level population that is located within the case study area in 2030 may reach 58.6%.

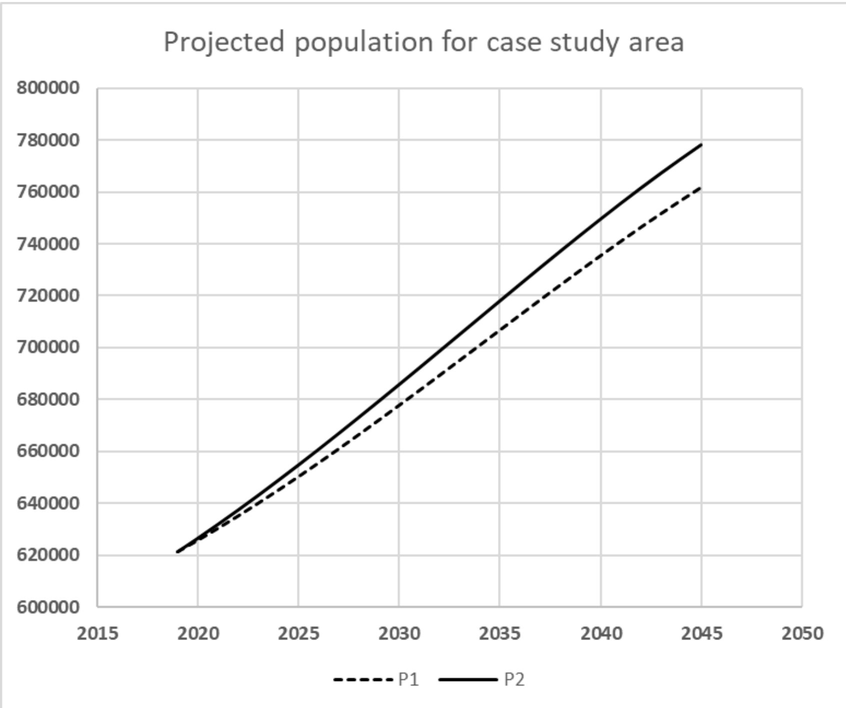

Figure 7. Projected total population at the case study level for scenarios P1 and P2.

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Table 1. Estimated population 2030 at district level and case study level for scenarios P1 and P2.

Spatial unit within case study area	**Population**		
	2019	**2030 (P1)**	**2030 (P2)**
Hue	351,622	373,582	
Huong Thuy	103,789	127,181	
Huong Tra	85,775	86,128	
Phu Vang	80,041	90,818	
Total	621,227	677,709	685,766

2.3 Spatially explicit modelling of 2030 population

2.3.1 Identification of 2030 target zones, and estimation of their population

Overall, the 2030 population is assessed as the sum of total population 2019, and estimated population change from 2019 to 2030. As described previously, under both scenarios P1 and P2, a population growth is expected for this period. As before, the 2030 total population is disaggregated to relevant ancillary (land-use) classes, forming the target zones for disaggregation. Overall, ancillary classes for 2030 remain identical to those for the 2019 timestep, i.e., rural and urban residential land are considered as target zones for the spatial disaggregation.

However, the disaggregation for 2030 also considers land-use changes from 2019 to 2030 in the disaggregation process, including: (i) conversion of residential land to non-residential land; (ii) residential land-uses remaining unchanged; (iii) densification of rural residential land to urban residential land; (iv) urban expansion, i.e., conversion of non-residential land to urban or rural residential land-use. To assume the potential impacts that these land-use changes may trigger in regard to spatial population patterns, so-called population transfer rules are formulated (Figure 8). Simultaneously, so-called population pools are formulated, that support the accounting of changes to patch population counts (Figure 9): (i) in case of urban or rural residential land that is anticipated to be converted to non-residential land-use from 2019 to 2030, it is assumed that the 2019 population count disaggregated to these patches needs re-distribution to other patches[2]; (ii) in case of no observed changes to residential land, or observation of potential changes only, it is assumed that patches remain equal to 2019 in terms of size and residential land-use type. Consequently, for these patches, 2019 population is held constant, i.e., there are no changes assumed to occur from 2019 to 2030 in terms of population count[3]; and (iii) in case of a densification of residential land from rural type to urban residential

[2] Forming population pool A („Share of 2019 population for redistribution"), corresponding to 194,787 inhabitants

[3] Forming population pool C as part of "Transferred population count", corresponding to 406,985 inhabitants

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land, it is assumed that for these patches, the 2019 population remains in place[4]. However, as outlined below, the total population count per patch is adjusted to reflect the increase in relative density (Figure 8).

It follows that due to a given count of population assumed to remain unchanged from 2019 to 2030 at the patch level (pool C), the total population relevant for spatial disaggregation includes the following counts (pools): (i) the population growth until 2030 as estimated by scenarios P1 and/or P2 (pool G); and (ii) the population count requiring re-distribution due to underlying conversions of residential to non-residential land-use types (pool A; cf. Figure 8). Hence, the combined total of pools A and G that forms population pool E is subsequently disaggregated to residential land as follows. First, for patches assumed to become urban (densification), population patch count is adjusted to the higher urban relative density. To do so, the population count of pool B (i.e., former rural population that is now considered urban) is increased by 63.55%, i.e., a total of 12,364 inhabitants (pool D) is disaggregated from pool E to these patches. Second, the remaining population count (pool R = E-D) is finally disaggregated to new residential land, i.e., non-residential land in 2019 that is converted to urban or rural residential land until 2030, denoting urban expansion, including expansion through deforestation or "ungreening"/conversion of former urban green spaces. In line with the dasymetric mapping method, disaggregation of pool R is conducted to relevant ancillary classes based on their (rural or urban) relative density.

The population transfer rules described above are summarized graphically in Figure 9; the disaggregated population is then shown in Figure 10 for scenario P1, and in Figure 11 for scenario P2.

[4] Forming population pool B as part of „Transferred population count", corresponding to 19,456 inhabitants

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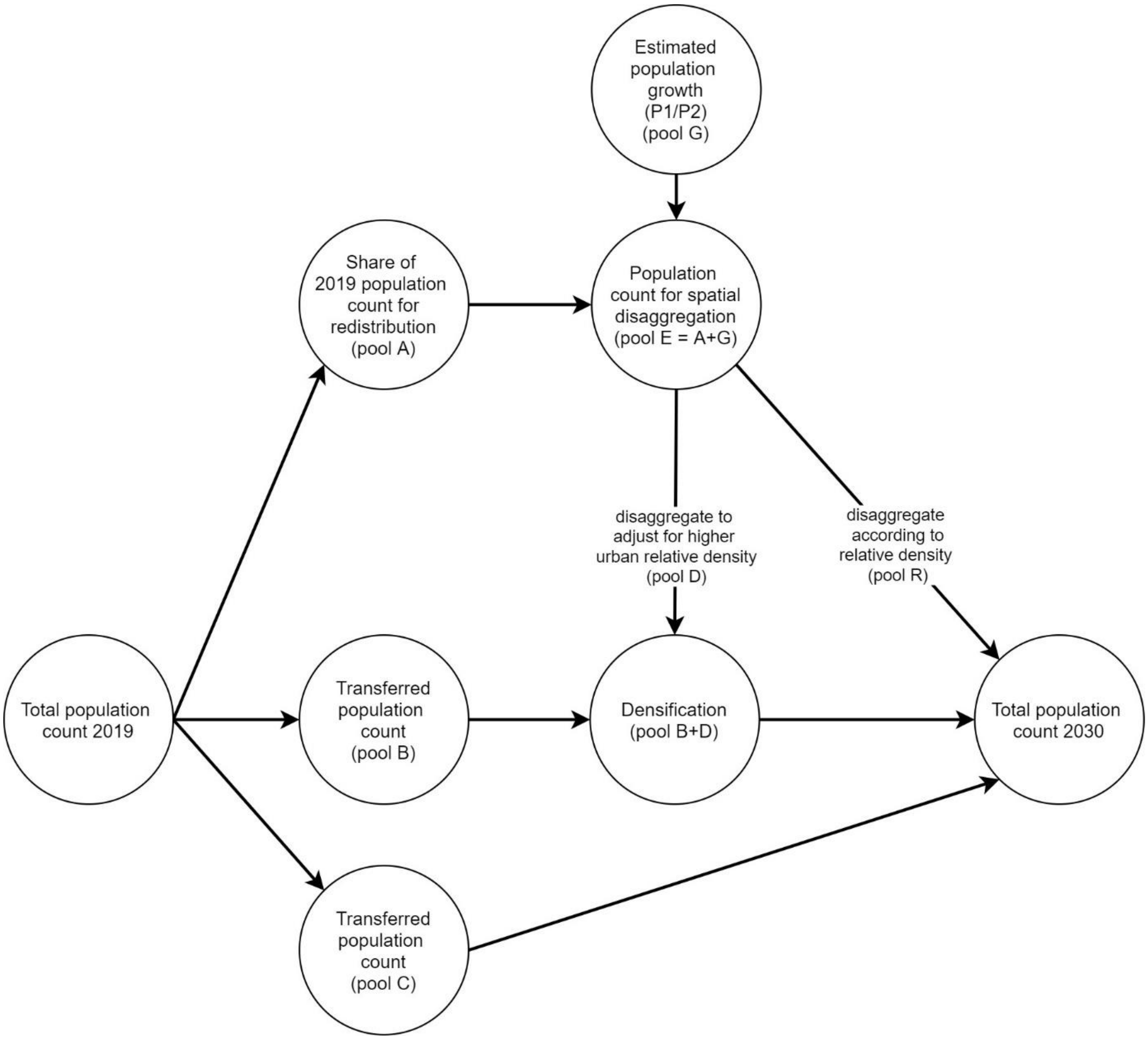

Figure 8. Population transfer rules supporting the spatial disaggregation of the population estimated for 2030. As shown, population pools are used for accounting changes in population from 2019 to 2030, where pool G refers to the estimated overall population growth.

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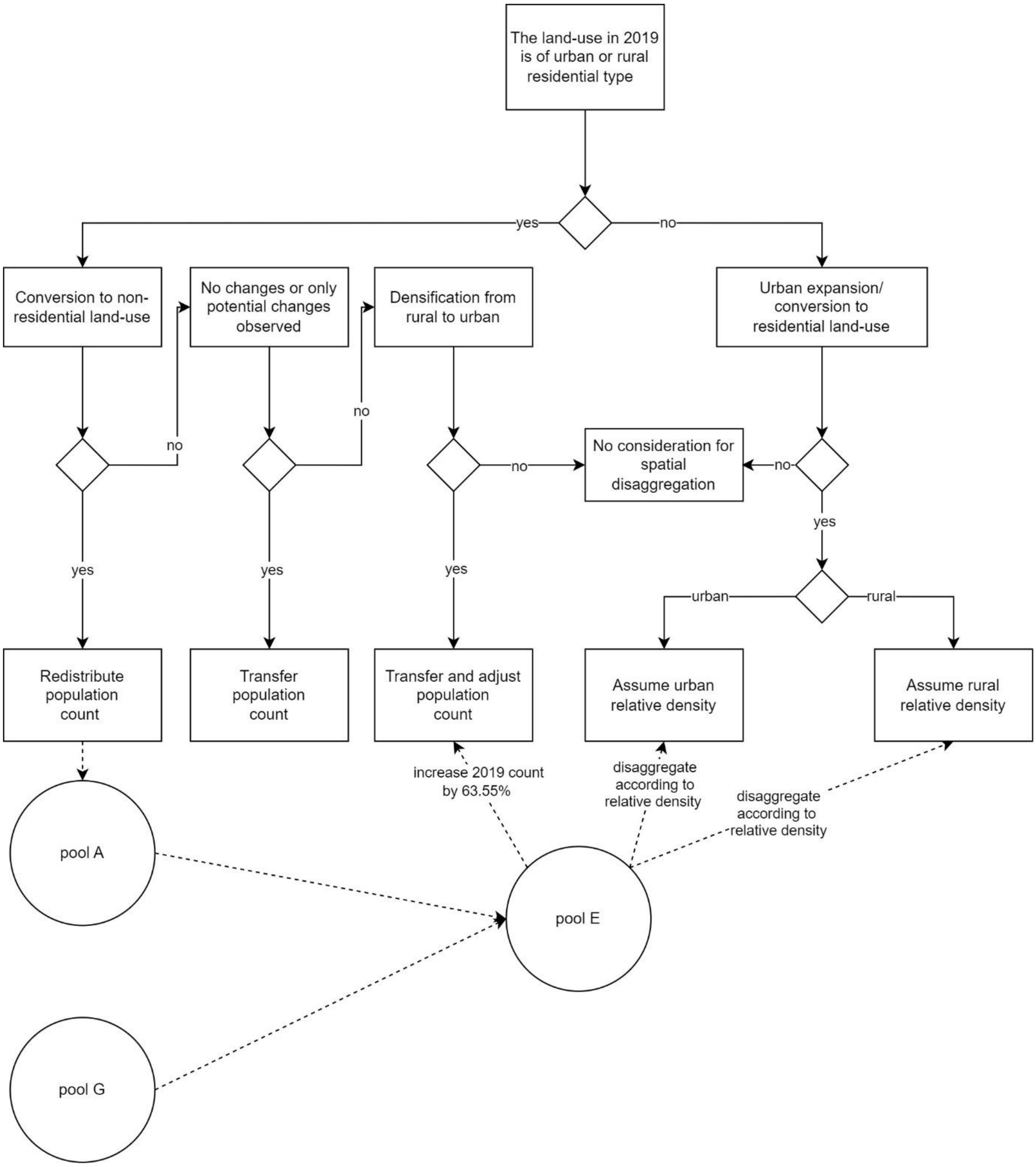

Figure 9. Flowchart of the disaggregation rules for the spatial disaggregation of estimated 2030 population.

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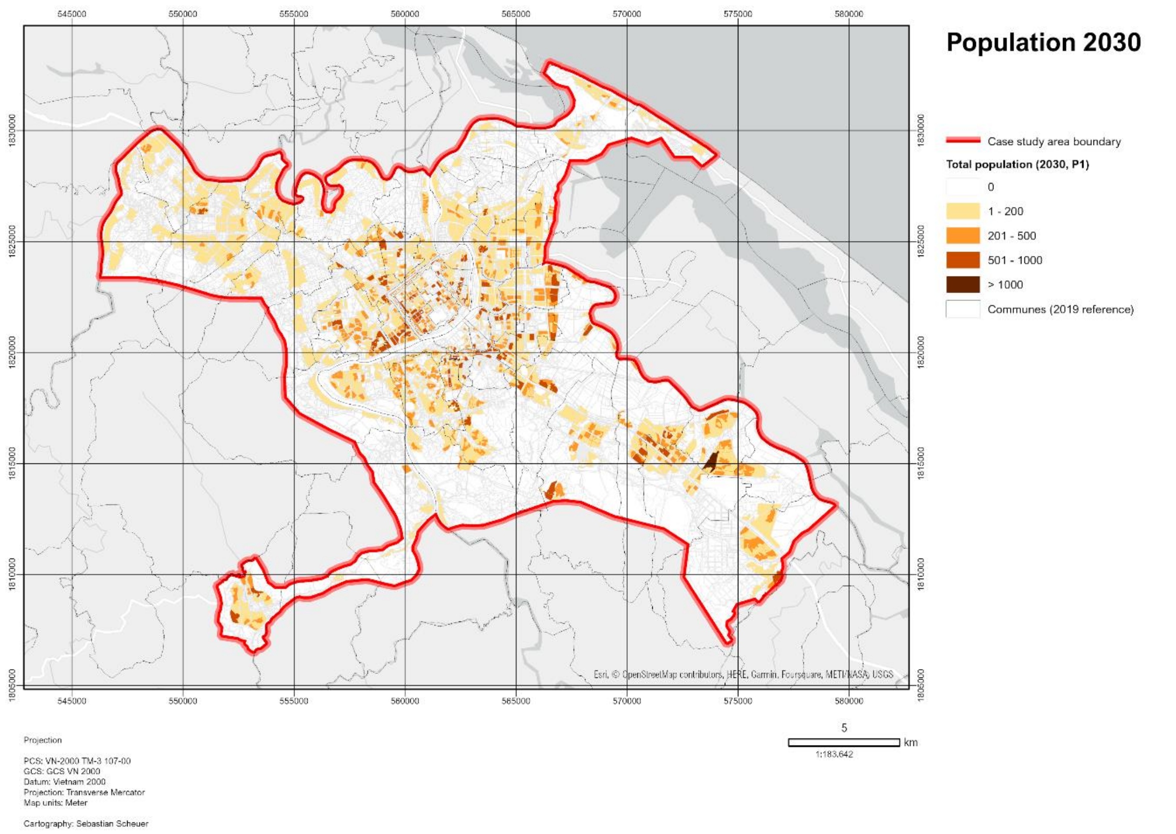

Figure 10. Modelled spatial distribution of population estimated for 2030 under scenario P1.

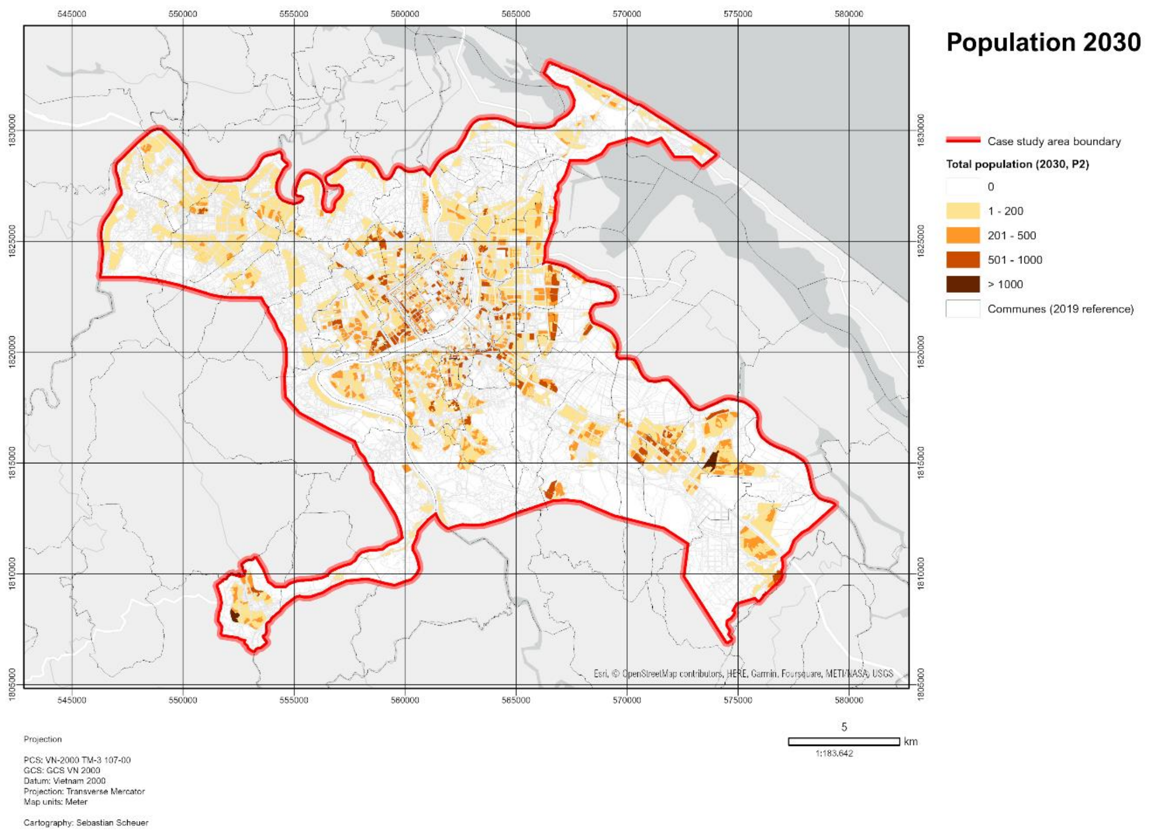

Figure 11. Modelled spatial distribution of population estimated for 2030 under scenario P2.

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The formulated transfer and disaggregation rules account particularly for the business-as-usual scenario, i.e., scenario A. Importantly, in contrast to scenario A, scenarios B to D foresee the conversion of parts of new residential land, amongst other target zones, for greening interventions. In these cases, theoretically, available target zone area may be reduced, e.g., by a few percent only, or the respective target zone may be consumed entirely by greening interventions. This may seem to necessitate a further “re-distribution of population disaggregated to these patches. However, to avoid the introduction of further assumptions and thus uncertainties, it is suggested below that an intermediate level is used for further analysis, that is obtained by hexagonal tessellation. So doing assumes that needs for further “re-distribution” due to greening interventions may be compensated for within each resulting tile.

3 Supply-side modelling and impact assessment

3.1 General considerations

Supply-side modelling as part of impact assessment is concerned with the assessment of ecosystem services supply. Impact assessment further assesses changes in green space availability, i.e., here, accessibility and provision of GBI elements. To assess these impacts of planned (baseline scenario A) and proposed interventions (scenarios B to D) on accessibility, provision, and ecosystem services supply, it is proposed to adopt tessellation for the construction of an intermediate feature space, with relevant data (area of GBI elements, ecosystem service supply, population, etc.) aggregated at this intermediate level. This approach is proposed for the following reasons: (i) the virtual allocation of interventions does not allow for a specific localization of resulting GBI features. Hence, a spatially explicit view on outcomes of planned or proposed interventions is more readily established through an aggregation of land-use changes at an intermediate level; (ii) the assessment of ecosystem service supply (and further indicators) is facilitated; (iii) aggregation to an intermediate level allows abstracting minor changes in spatial population patterns. This particularly refers to impacts on spatial population patterns due to conversion of (planned) residential land to green-blue features. In this case, as previously discussed, a re-distribution of population may be needed. However, the extent of such a need is challenging to estimate. For example, for smaller shares of land to be converted, changes in density of *to-be-constructed* residential built-up may already be suitable to account for such circumstances, whereas otherwise, it is assumed that necessary shifts in population may be accounted for within the intermediate spatial analysis units (i.e., at tile-level); and (iv) in this context, regarding per-capita assessments, the number and shape of administrative units remains uncertain for 2030, as evidenced by numerous changes to city boundaries throughout the project period. Here, per-capita green space area may therefore be assessed more readily at the intermediate spatial level.

For tessellation, hexagonal tiling is proposed, as hexagons feature equidistant boundaries when compared to rectangular grids. Subsequently, ecosystem service supply and green space availability are assessed at the level of hexagon tiles (Figure 12).

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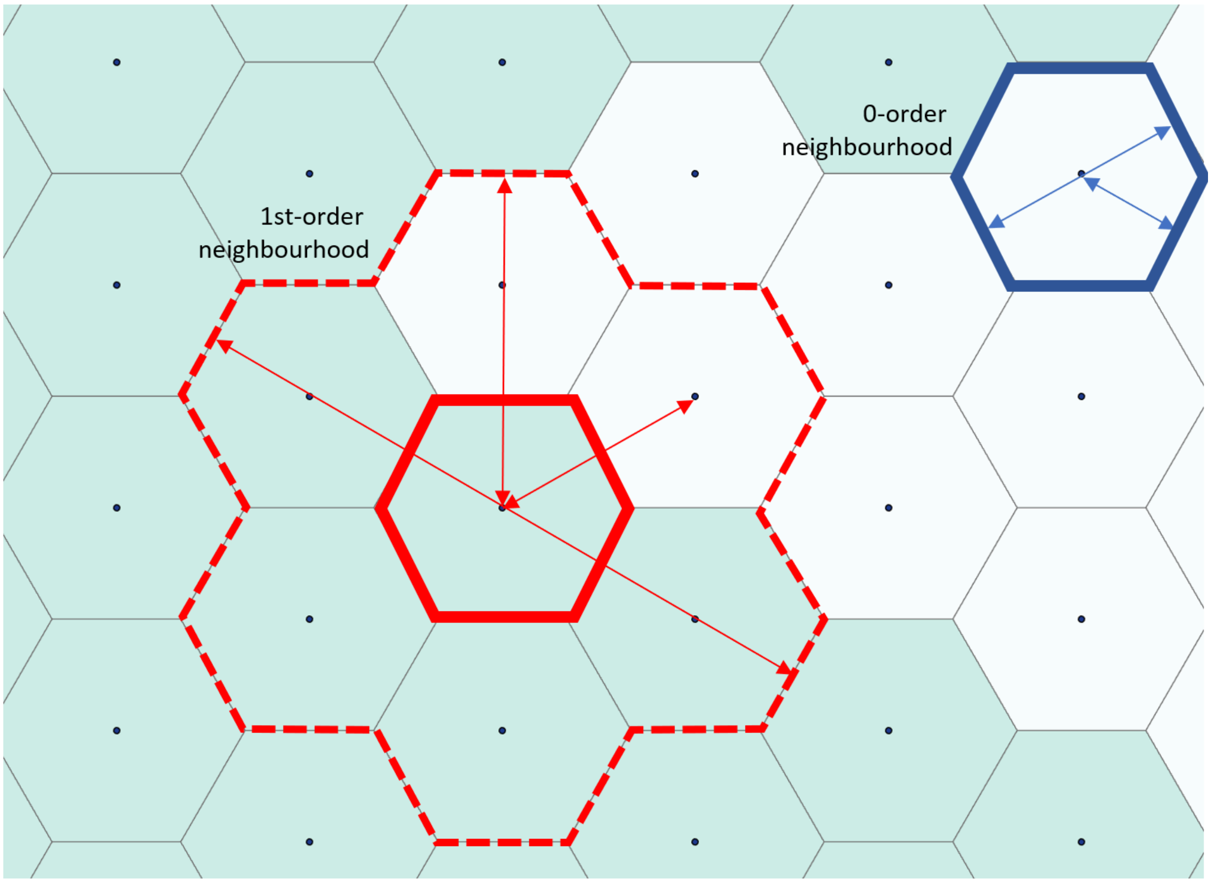

Figure 12. Depiction of hexagonal tessellation, and illustration of tile neighbourhoods.

To support the assessment, a second database (Microsoft Access-based) is compiled. In the database, the tessellation features (hexagons) are included with their ID key (table *HexagonShapes*). To determine population and land-use area/area of GBI elements per tile, the WP1 land-use change layer as well as disaggregated population layers were intersected with the tessellation layer, and the intersection results for each scenario A to D were imported into the database (tables *Intersect_A* to *Intersect_D*). Based on this data, total area of green spaces, forests and water bodies implemented under baseline scenario A, and proposed in addition to baseline as foreseen in scenarios B to D (tables *ConvertedLand_B* to *ConvertedLand_D*) has been determined. Using various queries, data was subsequently compiled and systematized at tile level. This includes urban green space features, forest features, and features of water bodies at tile level under baseline scenario A (tables *GS_A*, *Forest_A*, and *Water_A*), the total GBI area per tile in the different scenarios (queries *Grid_A_GBI* to *Grid_D_GBI*), as well as the area of GBI elements proposed for implementation in scenarios B to D per tile (queries *Grid_ConvertedLand_B_SumOfAreas* to *Grid_ConvertedLand_D_SumOfAreas*, as well as *Grid_B_Additional_GBI* to *Grid_D_Additional_GBI*). Estimated 2030 population count at tile level is given in table *Grid_POP*.

Subsequently, based on this data, ecosystem service supply as well as green space availability indicators as described below were determined (tables *tbl_ESS_A* to *tbl_ESS_D*, *tbl_ESS_ALL*, and *tbl_GS_capita*).

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3.2 Ecosystem service supply

In the following, potentials for the delivery of ecosystem services and/or associated benefits are elaborated for a set of selected ecosystem services. Here, a focus is on Hue's GBI elements, i.e., green-blue features (apart from agricultural land and special land-uses) under baseline conditions (scenario A), and on the additional interventions in accordance with scenarios B to D. Ecosystem services delivered by green-blue elements in other urban land-uses, e.g., within agricultural land, residential land, commercial land, etc., are ignored.

3.2.1 Regulation of air temperature and outdoor thermal comfort

GBI provides cooling through evapotranspiration and the regulation of solar radiation by shading, and further modifies air flow and humidity. In so doing, GBI contributes—under most conditions—to an improvement of outdoor thermal comfort (OTC; cf. Bowler et al., 2010; Livesley et al., 2016; Peng et al., 2022). A test case scenario has been implemented for a more-specific determination of the impacts of suggested GBI interventions on air temperature and OTC (Scheuer et al., in preparation). This test case confirms the effectiveness particularly of trees but also (smaller scale) grassy surfaces for delivery of cooling benefits and the improvement of OTC. Higher-density tree ensembles in shallow, wide street canyons were found to provide cooling of UTCI of potentially up to about 2K under muggy meteorological conditions, with higher impacts on air temperature and UTCI in shaded areas. Thereby, the importance of shading and evaporative cooling is emphasized for the regulation of air temperature and OTC. Moreover, despite considerably smaller impacts on air temperature, also green verges as grassy surfaces were found to contribute to OTC by reduction of UTCI, in line with Djekić et al. (2018), again underscoring the importance of evaporative cooling. In addition, synergistic cooling benefits, e.g., regarding combinations of tree-based interventions with grassy elements and/or green facades, could also be identified (Scheuer et al., in preparation; Graça et al., 2022; Jamei et al., 2020; Bartesaghi-Koc et al., 2020).

Therefore, evapotranspiration and shading are used as proxy to indicate GBI potential for the regulation of air temperature and OTC. High potential for evapotranspiration is tied to interventions with high share of green elements and pervious areas and thus, a high share of areas providing potential for evaporation or transpiration. Here, this is assumed to be the case for most of the intervention/land-use types under consideration, including water bodies, but apart from urban green spaces under baseline conditions. This is due to their potentially higher share of sealed surfaces, therefore, for this land-use type, only moderate evapotranspiration potential is assumed (Figure 13). Regarding shading, high potentials are tied to higher-density tree-based interventions/land-use types, i.e., improved (woody) green spaces and forests. Green spaces under baseline conditions as well as orchards with their comparatively lower share of tree cover are assumed to provide up to moderate shading potential. Extensive greening (i.e., meadows) and water bodies are considered to provide no shading opportunities (Figure 13).

3.2.2 Regulation of air quality/air purification

The regulation of air quality by GBI through removal of particulate matter and gaseous pollutants, e.g., SO_2, NO_2, or PM_x, is well-recognized in the literature (Grylls & van Reeuwijk, 2022; Hirabayashi, 2021; Irga et al., 2015; Jim & Chen, 2008; Kremer et al., 2016; Muresan et

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al., 2022; Nowak et al., 2002; Riondato et al., 2020). Uncertainty remains in local potentials, i.e., rates of pollutants removal through dry deposition of particulate matter or absorption of gaseous pollutants, which is a function of tree traits and tree health, humidity, wind speed, air temperature, soil moisture, and ambient pollutant concentration driving deposition velocity and thus pollutant flux (Jim & Chen, 2008; Lindén et al., 2023; Muresan et al., 2022). Generally, deposition velocities of different types of vegetation were found as coniferous trees > broadleaf, evergreen trees > deciduous trees; and shrubs > herbs (Wu et al., 2021). However, particularly against ambient pollutant concentrations, reported removal rates may vary considerably. For example:

- Applied in New York City, Kremer et al. (2016) report rates for coarse vegetation (trees) and fine vegetation (grass) based on Nowak et al. (2002) and Yang et al. (2008). Coarse vegetation removal rates include SO_2 (1.32 g m^{-2} a^{-1}), NO_2 (2.54 g m^{-2} a^{-1}), PM_{10} (2.73 g m^{-2} a^{-1}), O_3 (3.06 g m^{-2} a^{-1}), and CO (0.58 g m^{-2} a^{-1}); fine vegetation rates are reported for SO_2 (0.65 g m^{-2} a^{-1}), NO_2 (2.33 g m^{-2} a^{-1}), and PM_{10} (1.12 g m^{-2} a^{-1}).
- For Guangzhou, China, Jim and Chen (2008), averaged across districts, urban tree-based removal rates include 2.73 g m^{-2} a^{-1} for SO_2; 2.64 g m^{-2} a^{-1} for NO_2; and 10.93 g m^{-2} a^{-1} for total particulate matter. These rates are estimated on the basis of averaged deposition velocities of 0.55 cm s^{-1} for SO_2; 0.37 cm s^{-1} for NO_2; and 0.64 cm s^{-1} for particulate matter, and as a function of local pollutants concentrations of about 30 µg m^{-3} SO_2, 70 µg cm^{-3} NO_2, and 198 µg m^{-3} particulates.
- For Santiago, Chile, and against an ambient PM10 concentration of 74,13 µg m^{-3}, Escobedo et al. (2008) report PM_{10} removal rates of, on average, 7.63 g m^{-2} a^{-1} for urban trees; 6.66 g m^{-2} a^{-1} for shrubs; and 1.6 g m^{-2} a^{-1} for grass.
- For Ferrara, Italy, Muresan et al. (2022) report annual PM_{10} removal rates of 3.781 g m^{-2} a^{-1} for deciduous broadleaf trees; 5.603 g m^{-2} a^{-1} for evergreen broadleaf trees; and 14.544 g m^{-2} a^{-1} for conifers.
- For Tokyo, Japan, Hirabayashi (2021), averaged total pollutant removal rate for CO, NO_2, O_3, $PM_{2.5}$ and SO_2 combined is reported at 5.78 g m^{-2} a^{-1}; specifically, for SO_2, this is 0.255 g m^{-2} a^{-1}; and for NO_2, 1.23 g m^{-2} a^{-1}.

Here, an estimate for potential air pollutant removal is determined as follows:

$$F_{iv} = r_{iv} \cdot A_v \cdot T \quad (4)$$

where *F* … amount of pollutant *i* removed by type of vegetation *v*; *r* … rate of pollutant removal for pollutant *i* and type of vegetation *v*; *A* … relevant area of vegetation of type *v*, and *T* … time.

In the following, pollutants SO_2, NO_2, and PM_{10} are considered. Two types of vegetation are assessed in their potential for removal of these pollutants: coarse vegetation, i.e., trees, and fine vegetation, i.e., grass/meadow (extensive greening). Tree-based regulation of air quality is determined for the land-uses: (i) urban green space; (ii) improved urban green space; (iii) orchards; and (iv) forest, as a function of total land-use area and assumed tree cover level, as shown in Table 2. Regulation of air quality by extensive greening is determined based on total area of proposed extensive greening in the various scenarios. Contributions of grassy surfaces in other land-uses, as well as contributions of other green elements, are currently neglected.

Removal rates of coarse vegetation are determined as averages of pollutant-specific removal rates reported in the literature (Table 3). This is mainly due to uncertain or unknown ambient

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concentration of pollutants in Hue at the time of writing. Removal rates of extensive green are estimated based on Escobedo et al. (2008), suggesting that tree removal rates are approximated as being 2.5-fold those of grass.

Estimated totals of pollutants removed by coarse (tree) vegetation and extensive greening are summarized in Table 4, and overall totals of air pollutants removed per scenario are summarized in Table 5.

Table 2. Assumed share of tree cover per green infrastructure land-use type.

Land-use type	**Tree cover level (%)**		
	Lower	**Average**	**Upper**
Urban green space[5]	10.0	30.0	50.0
Improved urban green space[6]	60.0	65.0	70.0
Orchards[7]	20.0	43.0	66.0
Forest[8]	66.0	79.5	93.0

Table 3. Pollutant removal rates used.

Pollutant	**Coarse removal rate ($g\ m^{-2}\ a^{-1}$)**	**Averaged from**
SO_2	2.025	Kremer et al. (2016), Jim & Chen (2008)
NO_2	2.137	Kremer et al. (2016), Jim & Chen (2008), Hirabayashi (2021)
PM_{10}	5.321	Kremer et al. (2016), Muresan et al. (2022), Escobedo et al. (2008)

[5] Own assumption
[6] Own assumption based on narrative
[7] Based on figures by O‘Connell & Goodwin, 2005.
[8] Based on figures by Yen & Cochard, 2017.

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Table 4. Amount of pollutants removed, per type of pollutant, tree cover (TC) level, and scenario (Mg per year).

Pollutant	Scenario	By coarse vegetation/by tree cover level			By fine vegetation
		Lower TC	Average TC	Upper TC	
SO_2	A	77.36	106.94	136.53	0.00
	B	77.78	108.20	138.63	0.00
	C	78.34	109.41	140.49	0.37
	D	79.21	111.04	142.86	1.23
NO_2	A	81.63	112.86	144.08	0.00
	B	82.08	114.19	146.30	0.00
	C	82.67	115.46	148.26	0.39
	D	83.59	117.18	150.77	1.30
PM_{10}	A	203.27	281.01	358.75	0.00
	B	204.37	284.32	364.27	0.00
	C	205.84	287.50	369.15	0.98
	D	208.13	291.76	375.40	3.23

Table 5. Total combined amount of pollutants removed, per tree cover (TC) level and scenario (Mg per year).

Scenario	Lower TC	Average TC	Upper TC
A	362.26	500.81	639.36
B	364.22	506.71	649.20
C	368.59	514.12	659.64
D	376.68	525.73	674.78

As shown in Table 5, total combined pollutants removed may be in the range of 362.26 Mg a^{-1} (baseline conditions with lower tree cover assumptions) to 674.78 Mg a^{-1} (scenario D and with upper assumed tree cover level), depending on scenario and thus, degree of proposed interventions, and on assumed share of tree cover for corresponding land-use types (Figure 14).

In line with this, overall, high potential for the regulation of air quality is seen as tied to forest land (Figure 13). High potential is also tied to improved urban green spaces, due to their intended high share of woody vegetation in line with narratives, and through foreseen hedges as close-to-source greening measures to promote dry deposition (Janhäll, 2015). Moderate potential is tied to urban green spaces and orchards, due to their comparatively lower assumed tree cover. Lowest potential is attributed to extensive greening in the form of meadows (Figure

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13). However, considerable uncertainties remain due to unknown share of woody vegetation/tree cover in suggested green spaces, particularly under baseline conditions.

3.2.3 Support and maintenance of biodiversity

Central Vietnam is the most forested region of Vietnam (Yen & Cochard, 2017). Tropical rain forests and mangroves are ecosystems rich in endemism and biodiversity (Yen & Cochard, 2017; Do et al., 2017; Veettil et al., 2019). Therefore, high ecosystem service potential for supporting biodiversity is assumed to be strongly tied to forest land, under the assumption that their level of degradation is low. High potential is also associated with planned afforestation under scenario A and/or scenario D, assuming that afforestation aims at the implementation of diverse and natural forest areas, and therefore, does not aim at the implementation of plantation forest seeking to maximize timber production.

Urban green spaces are also recognized to provide (at least limited) support for urban biodiversity (Fischer et al., 2018; Liang et al., 2023), and provide, e.g., also opportunities for increasing plant community biodiversity or enhancing pollinator habitat (Castelli et al., 2021). Moreover, also avian biodiversity is supported by urban green spaces, with higher biodiversity being associated with larger total area, higher degrees of naturalness, higher heterogeneity, and higher shares of woody vegetation. Additionally, water bodies were found to improve diversity at local scale (Wong et al., 2023; Aida et al., 2016). Therefore, comparatively high potential for biodiversity support is tied to suggested improvements of green spaces, as well as creation of ponds in existing green spaces (Figure 13).

3.2.4 Amenity value and recreation

The importance of accessible and available green spaces for recreation, human health, and well-being is well-recognized in the literature. Urban green spaces with trees and grass promote use of outdoor spaces and provide opportunities for outdoor learning and recreational nature play (Davies et al., 2017). Trees, flower cover and ornamental characteristics were also found to provide aesthetical pleasure and attractiveness, thereby increasing duration of stay and willingness to visit (Davies et al., 2017; Grahn & Stigsdotter, 2010; Hoyle et al., 2018). In addition, amenity value and recreational potential are supported by amenity features such as fountains, that provide a sense of place, as well as facilities, e.g., seating opportunities etc., that accommodate social activities (Grahn & Stigsdotter, 2010). Consequently, here, assuming a selective provision of such amenities, moderate amenity value is already tied to urban green spaces under baseline conditions.

However, it has also been found that structurally more complex green spaces may be preferred by users (Davies et al., 2017), and that blue spaces may further enhance positive impacts to recreation, health, and well-being (Mishra et al., 2020). Hence, due to their intended character as more compositionally diverse woody green spaces, high amenity value is attributed to the improved green space class. This is also true for suggested ponds as additional blue features within green urban areas (Figure 13).

It is furthermore recognized that also forests provide recreative and restorative potential (Simkin et al., 2020). Therefore, high amenity values are also tied to forest land as well as orchards. Finally, extensive greening interventions in the form of meadows may also provide

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considerable amenity value (Figure 13), particularly if realized as structurally more diverse and with a larger number of plant species (Southon et al., 2017).

3.2.5 Synopsis

The previous subsections elaborated potentials of proposed interventions and associated land-use types for selected ecosystem services. In line with these assessments, adapting Hattam et al. (2021) and Schlutow and Schröder (2021), potentials for the delivery of ecosystem services per intervention/land-use type are determined along an ordinal scale, as shown in Figure 13. Here, ordination refers to the ranking of ecosystem service potential to indicate no, low, medium or high potential for the delivery of a given ecosystem service. The ordination considers evapotranspiration and shading as determinants of cooling and regulation of OTC, air purification based on the previous assessment, biodiversity, and amenity value.

Land-use	Evapotranspiration	Shading	Air purification	Biodiversity	Amenity value function
Urban green space (intensive greening)	2	2	2	1	2
Urban green space (improved)	3	3	3	3	3
Extensive greening	3	0	1	2	2
Orchards	3	2	2	3	3
Forest	3	3	3	3	3
Water bodies, incl. Ponds	3	0	0	2	3

Potential for ecosystem service	
0	No potential
1	Low potential
2	Medium potential
3	High potential

Figure 13. Ordinated ecosystem service delivery potential for intervention/land-use types under consideration.

Based on this ecosystem service ordination, at the level of hexagonal tiles, overall potential (thus, importance) *p* of a given tessellation feature for delivering a selected ecosystem service *i* is assessed as:

$$p_i = \sum_{c=1}^{n} (s_c \cdot r_c) \quad (5)$$

where *s* … relative share (%) of land-use/intervention type *c* per hexagon, and *r* … rank of ecosystem service *i* for land-use/intervention type *c*. Then, applying an equal weight to each ecosystem service, the aggregated ordinated ecosystem service potential *P* for all ecosystem services is determined for each hexagon unit as:

$$P = \sum_{i=1}^{n} p_i \quad (6)$$

The so-determined aggregated ordinated ecosystem service potential is shown in Figure 15.

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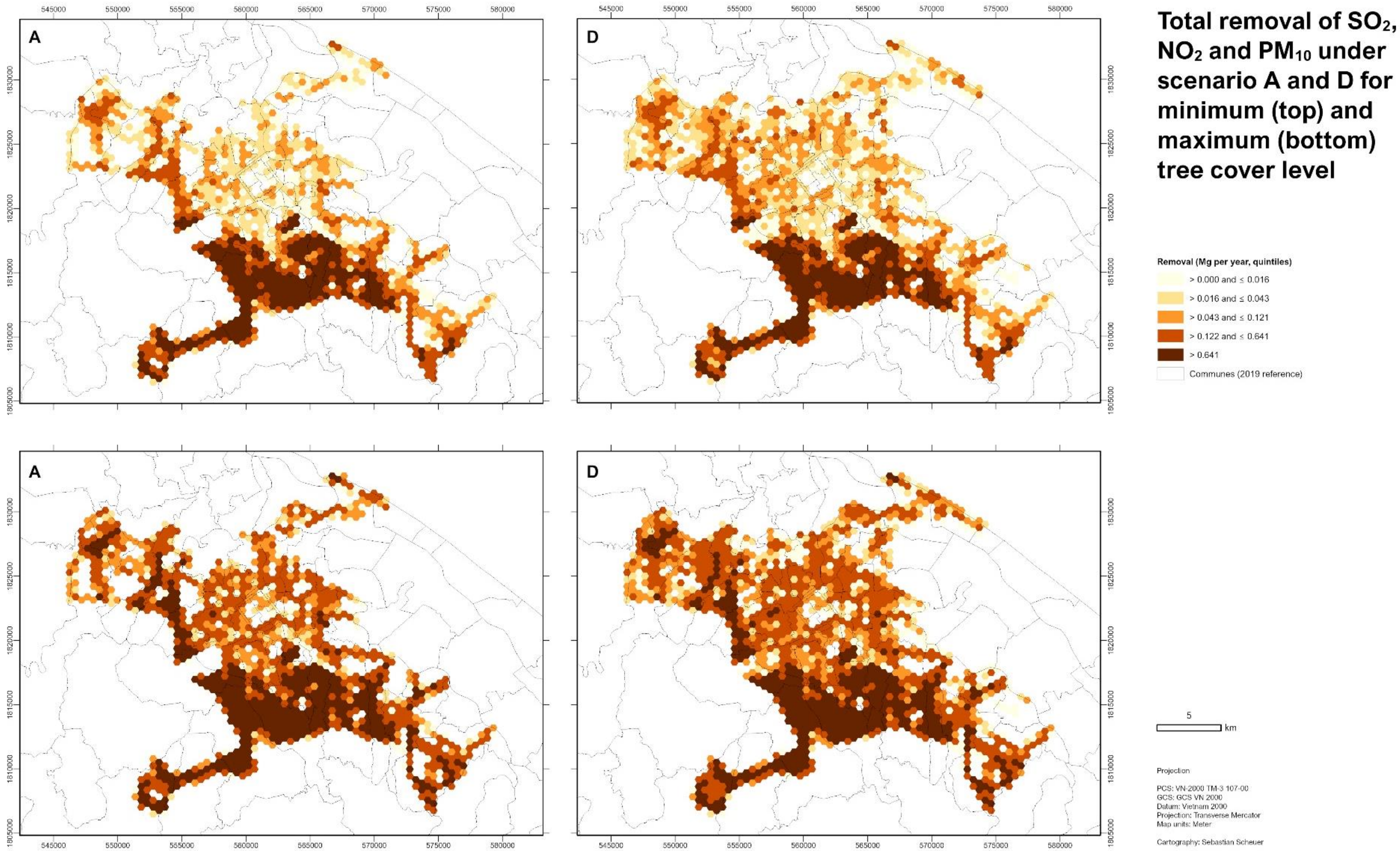

Figure 14. Estimated total removal of pollutants SO2, NO2, and PM10 by considered GBI elements (improved) urban green spaces, extensive greening, orchards, and forests. Estimates are shown for baseline scenario A (left) and scenario D involving highest degree of intervention (right). For either scenario, range of estimated pollutant removal is characterized by tree cover level assumed for the different GBI, i.e., lower tree cover level (top) and upper tree cover level (bottom).

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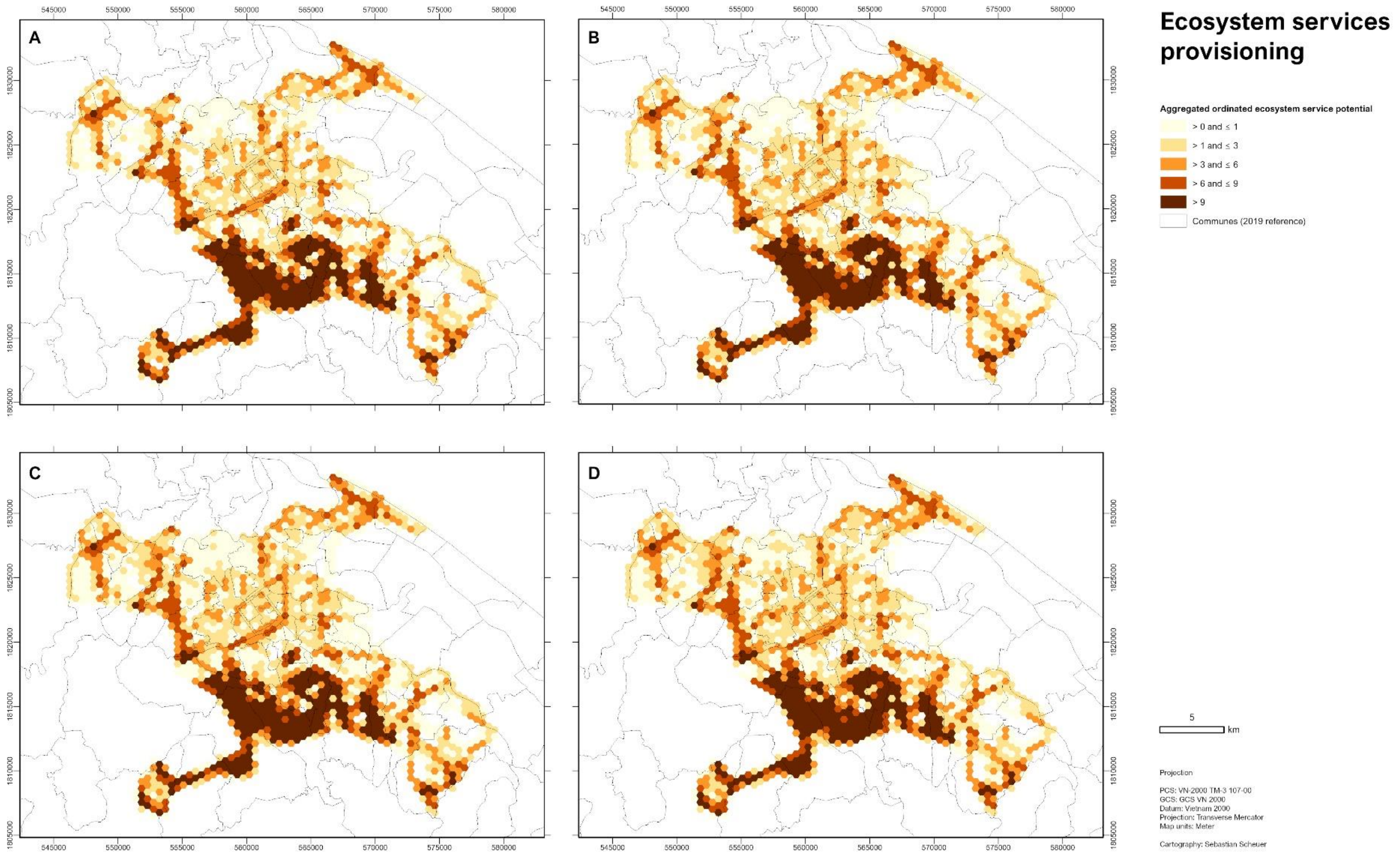

Figure 15. Aggregated ordinated ecosystem service potential for scenario A, B, C, and D.

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Additionally, to support a potential subsequent cost-effectiveness analysis, a valuation of ecosystem services delivery is attempted. This valuation is based on an ecosystem services valuation database by Brander et al. (2023). From this database, ecosystem service values (in international dollars per hectare per year with 2020 price levels, i.e., 2020$ per ha and year) can be retrieved.

Here, ecosystem service values are selected based on the CICES classification of ecosystem services (Haines-Young & Potschin, 2018). Therefore, ecosystem service values are retrieved for CICES classes relevant for the estimation of desired benefits, and at the level of ecozones mapped to suggested interventions/relevant land-use types. All records with location in Asia are considered in the data retrieval, with multiple reported ecosystem service values per CICES class and ecozone being averaged across cases. Not considered were database entries for which the given ecosystem service value cannot be clearly allocated to a given CICES class and/or ecozone, e.g., in case of records that valued multiple, combined ecosystem services.

The following CICES classes are considered, with corresponding ecosystem service values listed in Table 6:

- For the regulation of air temperature and OTC, CICES class 2.2.6.2 (Regulation of temperature and humidity, including ventilation and transpiration). All cases were included with focus on local (micro and meso-scale) climate regulation; cases with a focus on global climate regulation were excluded.
- For the regulation of air quality/air purification, CICES classes 2.1.1.2 (Filtration/sequestration/storage/accumulation by micro-organisms, algae, plants, and animals), 2.1.2.1 (Smell reduction), 2.2.6.1 (Regulation of chemical composition of atmosphere and oceans). For CICES classes 2.1.2.1 and 2.2.6.1, no cases with a focus on local improvement of air quality could be identified in the database;
- For support and maintenance of biodiversity, CICES classes 2.2.2.1 (Pollination or 'gamete' dispersal in a marine context), 2.2.2.2 (Seed dispersal), and 2.2.2.3 (Maintaining nursery populations and habitats including gene pool protection). No information could be retrieved for class 2.2.2.2. For class 2.2.2.3, only cases with a focus on habitat maintenance were considered, and biomass provisioning services excluded; and
- For amenity value and recreation, CICES classes 3.1.1.1 (Characteristics of living systems that that enable activities promoting health, recuperation or enjoyment through active or immersive interactions), 3.1.1.2 (Characteristics of living systems that enable activities promoting health, recuperation or enjoyment through passive or observational interactions), 3.1.2.3 (Characteristics of living systems that are resonant in terms of culture or heritage), 3.1.2.4 (Characteristics of living systems that enable aesthetic experiences), 3.2.1.1 (Elements of living systems that have symbolic meaning), 3.2.1.2 (Elements of living systems that have sacred or religious meaning), and 3.2.1.3 (Elements of living systems used for entertainment or representation). No information could be retrieved for classes 3.1.1.2, 3.1.2.4, 3.2.1.1, 3.2.1.2, and 3.2.1.3.

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Table 6. Ecosystem service values (2020$ per hectare and year) for relevant CICES categories and intervention/land-use types, based on Brander et al. (2023). Entries "-" denote that no ecosystem service value could be determined from the valuation database.

Ecosystem service benefit	**CICES class**	**Intervention/land-use type**			
		Urban green-blue spaces	**Forest land**[9]	**Meadow**[10]	**Water bodies**
Regulation of air temperature and OTC	2.2.6.2	2101.71		-	1904.37
Regulation of air quality/air purification	2.1.1.2	-	344.33	-	-
Support and maintenance of biodiversity	2.2.2.1	-	57.08	-	-
	2.2.2.3	-	579.10	348.41	-
Amenity value and recreation	3.1.1.1	-	32.05	1.45	-
	3.1.2.3	-	0.02	-	-

If a desired benefit is related to more than one CICES class, and therefore, if more than one ecosystem service value could be elicited, corresponding ecosystem service values are summed and then multiplied with the scenario-specific area (in ha) of corresponding intervention/land-use types. Regarding corresponding interventions/land-use types, here, reported values for regulation of air temperature and OTC by urban green spaces are additionally transferred to improved green spaces in scenarios C and D; and values for regulation of air quality/air purification as well as amenity value and recreation by forest land are transferred to improved urban green spaces, due to their intended character as features with high share of tree cover.

The so-determined ecosystem service values (2020$ per year) are summarized in Table 7 per assessed benefit and per scenario, and the overall total values delivered by GBI, per scenario, are listed in Table 8.

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[9] Adapted from or averaged across categories "Tropical and subtropical dry forests", "Tropical and subtropical lowland rain forests", "Tropical and subtropical lowland rain forests; Peatlands", "Tropical and subtropical lowland rain forests; Tropical and subtropical dry forests", "Tropical and subtropical lowland rain forests; Tropical and subtropical mountain forests", "Temperate rain or evergreen forests"
[10] Adapted from or averaged across categories "Tropical and sub-tropical grasslands" and "Derived semi-natural pastures and old fields"

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Table 7. Total ecosystem service value (2020$ per year) for selected ecosystem services/benefits, based on Brander et al. (2023).

Scenario	**Valuation of ecosystem service benefit**				
	Regulation of air temperature and OTC		**Regulation of air quality/air purification**	**Support and maintenance of biodiversity**	**Amenity value and recreation**
	Excluding water bodies	**Including water bodies**			
A	7,956,194	13,599,046	1,795,470	3,317,288	167,225
B	8,392,396	14,035,248	1,795,470	3,317,288	167,225
C	8,517,038	14,209,902	1,797,734	3,333,320	167,436
D	8,833,367	14,590,674	1,811,266	3,391,619	168,697

As shown in Table 8, annually and only regarding considered intervention/land-use types for which distinct ecosystem service values could be identified, ecosystems/GBI elements may provide services/benefits valued in the range from 13.2 Mio 2020$ (baseline scenario A, excluding water bodies) to 14.2 Mio 2020$ (scenario D, excluding water bodies). Water bodies may contribute additional services valued in the range of about 5.6 Mio 2020$ (baseline scenario A) to nearly 5.8 Mio 2020$ (scenario D).

Table 8. Total value (2020$ per year) of ecosystem services provided by considered intervention/land-use types.

Scenario	**Total ecosystem service value**	
	Excluding water bodies	**Value provided by water bodies**
A	13,236,177	5,642,852
B	13,672,379	5,642,852
C	13,815,529	5,692,863
D	14,204,949	5,757,307

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3.3 Assessment of green space availability

Previously, the importance of GBI elements, or respectively, qualitative characteristics of the associated features for provisioning selected ecosystem services has been elaborated. However, for citizens to benefit from delivered ecosystem services, particularly regarding cultural ecosystem services such as recreation, urban green spaces and/or other elements of the GBI need to be available to potential beneficiaries. Here, urban green space availability is seen to comprise both, green space accessibility and green space provision (Wang et al., 2015).

3.3.1 Green space accessibility

Green space accessibility is a metric of distance or spatial proximity to GBI elements and is an important predictor for restorativeness. For example, Maury-Mora et al. (2022) report that citizens within 300m from urban green spaces are likely to experience less stress than citizens with more-distant green spaces. The adopted hexagonal tiling is considered suitable to reflect on such distance-based recommendations. For population and green spaces located within the same hexagon tile (0-order neighbourhood), the theoretical maximum (edge-to-edge) distance from any point within the tile to a green space located anywhere within the same tile equals about 538 m; from the centre (midpoint to edge), this distance is about 268 m. When considering all adjacent neighbouring tiles (1st-order neighbourhood), the theoretical maximum distance to any green space within this neighbourhood (furthest edge to edge) is about 1612 m; from the centre point to furthest edge, this distance is 806 m, and from centre point to centre point about 538 m. The average distance is 563 m.

The 0-order neighbourhood is used as a proxy for denoting good accessibility to urban green spaces, as the distance to urban green is at maximum 538 m (assuming a direct connectivity). Good accessibility indicates green spaces in close proximity to home, with the threshold being comparable to recommendations by the Senate Department for Urban Development and the Environment of Berlin (2016). Translating distance threshold to walking time, green spaces with good accessibility are assumed to be accessible within a walking time of 7.1 min (assuming a walking pace of 4.5 km per hour for younger adults; cf. Alves et al., 2020; Schimpl et al., 2011; Silva et al., 2006) to 10.5 min (assuming an averaged walking pace of 3.06 km per hour for elderly >65 years of age; ibid.). The corresponding share of estimated 2030 population with good accessibility, i.e., to green-blue spaces within the 0-order neighbourhood, is summarized in Table 9. As shown in Table 9, at a minimum, about 2/3rd of the projected 2030 population in Hue is considered to have good access to green spaces (cf. baseline scenario A). When also considering forests in addition to public urban green spaces, this share for baseline increases to almost 73%, and when also considering water bodies, i.e., any type of GBI, then about 85% of inhabitants are estimated to have good accessibility to green-blue spaces in 2030. Each of these shares is increased considerably, up to about 10%, through the implementation of additional GBI elements in scenarios B, C, and D (Table 9).

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Table 9. Share of population with good access to green-blue infrastructure.

Scenario	Share of population with good access		
	(Public) urban green spaces and green sports facilities	(Public) urban green spaces and green sports facilities, and forests	Any type of green-blue infrastructure, including water bodies
A	67.33	72.86	84.98
B	81.37	84.22	92.56
C	78.84	82.33	93.96
D	79.33	83.13	93.98

Additionally, accessibility is estimated for the 1st-order neighbourhood, therefore, within up to a theoretical (straight-line) maximum distance of 1612 m, however, with centre-to-edge distances being less than 1000 m. This is referred to as having at least reduced accessibility to green-blue elements. This is comparable to the classification of corresponding green spaces as being in close proximity to residential land (The Senate Department for Urban Development and the Environment of Berlin, 2016). The corresponding shares of population with at least reduced accessibility is shown in Table 10. It becomes clear that independent from the type of green-blue features considered, and across all scenarios, more than 9 out of 10 inhabitants are estimated to have at least reduced accessibility to green spaces in 2030. When considering all types of GBI including water bodies, according to scenario B, C, and D, it is estimated that virtually all inhabitants of the case study area will have at least reduced access to green-blue infrastructure elements.

Table 10. Share of population with at least reduced access to green-blue infrastructure.

Scenario	Share of population with at least reduced access		
	(Public) urban green spaces and green sports facilities	(Public) urban green spaces and green sports facilities, and forests	Any type of green-blue infrastructure, including water bodies
A	92.21	94.70	98.76
B	95.06	96.08	> 99.00
C	94.42	95.48	> 99.90
D	95.64	96.92	> 99.90

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3.3.2 Green space provision

Whilst accessibility indicates proximity to urban green spaces, green space provision is an indicator of the amount of green space available to residents. Green space provision is expressed as the area of green space per capita (m² per capita). However, this indicator may be determined for different spatial scales: At the tile level, or at the case study level. At the tile level, green space provision refers to the area of green spaces within 0-order or 1st-order neighbourhood, put in relation to the population estimated for the same spatial extent. It thus assesses green space provision in a spatially explicit manner, thus complementing the green space accessibility metric. Below, a focus is on green spaces with good accessibility, i.e., the 0-order neighbourhood, where higher green space provision, i.e., larger green space per capita, may particularly promote recreation and restorativeness by supporting personal retreat or by promoting active-recreational uses (Grahn & Stigsdotter, 2010; Hartig et al., 1997). At the case study level, green space provision, as an aggregated measure, indicates total amount of green space in the case study area in relation to the total population estimated for 2030. This indicator is particularly suitable to inform on green space provision policy targets.

In the following, the tile-level green space provision is determined for the 0-order neighbourhood, i.e., considering good accessibility. To support this assessment, a classification of green space provision is suggested to indicate tiles with no, poor, reduced, or good green space provision, as follows (adapted from The Senate Department for Urban Development and the Environment of Berlin, 2016):

$$c = \begin{cases} good & if\ a\ \geq t \\ reduced & if\ 0.5t < a < t \\ poor & if\ 0.1 < a \leq 0.5t \\ none & otherwise \end{cases} \tag{7}$$

Where *c* … classified green space provision, *a* … area of green space per capita (m² per capita), and *t* … policy target. Here, a target value of *t*=11 m² per capita was chosen, in line with de la Barrera et al. (2023).

The so-classified green space provision is shown in Figure 17, considering only urban green spaces. Looking at this Figure, it becomes clear that tile-level green space provision is poor particularly in the Citadel area and in the northern and southern case study periphery. It is in these areas where green space areas are lacking under baseline conditions (and where no interventions were proposed in the remaining scenarios). When considering both urban green spaces as well as forests in the assessment of green space provision (Figure 18), green space provision is improved and thus classified asgood especially in the southern parts of the case study region.

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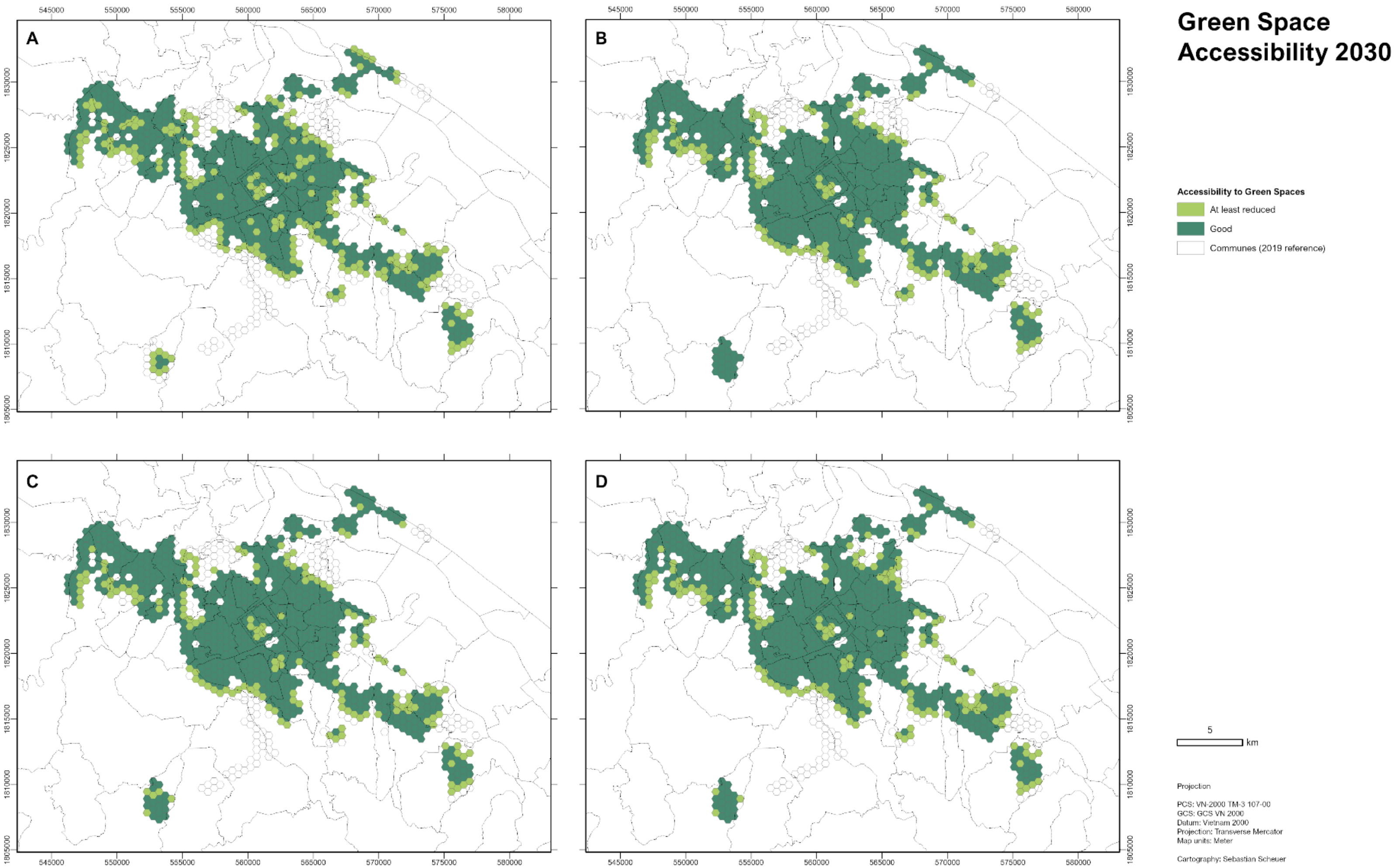

Figure 16. Estimated accessibility to green spaces in 2030, classified as good or at least reduced accessibility. With green spaces implemented in addition to baseline, good accessibility is increased significantly in scenarios B, C, and D compared to scenario A. However, it also needs to be noted that there is a trade-off in the latter scenarios with respect to proposed size of green elements (characterized by their spatial footprint) and resulting accessibility. As spatial footprint is increased from scenario B to D, the number of interventions to achieve desired conversion rates decreases; simultaneously, accessibility may be reduced. This is mainly due to lower density of proposed interventions in scenarios C and D when compared to higher numbers of smaller interventions in scenario B.

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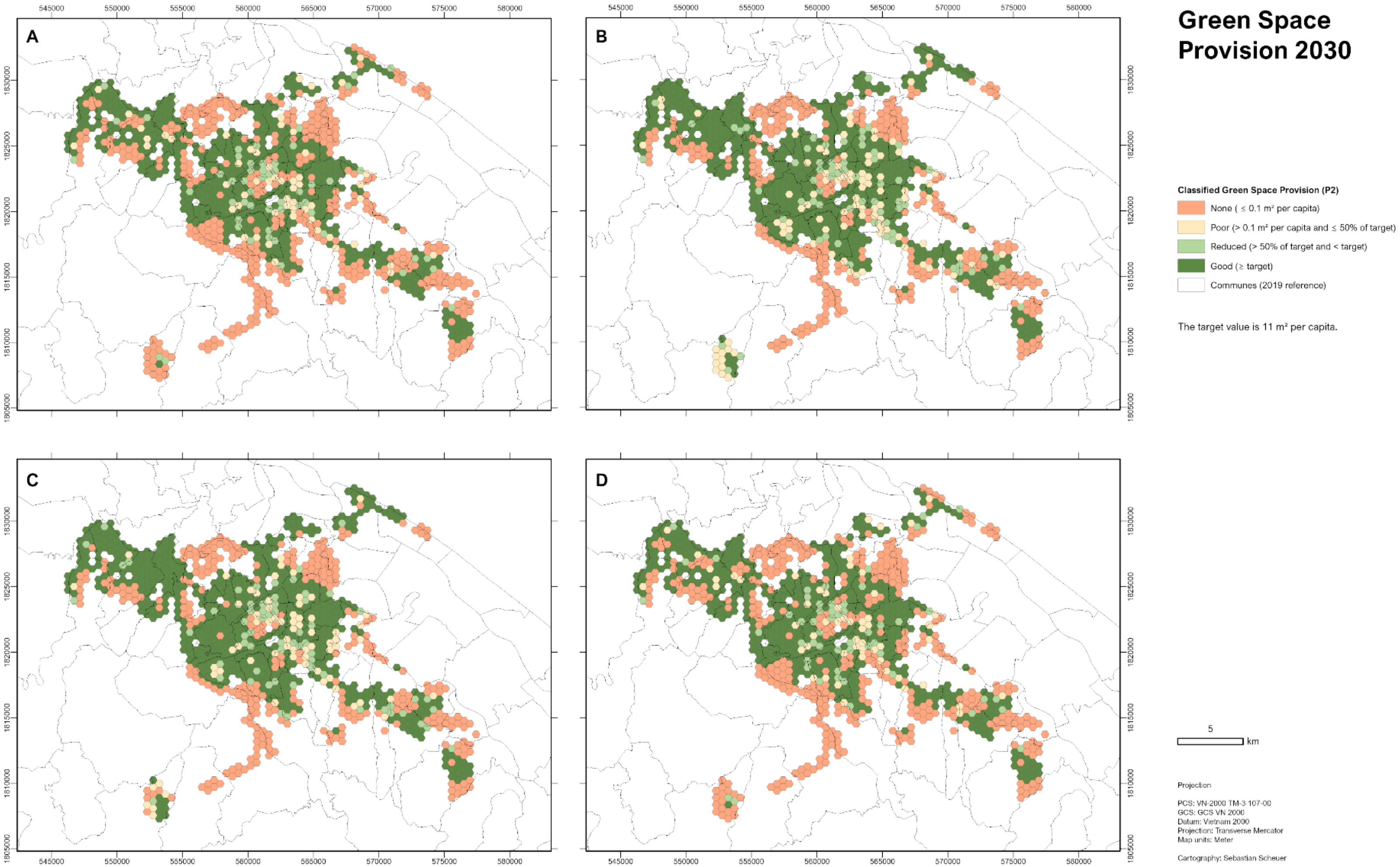

Figure 17. Classified tile-level green space provision (considering only urban green spaces) for scenarios A, B, C, and D.

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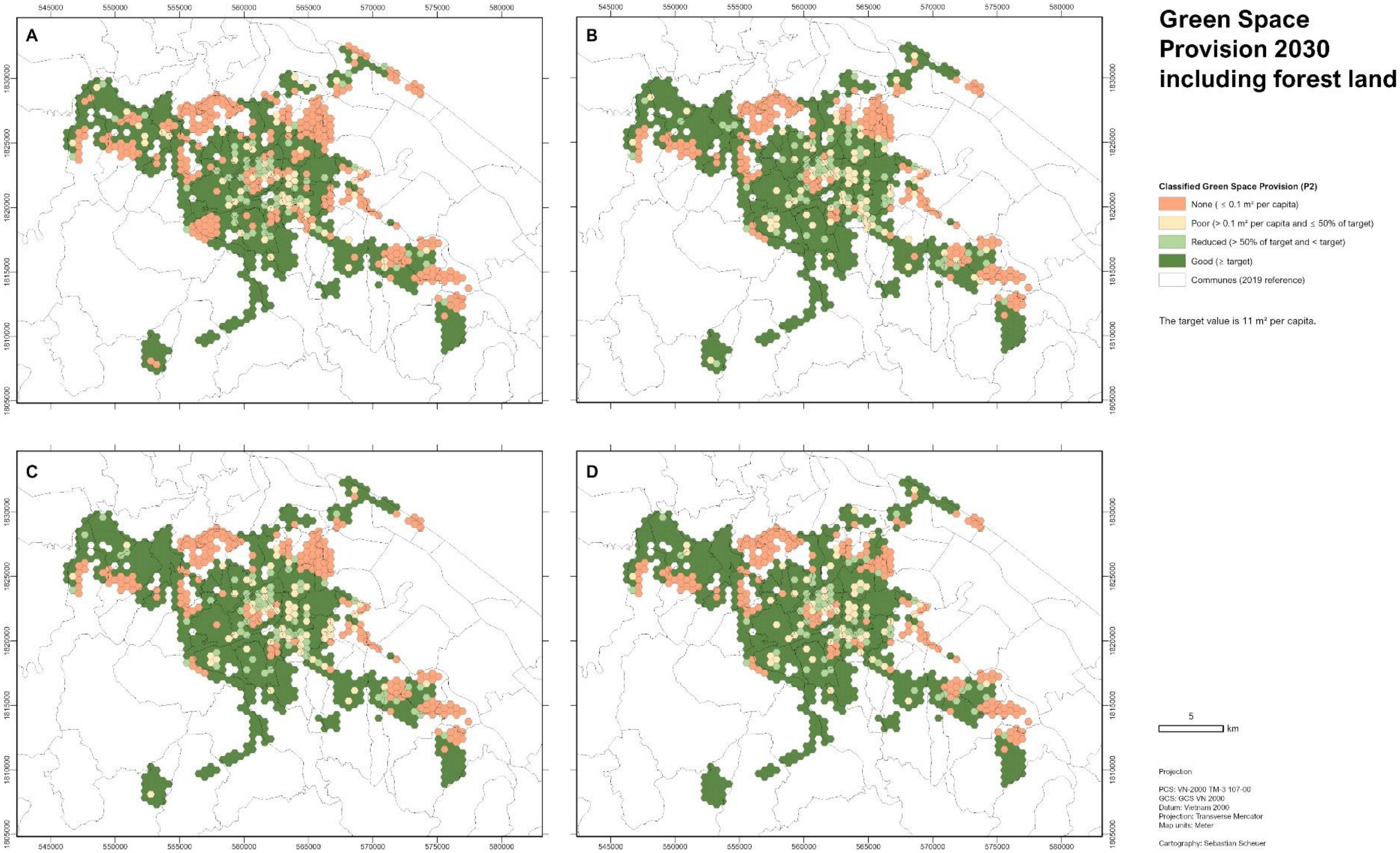

Figure 18. Classified tile-level green space provision (considering urban green spaces and forest land) for scenarios A, B, C, and D.

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Subsequently, green space provision is also assessed at case study level. In so-doing, the overall impact of scenarios at city level may be compared more easily. Overall, at the case study level, green space provision for the status-quo 2019 is equal to 3.22 m²/capita when considering urban green spaces only, and 59.7 m²/capita when also considering forested areas. Under baseline scenario A, the available green space per capita increases drastically, due to the large urban green spaces planned. When considering all (public) green spaces and green sports facilities identified in scenario A, green space provision increases to 55.86 m²/capita under scenario P1, and to 55.2 m²/capita under scenario P2[11]. When also considering forested areas, green space provision under baseline scenario A increases to 132.8 m²/capita (P1) or 131.2 m²/capita (P2), respectively.

Under scenarios B, C, and D, available green space per capita is further increased due to proposed interventions (cf. Figure 19). Under scenario B, an additional 3.06 m²/capita (P1), respectively, 3.03 m²/capita (P2) would be implemented in addition to baseline. Under scenario C, the added green space per capita would equal 6.75 m²/capita (P1) and 6.67 m²/capita (P2), respectively. For scenario D, these values are equal to 11.74 m²/capita (P1) and 11.6 m²/capita (P2). However, in scenario D, also afforestation measures are suggested. When also considering this afforestation action, scenario D adds 12.24 m²/capita (P1) and 12.09 m²/capita (P2), respectively, to baseline.

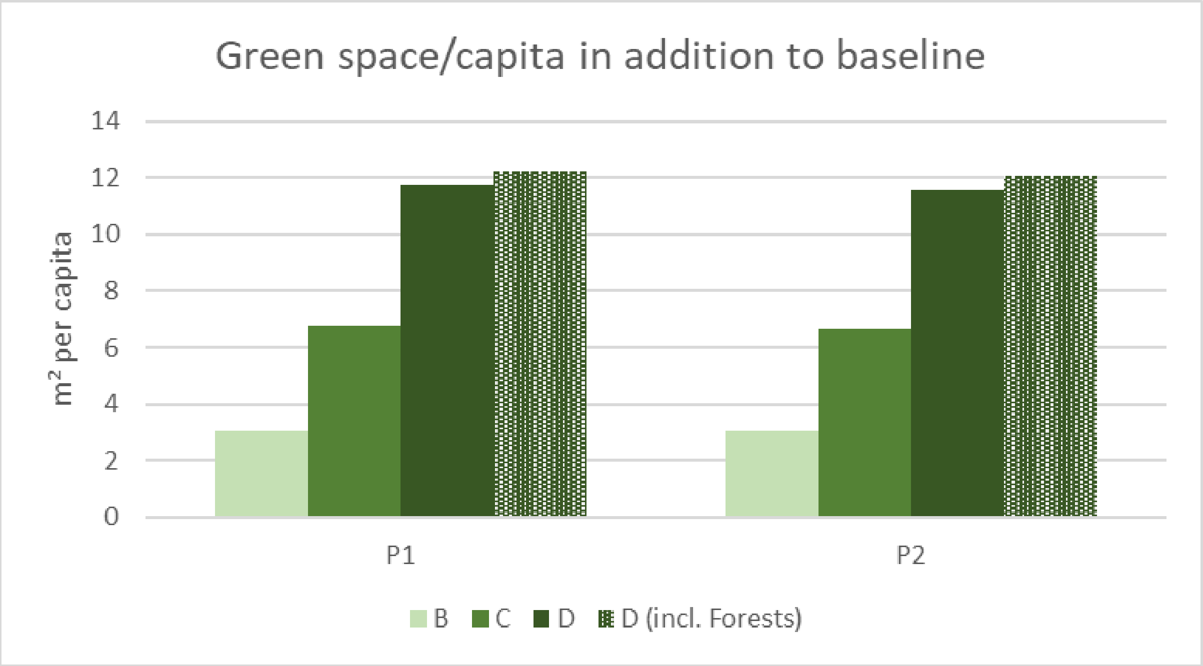

Figure 19. Green space (m² per capita) foreseen in scenarios B to D in addition to baseline (scenario A). For scenario D, additional green spaces and the combined additional green spaces and forests are shown separately. This is due to the fact that afforestation action is only considered in scenario D.

[11] The green space provision value for scenario P2 is slightly lower compared to P1 due to the higher estimated 2030 population

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Table 11. Green space provision (m² per capita) at case study level. For the status-quo (2019) and baseline scenario A, total per-capita green space provision is shown. For scenarios B to D, the amount of per-capita green space that is foreseen under each scenario in addition to baseline is shown. Considered are entities classified as urban green spaces, including suggested blue elements within green spaces, and forests. Water bodies are not considered.

Year	**Green space provision (total, m² per capita)**	
	Urban green space	Urban green space and forests
2019 (Status-quo)	3.22	59.70
2030 (Baseline A)	55.20	131.20
	Green space provision (additional to baseline, m² per capita)	
2030 (B)	3.03	3.03
2030 (C)	6.67	6.67
2030 (D)	11.60	12.09

4 Discussion

In line with scenarios developed and implemented in WP1, the work described in this report assesses the potential impacts of planned and proposed greening interventions at the level of the case study city. Importantly, with the underlying scenarios representing rather potential, non-conclusive and non-exhaustive pathways for enhancing GBI in Hue, it needs to be noted that also this impact assessment remains non-conclusive. However, it elaborates important methodological steps and metrics for assessing potential impacts of scenario-specific future changes in the case study area on selected ecosystem services and green space availability, and exemplifies potential outcomes on the basis of these scenarios.

It has been elaborated that the planned and/or suggested interventions provide various opportunities for the delivery of ecosystem service benefits. However, the uncertain design of certain GBI particularly under baseline scenario A also poses challenges for the assessment of ecosystem services. This is also true for the ecosystem services provided by green elements within existing or planned urban built-up, where gaps of knowledge remain regarding the extent of integration of certain GBI elements, particularly trees. Consequently, this deliverable focuses exclusively on Hue's GBI in the context of planned or proposed interventions, and associated land-use types. In so-doing, a potential minimum level of ecosystem services provided is assessed, with actual provisioning likely being higher if built-up land/remaining land-uses would also be considered.

However, due to the aforementioned uncertainties, specific impacts are challenging to assess for most of the ecosystem services in focus. In case of regulation of air temperature and OTC, this is particularly due to the highly local impacts of greening interventions, and the manifold mediating factors affecting the delivery of cooling. These factors include, e.g., urban morphology, choice and density of green-blue infrastructure elements, choice of tree species, planting patterns and orientation within street canyons or green spaces, but also include meteorological conditions and climatic background (Fu et al., 2022; Graça et al., 2022; Koc et al., 2018; Kong et al., 2014). For example, although Andersson et al. (2020) report a

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"coolscape" extending up to 500m from GBI depending on the density of urban built-up for a temperate city, direct transferability of this finding to the local, i.e., (sub-)tropical climate context remains uncertain. Findings by Scheuer et al. (in preparation) suggest that likely, such a coolscape may be more localized in Hue, particularly in the case of street-level interventions (e.g., planting of street trees), for which especially the importance of shading, i.e., regulation of solar radiation, has been shown. This is likely due to local climate and therefore, unfavourable meteorological conditions with respect to OTC due to high solar radiation, high air temperature, high humidity, and low wind speed, resulting in widespread demand for the improvement of outdoor thermal comfort. The presented approach proposes an ordination of intervention- or land-use-specific potential for providing evapotranspiration and shading, as proxy for cooling and improvement of OTC, in order to indicate potential for the delivery of this benefit. Here, ordinated potentials are considered a function of: (i) assumed evaporative potential, that is generally considered as high for GBI elements, and that—irrespective of water availability—is limited by the presence of nonpermeable surfaces, e.g., as part of urban green space landscaping; and (ii) the potential to provide shading, as a function of tree cover. Impacts on air temperature and OTC may be uncovered more specifically within the assessed intervention types and associated land-use classes by an evaluation of site-level models (WP2, T2.2). However, it also needs to be noted that under local conditions, potentially adverse impacts may be associated with greening interventions. For example, although ponds were found by Ngo et al. (2022) to effectively reduce air temperature, OTC may be negatively affected. This is also true for tree-based interventions, which may adversely impact OTC particularly during daytime margins and at night, due to the trapping of heat under tree canopies and blocking of wind ventilation (Coutts et al., 2016; Gál et al., 2021). Again, such adverse impacts may be uncovered more specifically by an evaluation of site-level models.

Regarding the regulation of air quality/air purification, the importance of trees but also grassy surfaces has been identified in the literature, and potential of green-blue infrastructure elements for removing gaseous air pollutants (including SO_2, NO_2, O_3) as well as suspended particulate matter (including PM_{10}, $PM_{2.5}$) have been reported. Here, for the city-scale, regulation of air quality has been determined particularly based on tree cover as coarse vegetation that features comparatively high pollutant removal potential. However, specifically in the case of meadows, potential of finer vegetation has also been considered. As the exact design of greening interventions under all scenarios remains uncertain, ranges of potential tree cover share have been assumed for urban green spaces under baseline conditions, and for improved (woody) urban green spaces in line with existing narratives. For orchards and forests, tree cover is assumed based on the literature (O'Connell & Goodwin, 2005; Yen & Cochard, 2017). Removal of air pollutants was then determined based on averaged rates reported in the literature. Locally more-specific dry deposition rates could not be determined due to lack of data on local air quality and thus, ambient concentration of air pollutants.

Regarding the support and maintenance of biodiversity, this analysis particularly reflects on the importance of forest for biodiversity, in line with Yen & Cochard (2017), Do et al. (2017) and Veettil et al. (2019). The assessment of support and maintenance of biodiversity as ecosystem service is then primarily based on ordinated potential of each land-use/intervention, as well as area of each land-use/intervention at the tile level, hence reflecting on patch capacity. However, also connectivity of these patches and thus, green-blue infrastructure as a whole to counter habitat fragmentation, play a crucial role for biodiversity (Donati et al., 2022). However, assessing connectivity remains challenging due to the virtual allocation of interventions within potentially large spatial units.

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Regarding amenity value and recreation, amenity features promoting sense of place, aesthetical pleasure, engagement with nature particularly among children, and social-cultural experiences are found to play a considerable role in the promotion of urban green space use (de la Barrera et al., 2023; Grahn & Stigsdotter, 2010). Moreover, naturalness, structural complexity and tree cover were identified as important determinants of recreative and restorative potential (Davies et al., 2017; Simkin et al., 2020). Ordinated ecosystem service potentials reflect on these findings. It however needs to be noted that locally, preferences of citizens may differ from reported traits with respect to optimal design of green spaces, i.e., questions need to be raised whether amenity values commonly reported in the literature are readily transferable to the case study setting. For example, well-maintained parks may be preferred to more natural, structurally more-complex appearances, which in turn may pose barriers for users of green urban features in the local, cultural context. It is suggested that the GreenCityLabHuế project attempts to elicit such local preferences more explicitly as part of WP3, in order to align best practices and development of the Green Vision Huế with citizen demands, preferences, and acceptance towards traits of GBI elements.

Furthermore, green space availability, here understood as subsuming both green space accessibility and green space provision, is a recognized enabler of recreation and further cultural ecosystem services to promote human health and well-being. For example, the UN's sustainability goals target the provision of inclusive green spaces with safe and universal access particularly to women and children, the elderly, and persons with disabilities (SDG 11; de la Barrera et al., 2023). As described above, accessibility has been assessed on the basis of tessellation units. This is particularly due to the virtual allocation of greening interventions in scenarios C-D, that challenges a buffer- or walking distance/time-based assessment of accessibility. The latter is also prevented by the lack of available street- or path layouts for 2030. However, the resulting tessellation-based within-tile walking distances denoting good accessibility are seen to adequately approximate recommendations for green space proximity: Within a given tile, maximum distance to green spaces is up to about 538m, however, centre-to-edge distances are less than 300m. Therefore, it is assumed that proximity to GBI in the good accessibility-category are well in line with recommended proximities to urban green spaces (Schindler et al., 2022; WHO, 2016). It has then been shown that already under baseline conditions, about two-third (only urban green spaces) up to three-quarter (urban green spaces and forests) of the population expected for 2030 benefit from good accessibility to GBI.

The second component of green space availability, i.e., green space provision, has also been assessed at tile-level (0-order neighbourhood), i.e., assuming good accessibility to GBI. Again, with respect to tile dimensions and (straight-line) within-tile distances, the corresponding population intake area for assessing green space provision corresponds well to recommendations (The Senate Department for Urban Development and the Environment of Berlin, 2016). At the case study level, overall, provision of green spaces with good accessibility amounts to 38.7 m²/capita under baseline scenario A; for scenarios B, C, and D, this value would be equal to 41.5, 42.6, and 45 m²/capita, respectively. When also considering forest land, this green space provision is considerably higher, ranging from 61.09 m²/capita (A) to 67.82 m²/capita (D).

It has also been shown that urban green spaces remain comparatively scarce under all scenarios in the case study periphery. Importantly, it is particularly here where forests may help compensate for a lack of urban green spaces. However, accessibility of forest land, and

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its relevance for recreation and restoration remain uncertain. It is proposed to further investigate the role of local forests for these ecosystem services.

It also becomes clear that green space accessibility denotes a nexus point of both, the supply side—i.e., of GBI—and demand, i.e., citizens of the case study area as the potential users of urban green spaces and other types of GBI, and as potential beneficiaries of the ecosystem services provided by them. More specifically, the demand side has been estimated as population within the case study area for the year 2030. This modelling of population has been conducted in a three-step process. First, the population for the status-quo year 2019 has been spatially disaggregated to both rural and urban residential land within the case study area. Second, based on this 2019 population total, the population for 2030 has been estimated. Two methods were suggested to do so, due to a lack of available, more-specific data at the case study area level. First, a year-on-year change of population was estimated based on population estimates at the level of the Thua Thien Huế province, and subsequently transferred to individual districts. In so doing, the population trend forecasted for the province was mirrored to the districts relevant for the case study area, thereby projecting the known 2019 population figures forward until 2030 (scenario P1). An alternative method suggests the transfer of province-level trend to the case study area as a whole (scenario P2). In so-doing, an overall population increase of 9.1% (P1) and 10.3% (P2) has been projected from 2019 to 2030. Third, through a set of transfer rules that also reflect on underlying land-use changes 2019 to 2030, the estimated 2030 total population has been spatially disaggregated.

There are however numerous uncertainties with respect to the suggested city-level impact assessment methodology. Regarding the assessment of ecosystem services supply, considerable uncertainties remain in the ecosystem service-specific value transfer to the local context. On the one hand, this is related to the transfer of actual ecosystem service potentials, e.g., pollutant removal rates, and is also due to the unclear (design) characteristics of GBI to be implemented particularly under baseline scenario A, e.g., in terms of morphological composition/tree cover or share of woody vegetation and overall vegetation complexity, choice of tree species, share of pervious or impervious areas, management intensity, etc. These characteristics are considered important determinants for ecosystem service provisioning (Vieira et al., 2018), particularly for regulating ecosystem services including cooling, the improvement of air quality, or the improvement/maintenance of biodiversity. Although a focus was put on determining these characteristics in the second stakeholder workshop, required information could not be satisfactorily elicited from participating stakeholders. On the other hand, this concerns the valuation of ecosystem services. In this regard, few if any ecosystem service values could be identified in the database by Brander et al. (2023) for the relevant CICES classes and for the local context. Consequently, records from across Asia were considered, thus averaging reported ecosystem service values. However, even then, significant knowledge gaps remain regarding the valuation of the selected ecosystem services across the various interventions/land-use types. The conducted ecosystem services valuation thus remains partial and incomplete. Similarly, also the assessment of ecosystem services may be underestimated, as ecosystem services delivered by green elements within the built fabric were not considered in this case study. This is, e.g., due to lack of tree cadastral data, that allows determining street tree cover. Similarly, benefits provided by green elements within courtyards, allotments, etc. were also excluded from this analysis.

Regarding the demand-side assessment, uncertainties particularly stem from the estimation of 2030 population based on transfer of a uniform trend from the province level to the level of heterogeneous spatial units (districts). Despite the manifold uncertainties involved in doing so,

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this has been conducted due to a lack of available population projections at sub-provincial level. In contrast to data by the Statistical Office of Vietnam (2020b) for the province level, only for the city of Hue itself could population projections be identified, with a city-level 2030 population estimate of 501,000 inhabitants (United Nations Department of Economic and Social Affairs, 2019). With a population increase of approx. 27.8% from 2019 to 2030, this estimate is considerably higher than modelled. However, as no estimates are available for other administrative units within the case study area, it remains uncertain how these projections relate to these districts. Further uncertainties result from assumptions for the spatial disaggregation of population. Here, non-residential but potentially mixed-use areas, e.g., ecotourism land or commercial areas were not considered for disaggregation. Moreover, relative densities estimated for rural and urban residential land-use classes are both spatially and temporally homogeneous, thereby neglecting potential differences in density along urban-rural gradients, between core city and periphery, or between existing and planned residential land to be developed until 2030.

5 Relationship of the work described with other tasks and work packages in the project

The impact assessment, respectively, the supply- and demand-side modelling at city level described in this report is based on work conducted in WP1 (Modelling the implementation of land-use changes and nature-based solutions: Land-use change modelling at city-level, cf. Scheuer et al., 2022). Together with the impact assessment at site level (WP2, T2.2), findings provide a basis for the elaboration of best practice guidelines (WP4), that will in turn inform the development of the Green Vision Huế. (WP5). Furthermore, it is suggested that identified needs for further research in the local-cultural context, e.g., regarding characteristics of proposed urban green spaces under baseline conditions as well as contrasting citizen demands and preferences, are considered in the ongoing work in WP3.

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CONCLUSION

The work described in this report presents a methodology for estimating potential impacts of GBI interventions. It is based on the scenarios implemented in WP1. Therefore, potential impacts of interventions suggested in these scenarios as pathways for a greener city are illustrated at the city level.

The described work is conceptually aligned to an assessment of supply and demand. Here, demand refers to the spatially explicit modelling of population projected for 2030 as the potential beneficiaries of GBI benefits. Supply-side modelling focuses more closely on the assessment of the potential benefits of GBI interventions in form of selected ecosystem services, including, e.g., tree cover-based removal of air pollutants. Moreover, impacts on green space availability, here subsuming both green space accessibility and green space provision, are estimated. It is in this regard where the nexus between supply and demand becomes evident.

It has been shown that scenario-based GBI interventions may have considerable supply-side impacts. However, it has also been discussed that overall, the assessment made in this WP remains challenging, often due to insufficient knowledge about baseline conditions, e.g., regarding desired design characteristics of planned urban green spaces. In the following development of best practices recommendations and accordingly, in the Green Vision Huế., it is consequently suggested to revisit identified knowledge gaps.

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