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How Many Small Agglomerations Do Exist in the European Union, and How Should We Treat Their Wastewater?

Alberto Pistocchi · Vanessa Parravicini · Guenter Langergraber · Fabio Masi

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Abstract The European Union (EU)'s legislation on urban wastewater requires all agglomerations with a population equivalent (PE) above 2000 people to undergo a secondary (mechanical/physical and biological) wastewater treatment. Agglomerations below 2000 PE, though, fall outside the scope of the current EU's legislation. As such, their regulation is heterogeneous across the various EU member states, and there is no systematic collection nor reporting of data enabling an estimation of their actual significance as a source of pollution for the receiving water bodies. Here we present a spatial model to delineate agglomerations in a GIS, based on population distribution and land cover. From the model results, in the EU, we identify 364,650 agglomerations with 2000 PE or less, housing a cumulative population of about 75 million inhabitants. We then calculate the organic matter and nutrient loads these agglomerations can discharge, assuming they presently undergo primary wastewater treatment, and the reduction of loads that can be expected under different treatment scenarios, together with the corresponding treatment costs based on a simple cost model. Using a conventional shadow price for the organic matter and nutrients removed, we show that all treatment scenarios show a benefit-to-cost ratio (B/C) above (or close to) 1. However, only a scenario of secondary treatment applied to all agglomerations above 1000 PE provides sufficient safety margins on the B/C. This suggests the opportunity to expand the scope of the current legislation down to agglomerations of this size, while addressing smaller agglomerations depending on their actual impacts on the receiving water bodies, through "appropriate treatments" defined by the local authorities.

Keywords Wastewater treatment · Small agglomerations · Spatial model · Benefit-to-cost ratio · European Union

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A. Pistocchi (✉)
European Commission Joint Research Centre, Ispra, Italy
e-mail: Alberto.pistocchi@ec.europa.eu

V. Parravicini
TU Wien/Institut Für Wassergüte Und Ressourcenmanagement, Vienna, Austria
e-mail: vparravi@iwag.tuwien.ac.at

G. Langergraber
Department of Water, Atmosphere and Environment, Institute of Sanitary Engineering and Water Pollution Control, University of Natural Resources and Life Sciences, Vienna (BOKU), Vienna, Austria
e-mail: guenter.langergraber@boku.ac.at

F. Masi
IRIDRA Srl, Florence, Italy
e-mail: masi@iridra.com

1 Introduction

The European Union (EU)'s Urban Wastewater Treatment Directive (UWWTD) 91/271/EEC sets obligations to collect and treat wastewater for agglomerations with 2000 population equivalents (PE) or more. Smaller agglomerations (less than 2000 PE) are not regulated, although Art. 7 of the UWWTD requires member states to implement an appropriate wastewater treatment for agglomerations below 2000 PE served by a sewer network. In all other cases, member states are not due to act nor report on the matter in the absence of well-identified impacts on the receiving water bodies. Because of this, we have limited evidence in order to quantify the pollution coming from small agglomerations at European scale. In this contribution, we present an estimation of the population living in agglomerations smaller than 2000 PE, the associated loads of pollutants and the costs of treatment under scenarios where we extend the obligations of the UWWTD to agglomerations of less than 2000 PE. Based on the results, we propose recommendations for the regulation of small agglomerations in the EU.

2 Materials and Methods

2.1 Delineation of Small Agglomerations

As a first step in the analysis, we generate a distribution of agglomerations statistically consistent with the agglomerations delineated by the EU member states compliant with the UWWTD. To this end, we examine the spatial distribution of population in Europe and we group it into agglomerations through a mathematical morphology algorithm. We refer to the 100-m population density map of Europe of Freire et al., (2016). We denote this map as Pop, and we process it through the following steps:

1. We define a threshold population density $\boldsymbol{\rho_0}$, above which we assume collection of wastewater is justified, and generate a Boolean map $C_0 = \begin{cases} 1, \text{ if Pop} \geq \rho_0 \\ 0, \text{ if Pop} < \rho_0 \end{cases}$.
2. We expand the zones where $C_0 = 1$ by a number of cells $\boldsymbol{n}$. We call the resulting map C1.
3. We apply a clumping (or region-grouping) algorithm to map C1 (Pistocchi, 2014). We consider the regions grouped with this operation as agglomerations, and we assign each of them a univocal identifier.
4. For each of these simulated agglomerations, we can compute the total population from map Pop, using a zonal statistics algorithm.

All the above calculations are performed using ESRI ArcGIS 10.7 Spatial Analyst software.

The steps of the procedure are illustrated in Fig. 1. The procedure yields different results depending on the parameters $\boldsymbol{n}$ and $\boldsymbol{\rho_0}$. These were calibrated in order to reproduce as close as possible the reported number of agglomerations and cumulative population by agglomeration size at EU level.

The results of the calibration are provided in Annex 4.. The values finally chosen were $\boldsymbol{n} = 2$, $\boldsymbol{\rho_0} = 10$ persons/ha. *This implies that the smallest possible agglomeration is composed of 10 persons and has an extent of 1 ha*.

2.2 Costs of Wastewater Collection and Treatment

For the sake of a quantification at European scale, we assume a bottom line scenario where all households in agglomerations below 2000 PE are served by a primary treatment, most often consisting of a single household septic tank, with no collection network. Under this assumption, treating wastewater of small agglomerations to the level required by the UWWTD for agglomerations above 2000 PE entails a cost, which will include the collection network and the treatment plant. The cost will generally depend on location-specific factors such as the extent of the network and the design of the plant. As a first approximation, we estimate the additional cost of collecting wastewater in a small agglomeration and treating it to a secondary level (compliant with art. 4 of the UWWTD) as the difference of the cost of the network combined with the cost of a secondary plant and the cost of primary treatment. For the cost of the network and the primary treatment (assumed to coincide with a septic tank), we refer to OECD's FEASIBLE model expenditure functions (COWI, 2010; OECD, 2004) while for the secondary treatment of small agglomerations, we refer to the average of cost functions for various types of treatment wetlands (TWs) described in Pistocchi et al. (2020). TWs are common solutions for small plants, together with other technologies including, e.g. sequential batch reactors (SBR) and other small technical aerobic systems (see, e.g. Langergraber et al., 2018). TWs can be designed to achieve performances at least comparable to biological treatment processes for larger agglomerations (Dotro et al., 2017; Langergraber et al., 2019).

Eventually, the additional combined costs of collection and *secondary* treatment are given by the average of costs functions of TWs, plus the cost function

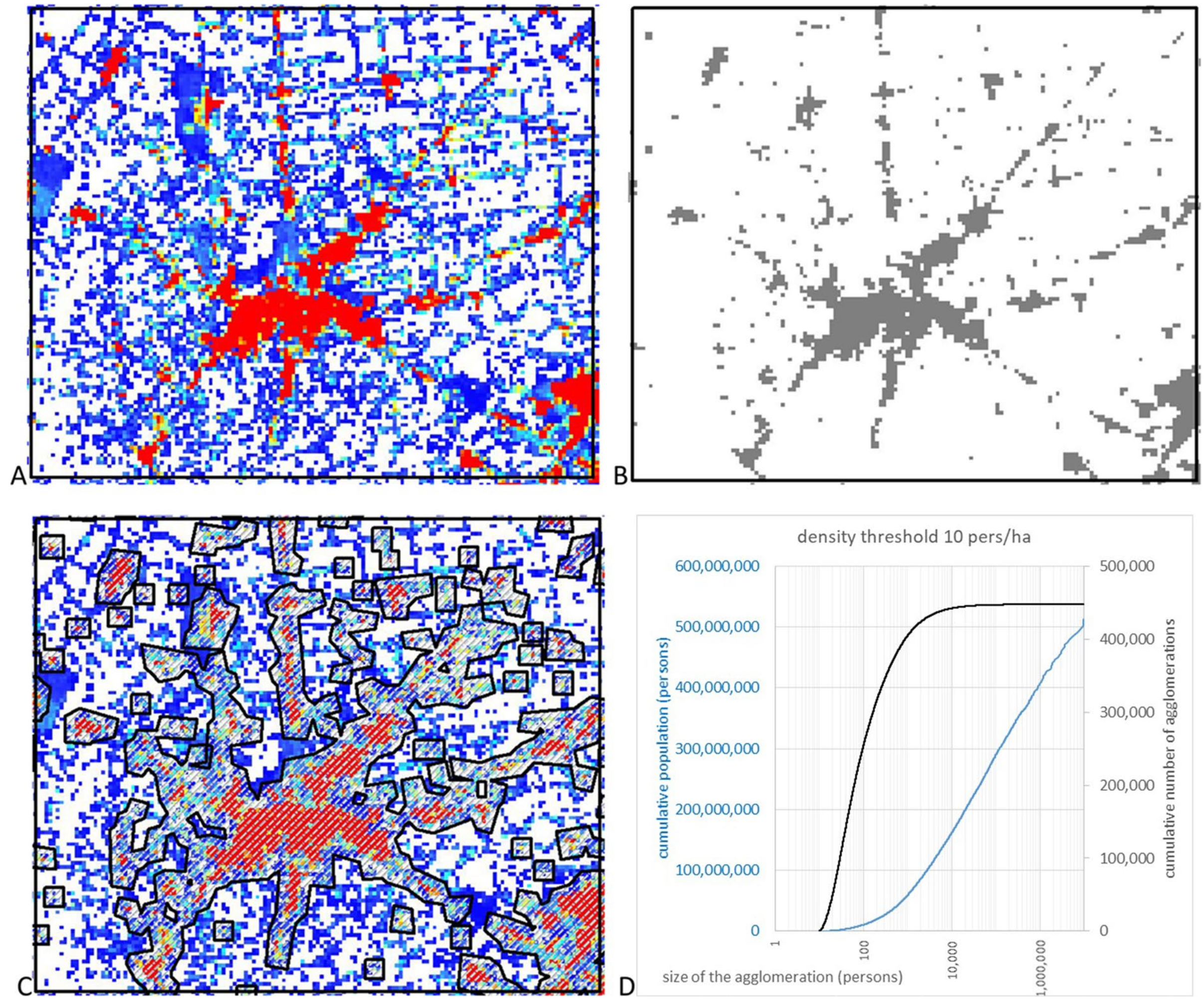

Fig. 1 Example of generation of agglomerations. **A** Map of population density; **B** Map of areas with density ≥ 10 persons/ha; **C** Agglomerations identified with a buffer of 200 m around each continuous zone in map (**B**); **D** European cumulative population and number of agglomerations by increasing agglomeration size

of the collection network, minus the cost function of the septic tank. The combination is described by the following cost function:

$$C_{\mathrm{Cremoval}} = 218.36\mathrm{Pop}^{-0.37}$$

where C_{Cremoval} is the annualised cost (Euro/PE) and Pop the agglomeration's population. All details of the derivation of the above equation are provided in Annex 5..

In addition to secondary treatment, we consider the additional cost of TWs in order to provide denitrification. Assuming that a hybrid TW, providing up to 80% N removal, corresponds to the highest cost function among those described in Pistocchi et al. (2020), such additional cost is estimated as:

$$C_{\mathrm{Nremoval}} = 61.44\ \mathrm{Pop}^{-0.141}$$

where C_{Nremoval} is the annualised cost (Euro/PE) in addition to C_{Cremoval}, in order to have denitrification. All details of the derivation of the above equation are provided in Annex 5.. Finally, for P removal, we refer to the cost function:

$$C_{\mathrm{Premoval}} = 102.10\ \mathrm{Pop}^{-0.315}$$

C_{Premoval} being the additional cost of P removal. The above equation derives from the OECD FEASIBLE model expenditure functions as explained in Annex 5..

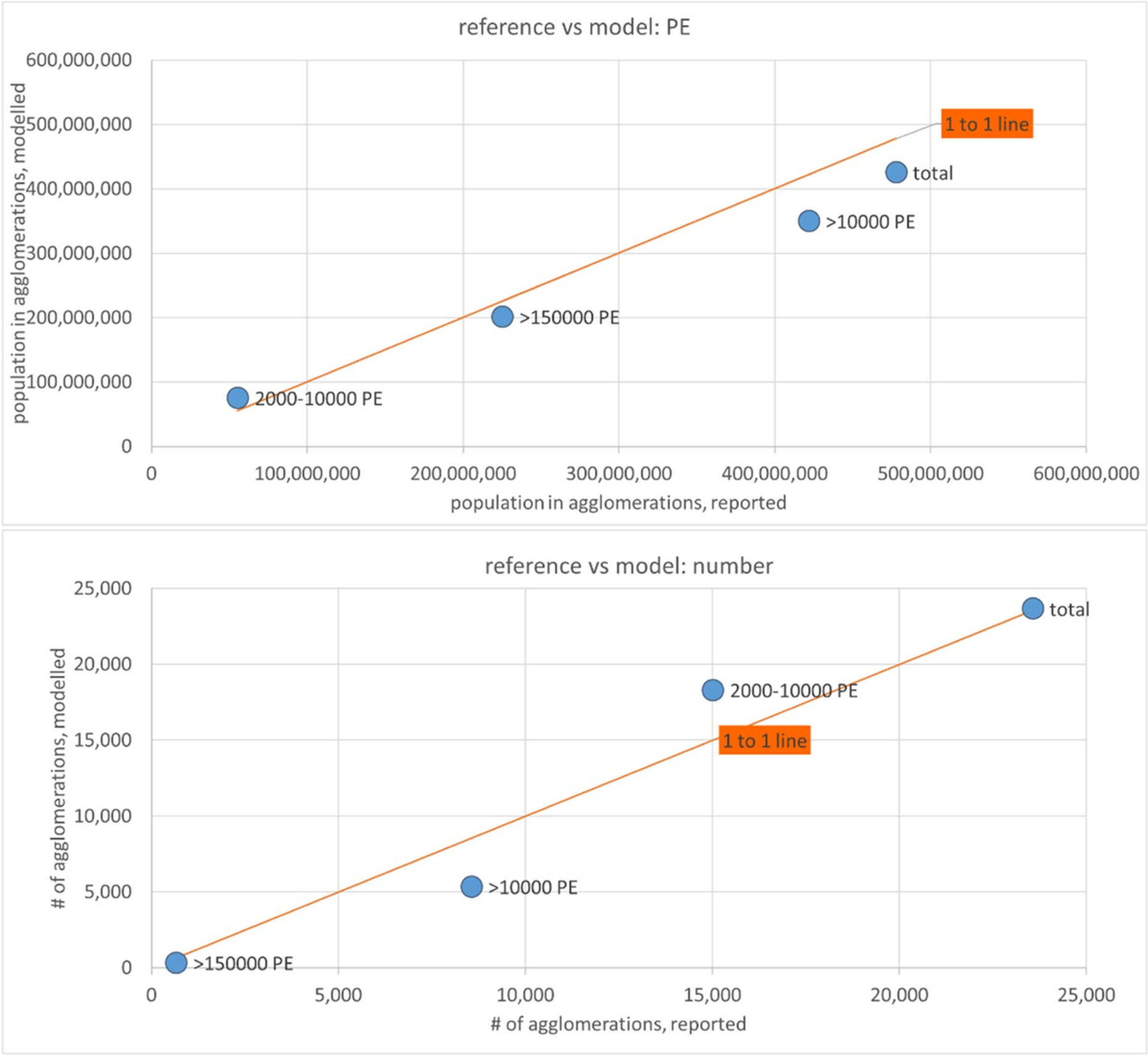

Fig. 2 Comparison of modelled and reported population equivalents (PE) and number of agglomerations for different size classes *above* 2000 PE. Reported data are gathered from the 8th UWWTD Implementation Report. Only for Poland and Italy, and due to their unavailability, the information is based on data gathered from the 7th Implementation Report (https://ec.europa.eu/environment/water/water-urbanwaste/implementation/implementationreports_en.htm)

2.3 Cost-effectiveness Analysis of Treating Small Agglomerations

We assume the load of organic matter (as 5-day biochemical oxygen demand, BOD) associated with untreated wastewater equal to 60 g/PE/day, while those of total nitrogen (N) and total phosphorus (P) are estimated through country-specific emission factors as in Malagò and Bouraoui (2021). In order to compare scenarios, we assume a constant removal efficiency for BOD, N and P for primary, secondary and tertiary (nutrient removal) treatment processes.

The constant removal efficiency is set for primary, secondary and tertiary treatment, respectively, to:

- 50%, 94% and 96% for BOD;
- 25%, 55% and 80% for N;
- 30%, 60% and 90% for P.

Table 1 Modelled population in agglomerations below 2000 PE, for the EU and its 27 member states

Country code	Country	$P >= 1000$ PE	$1000 > P >= 500$ PE	$500 > P >= 100$ PE	$100 > P >= 50$ PE	$P < 50$ PE
AT	Austria	626,980	476,319	623,121	161,575	130,522
BE	Belgium	302,158	224,738	252,388	44,797	40,006
BG	Bulgaria	549,109	441,696	380,966	47,555	25,076
CY	Cyprus	59,667	49,944	51,852	5458	2264
CZ	Czech Rep	732,611	708,391	986,499	156,158	79,469
DE	Germany	4,879,246	3,838,093	4,007,845	640,415	480,872
DK	Denmark	303,446	221,686	267,743	73,311	113,023
EE	Estonia	40,581	36,073	87,505	15,618	19,618
EL	Greece	532,129	570,101	732,387	87,168	46,094
ES	Spain	1,410,037	1,072,767	1,516,409	316,036	196,986
FI	Finland	173,041	126,647	120,795	41,223	55,418
FR	France	3,657,903	3,307,603	4,475,790	1,030,960	982,352
HR	Croatia	254,233	238,719	365,916	88,373	56,760
HU	Hungary	873,472	489,366	384,855	49,509	36,628
IE	Ireland	177,740	138,753	284,710	158,278	243,034
IT	Italy	2,714,161	1,700,447	1,813,088	406,245	303,212
LT	Lithuania	108,735	131,129	270,537	53,861	53,125
LU	Luxembourg	49,598	25,820	38,657	3419	1116
LV	Latvia	80,038	63,104	147,093	26,555	45,472
MT	Malta	1640	1345	938	59	377
NL	Netherlands	443,078	227,126	210,918	75,576	93,239
PL	Poland	1,835,627	2,012,795	4,139,172	922,783	662,855
PT	Portugal	399,152	400,040	674,300	134,266	68,839
RO	Romania	2,091,721	1,492,159	1,202,507	124,657	80,672
SE	Sweden	371,611	294,508	357,273	138,854	243,140
SI	Slovenia	98,317	107,024	176,653	49,834	46,601
SK	Slovakia	686,235	481,446	342,176	29,504	13,011
EU	Total	23,452,264	18,877,839	23,912,093	4,882,047	4,119,794

These values correspond to the assumptions made in the GREEN model used for EU scale assessment of nutrients (Grizzetti et al., 2021; Pistocchi et al., 2019).

While septic tanks may remove virtually no N, we assume their effluents are discharged to small ditches or ponds, or subsurface drainage before reaching the receiving water bodies, and this is sufficient to achieve some natural attenuation of nutrient concentrations.

The removal of N and P assumed for secondary treatment may require already the design of hybrid, multistage TW systems including a P trap, while the removal of N at tertiary level may require a more expensive TW of similar type.

We compute the loads of pollutants discharged by settlements below 2000 PE under scenarios of increasing stringency of treatment, and we compare them with a scenario with all settlements undergoing primary treatment.

3 Results

3.1 Number and Population of Small Agglomerations in the EU

The calibrated model allows delineating agglomerations throughout Europe and an estimation of their population. Figure 1D shows the cumulative frequency distribution of agglomerations by size and the cumulative population as a function of agglomeration size. The model allows reproducing quite well the number and resident population of agglomerations larger than 2000 PE in Europe (Fig. 2). For agglomerations smaller than 2000 PE, we estimate a total population in the EU of about 75 million persons, distributed by country as

Table 2 Modelled number of agglomerations *below* 2000 PE, for the EU and its 27 member states

Country	$P>=1000$ PE	$1000>P>=500$ PE	$500>P>=100$ PE	$100>P>=50$ PE	$P<50$ PE
AT	447	667	2812	2299	4675
BE	214	322	1084	620	1468
BG	397	624	1495	643	878
CY	42	72	210	74	77
CZ	533	1009	4280	2166	2596
DE	3483	5407	16,911	8954	16,930
DK	214	313	1172	1054	4654
EE	30	50	381	218	760
EL	384	816	2954	1203	1499
ES	1000	1511	6827	4421	6651
FI	121	176	531	601	2139
FR	2640	4713	19,901	14,583	36,047
HR	179	345	1668	1245	1841
HU	603	677	1546	692	1311
IE	128	195	1397	2319	9050
IT	1916	2391	7922	5733	10,127
LT	77	190	1188	754	2035
LU	34	37	152	46	36
LV	57	94	614	388	1930
MT	1	2	4	1	11
NL	309	310	960	1099	3477
PL	1329	2914	19,171	12,881	22,838
PT	287	575	2992	1880	2104
RO	1495	2094	4645	1741	2725
SE	256	412	1612	2003	9896
SI	72	155	843	717	1674
SK	486	679	1274	409	447
EU total	16,734	26,750	104,546	68,744	147,876

shown in Table 1, in a total of 364,650 agglomerations distributed by country as shown in Table 2. The number of agglomerations by size is reasonably consistent with a benchmark of independently estimated data (Wood plc, pers.comm., 2022), as shown in Fig. 3. This indicates the simulation model may represent the number of small agglomerations in the EU within a factor 2 of accuracy in more than 75% of the cases and within a factor 10 in all cases, with the exception of Croatia. In this case, modelled agglomerations below 500 PE are a factor > 60 more than reported. We find larger errors on the number of smaller agglomerations and a clear tendency to overestimation (Fig. 3).

The modelled distribution of agglomerations below 2000 PE indicates these house around 15 to 20% of the population in the EU, with some variability among member states (Fig. 4). Population in small agglomerations is a particularly small percentage of the national total in Malta, and below average in Belgium, Spain, Finland and Ireland, while it is higher than average in the Czech Republic, Slovakia, Croatia, Slovenia, Romania and Poland. Usually agglomerations between 1000 and 2000 PE account for 20 to 30% of the population in small agglomerations, while very small agglomerations (below 100 or 50 PE) usually account for 10 to 20%.

3.2 Discharges of Pollutants with Wastewater from Small Agglomerations

We could estimate the discharges of BOD, N and P through wastewater from small agglomerations, based on the emission factors described in Section 2.3

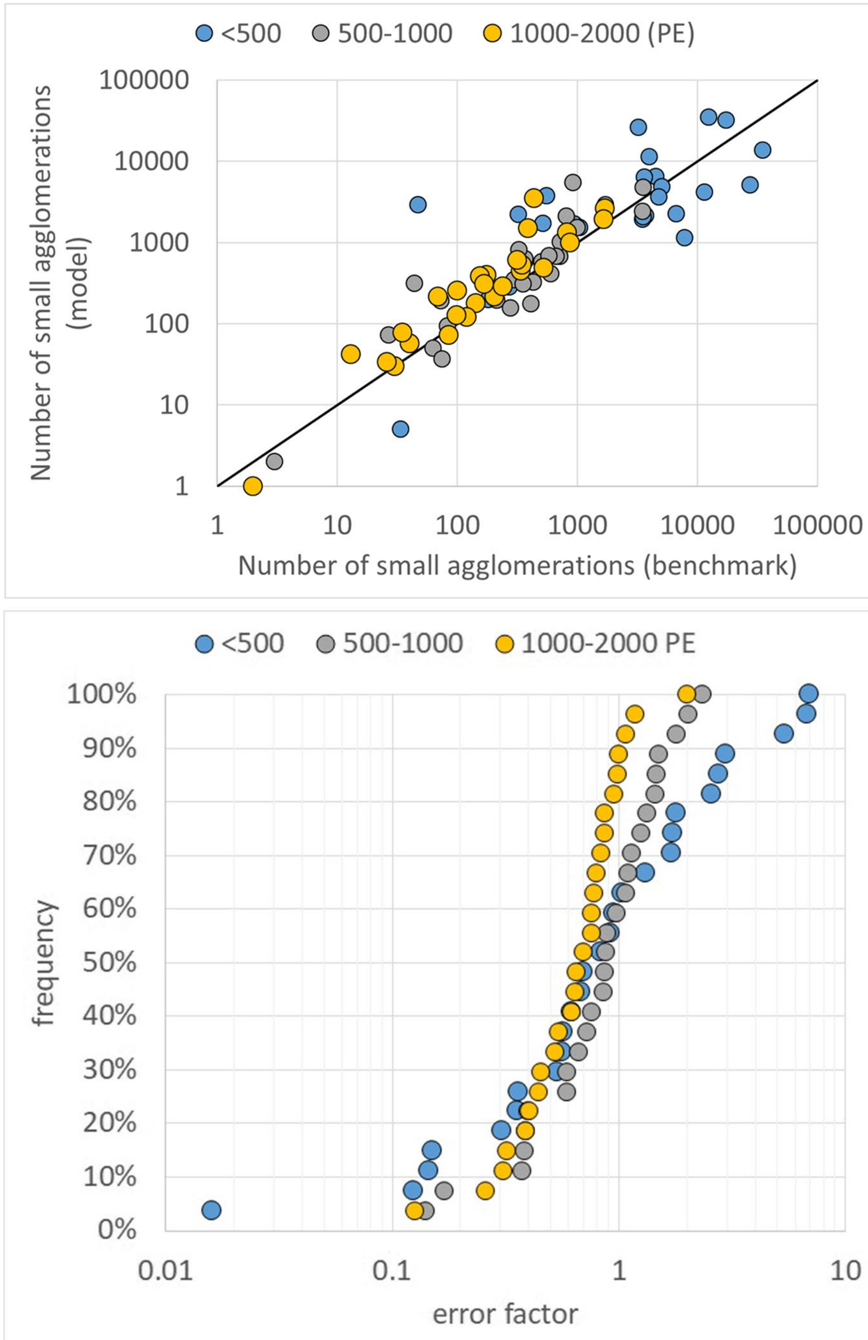

Fig. 3 Comparison of the number of agglomerations in different ranges of population equivalents (PE: less than 500, 500–1000 and 1000–2000) according to the model in this paper and an independent benchmark (estimated number of agglomerations below 2000 PE based on data provided by EU member states, Wood plc, pers. comm. 2022). Above: scatter plot (one point is one EU Member State). Below: cumulative frequency distribution of the corresponding error factors (i.e. the number of agglomerations in the benchmark divided by the number of agglomerations modelled)

above. We assume a baseline scenario where wastewater from small agglomerations undergoes treatment equivalent to a primary level. Under this scenario, we estimate small agglomerations to discharge 823,922 tonnes per year of BOD, 237,606 tonnes per year of N and 31,233 tonnes per year of P. We then consider various scenarios where we assume that small agglomerations are required to undergo secondary wastewater treatment, and possibly also tertiary (N and/or P removal) depending on their size

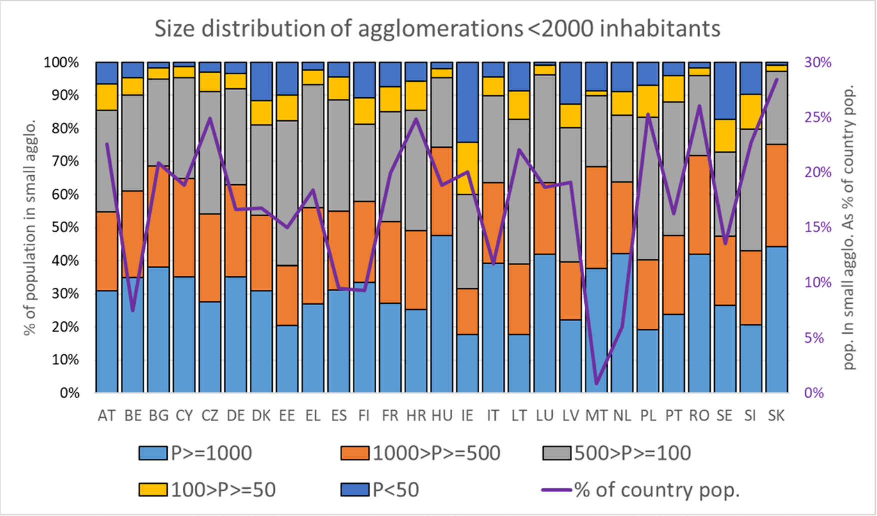

Fig. 4 Distribution of population in agglomerations *below* 2000 PE in the EU's member states

and whether they fall in areas defined as "sensitive" according to the UWWTD (Table 3). For the identification of sensitive areas, we refer to the available map discussed in Bouraoui et al. (2022), and Pistocchi et al. (submitted). In the most extreme scenarios, we assume that agglomerations above a size threshold undergo tertiary treatment for N and P in the whole territory of the EU.

With these assumptions, we calculate the discharges of BOD, N and P under each scenario, as shown in Fig. 5. Obviously, discharges decrease as we lower the size threshold above which treatment is required. BOD discharges do not change significantly after adding tertiary treatment, while the reduction of N and P is substantial. For BOD, the maximum reduction of discharges is in the order of 80% under the most extreme scenarios (requirement of a secondary treatment or higher for all agglomerations above 50 PE). Limiting the requirement of secondary treatment to agglomerations above 1000 and 500 PE would reduce baseline BOD discharges of about a quarter and a half, respectively. For N and P, Fig. 6 presents the marginally avoidable discharges attainable under increasingly demanding scenarios. Requiring secondary treatment for agglomerations above 1000 PE would

Table 3 Definition of scenarios and estimated benefit-to-cost ratio (B/C). SA = N or P removal required in sensitive areas. B/C is discussed in Section 4

Scenario	Size threshold (PE)	N or P removal	Benefit to cost ratio (B/C)
Secondary > = 1000 PE	1000	No	2.07
Secondary > = 500 PE	500	No	1.84
Secondary > = 100 PE	100	No	1.47
Secondary > = 50 PE	50	No	1.37
Secondary > = 1000 PE, tertiary in SA	1000	SA	1.29
Secondary > = 500 PE, tertiary in SA	500	SA	1.19
Secondary > = 100 PE, tertiary in SA	100	SA	1.01
Secondary > = 50 PE, tertiary in SA	50	SA	0.96
Tertiary > = 1000 PE	1000	Whole territory	1.21
Tertiary > = 500 PE	500	Whole territory	1.11
Tertiary > = 100 PE	100	Whole territory	0.96
Tertiary > = 50 PE	50	Whole territory	0.91

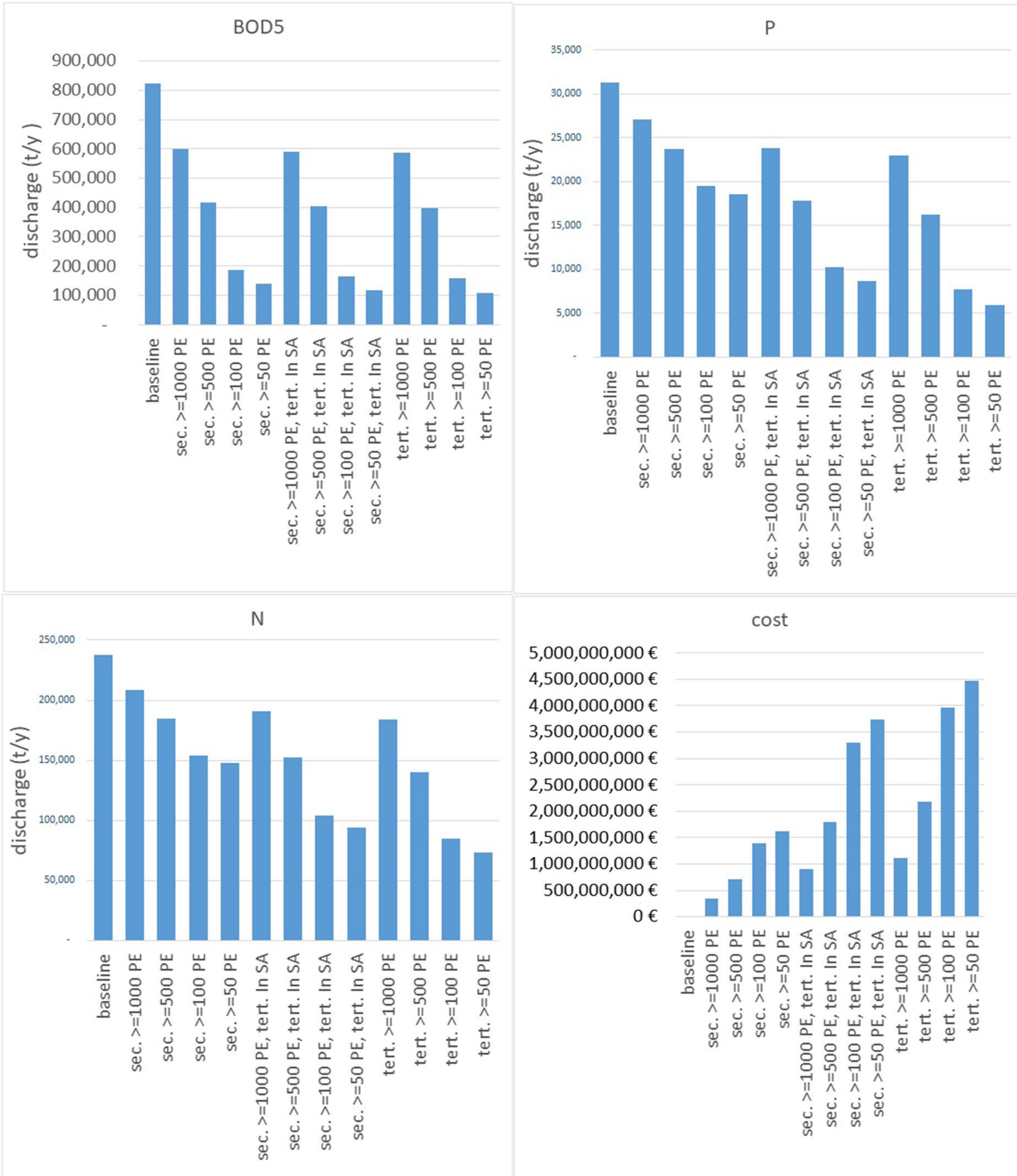

Fig. 5 Discharges of BOD, N and P under different scenarios, and the corresponding estimated costs

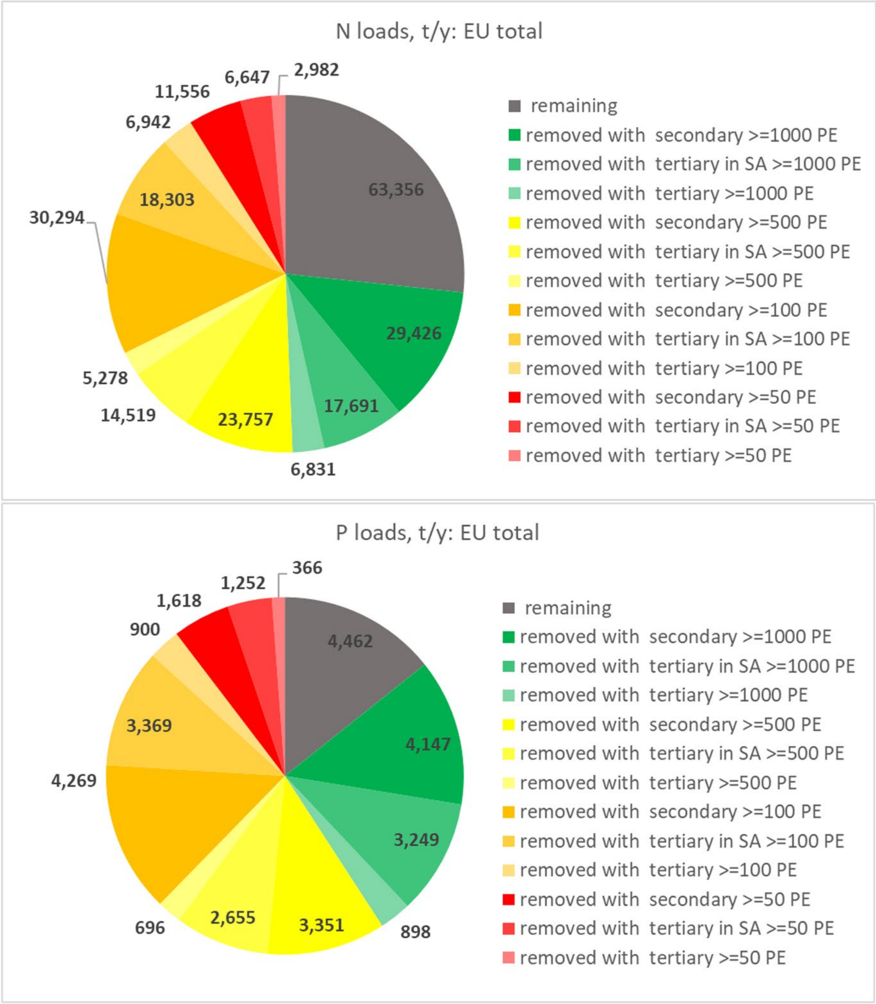

Fig. 6 Incremental reduction of discharges of total P and total N from small settlements, depending on the level of treatment enforced. The colour coding reflects increasingly demanding scenarios, and the grey sector indicates the share of baseline discharges that would remain under the most demanding scenario

reduce discharges of about 12% for N and 13% for P. Under the most extreme scenario of tertiary treatment in the whole territory for all agglomerations above 50 PE, discharges could be reduced of about 70% for N and 80% for P.

3.3 Costs and Benefits of Treatment for Smaller Agglomerations

Using the cost model described in Section 2.2, we can estimate the total costs of the various scenarios. These are presented together with the discharges of N, P and BOD in Fig. 5. While the costs increase quite mildly under scenarios entailing secondary treatment only, tertiary treatment causes a much sharper rise as shown by the graphs. We can appraise the cost-effectiveness of the various scenarios by referring to a conventional benefit-to-cost ratio (B/C) by valuing the removal of 1 kg of BOD, N and P through a shadow price of € 0.05, € 20 and € 30 respectively, in line with UNEP, 2015. A conventional benefit is calculated for each scenario by multiplying the removed quantities of BOD, N and P (Figs. 5 and 6) by the respective shadow prices. The benefit value of each scenario is then divided by the corresponding cost shown in Fig. 5. The calculated B/C for all scenarios is shown in Table 3. While B/C for a scenario of secondary treatment for all agglomerations above 1000 PE exceeds 2, this ratio decreases with more demanding scenarios and comes close to 1, or even below 1 when more advanced treatment is required for the smallest agglomerations.

4 Discussion and Conclusions

Based on the model discussed in Section 2.1, we have identified 364,650 agglomerations below 2000 PE in the EU and the corresponding population of about 75 million people. This estimate is a factor 3 higher than a previous pan-European estimate by Vigiak et al. (2020), who estimated agglomerations not covered by the UWWTD to account for about 23 million people. Our estimates are rather in line with information recently collected and extrapolated from EU member states (Wood plc, pers. comm., 2022) as shown in Fig. 3. However, data on small agglomerations are not systematically collected and reported, limiting at present the accuracy of estimations within a factor of about 2. The GIS model used for the delineation of agglomerations is likely to overestimate the number and population of small agglomerations, because it identifies as small agglomerations any cluster of population meeting the continuity and separation criteria discussed in Section 2.1. In this way, we count as small agglomerations also many small clusters at borderline distance from larger agglomerations that in reality are likely to be connected to the wastewater treatment plant of the latter. For example, data for Austria (where all agglomerations above 50 PE can be assumed to always have a WWTP) indicate a number of 1040 agglomerations between 50 and 500 PE, 135 between 500 and 1000 PE and 120 between 1000 and 2000 PE (Lenz et al., 2021), much smaller than our estimates (5111, 667 and 447 agglomerations respectively). The population equivalents of agglomerations below 2000 PE is about 728,000 PE (Lenz et al., 2021), while we estimate about 1.4 million PE. At the same time, our estimate of the number of agglomerations below 50 PE is lower than the number reported by Langergraber et al. (2018) (4675 vs 6372 agglomerations), suggesting that the model is prone also to the error of identifying as a single agglomeration what can be in reality a cluster of separate, smaller agglomerations. However, in this case, the discrepancy seems less severe.

In principle, the model could be refined through a better calibration with known and reported data on the number and served population of small agglomerations. However, it is unlikely that we can achieve better results unless we use statistics for the majority of countries. At the same time, if these were effectively available, the model itself would not be needed anyway.

In spite of these limitations, the model provides a means to estimate the consistency of small agglomerations in the EU in the absence of reported data. Based on our calculations, we can conclude that small agglomerations in the EU generate significant loads of BOD and nutrients, and may represent important pressures on water bodies.

The scope of the UWWTD covers agglomerations above 2000 PE, requiring secondary treatment unless they discharge in sensitive areas, in which case nutrient removal is also required. While various member states have already regulated smaller agglomerations, in many cases, quality standards correspond to primary treatment, if not even untreated wastewater. In principle, by enforcing an appropriate level of treatment, we could reduce pollution loads from small agglomerations to a significant extent.

However, in order to decide on appropriate treatment standards, we need to compare the costs and effectiveness of investments in treatment of small agglomerations with those in the control of other sources of pollution. The benefit-to-cost ratio (B/C) of various scenarios of treatment for small agglomerations suffers from the variability of the value of BOD removal and, most importantly, N and P removal depending on the conditions of the receiving water body. Moreover, the costs of wastewater treatment for small agglomerations depend on the initial conditions of the infrastructure and local factors such as the availability of space and the possibility to automate the management of processes. Still, the simple conventional calculation presented above provides, as a first approximation, some support for decisions in this matter. Particularly, the B/C shows that investments in secondary and even more in tertiary treatment for small agglomerations may not be the most cost-effective options to improve water quality. Nevertheless, requiring secondary treatment for agglomerations above 1000 PE shows a B/C around 2; hence, lowering the current threshold of the UWWTD above which secondary treatment is required, from 2000 to 1000 PE, may be justified by a reasonable safety margin.

An alternative to tertiary treatment for small agglomerations, yielding comparable results in terms of avoiding N and P discharges to the receiving water bodies, could be a treatment configuration enabling the reuse for water and nutrients for agricultural fertilisation via irrigation ("fertigation"). This could help avoid use of mineral fertilisers to some extent, while reducing the investment and operation costs of plants. At the same time, fertigation requires an appropriate control of pathogens and micropollutants released with wastewater, which may prove expensive and undermine the feasibility of this solution.

In addition to the costs of action and the benefits from BOD, N and P removal, in principle, it is relevant to consider the potential change of greenhouse gas (GHG) emissions under the various scenarios. To this end, a GHG emission assessment was performed including different small plant typologies (see Annex 6.). The evidence available (Fig. 11) does not suggest a clear advantage of one treatment system over the other for small agglomerations: while replacing septic tanks and similar primary systems with secondary systems significantly contributes to reducing direct emissions of methane, increased direct emissions of nitrous oxide may offset the improvement. Moreover, more advanced treatments may entail larger energy consumption, which usually corresponds to higher GHG emissions. However, we can observe that some TW typologies combined with a primary settler and sludge stabilisation in a reed bed offer lower emission factors among the considered options. Thus, hybrid multistage systems need to be applied to improve N removal, triggering higher methane emissions. The off-site treatment and disposal of sludge may add to the GHG emissions and must be properly considered. Last but not least, the infrastructure for the collection of sewage entails use of concrete and other resources and is associated to relatively high GHG emissions (Morera et al., 2016). Changes in GHG emissions must be evaluated on a case-by-case basis in order to account for the specific factors that may make a solution preferable over the others. Furthermore, it must be stressed that GHG emissions are affected by a high uncertainty related to the estimation of N_2O and CH_4 generation in passive systems such as treatment wetlands. There is also evidence that poorly managed or overloaded systems can lead to much higher CH_4 emissions than assumed in theoretical calculations. All things considered, more stringent treatment of smaller agglomerations is not likely to change the GHG balance significantly.

Measures for smaller agglomerations, and generally a requirement of more stringent treatment for agglomerations below 2000 PE, may not be justified in all cases, although usually our calculated B/C remains above 1. In these cases, it may be most appropriate to make a decision case by case also depending on the conditions of the receiving water bodies, as currently required by the Water Framework Directive 2000/60/EC.

Data Availability A GIS layer of agglomerations delineated with the calibrated model of Section 2.1 is made available as Supplementary material to this article.

Declarations

Conflict of Interest The authors declare no competing interests.

Annex 1. Model calibration

For different combinations of $\boldsymbol{n}$ and ρ_0, we check how the corresponding simulation of agglomerations compares with the reported numbers and populations in agglomerations above 2000 PE (Table 4), assumed to be the reference.

If $\mathbf{x}$ is the vector of reported PEs (first row in Table 4), $\mathbf{y}$ is the vector of reported number of agglomerations (second row in Table 4) and $(\mathbf{x_k}, \mathbf{y_k})$ is the tensor of the corresponding data coming from statistics of the kth simulation of agglomerations, we compute distance metrics as:

Table 4 Source: 8th UWWTD Implementation Report. Only for Poland and Italy, and due to their unavailability, the information is based on data gathered from the 7th Implementation Report (https://ec.europa.eu/environment/water/water-urbanwaste/implementation/implementationreports_en.htm)

	All	2000–10,000 PE	> 10,000 PE	> 150,000 PE
Total load discharge (million PE)	588	68	519	277
Total number of agglomerations	23,569	15,011	8558	662

$$D_x = \sqrt{\sum_{i=1}^{4}\left[\frac{x_i - x_{k,i}}{x_i}\right]^2}$$

$$D_y = \sqrt{\sum_{i=1}^{4}\left[\frac{y_i - y_{k,i}}{y_i}\right]^2}$$

Ideally, a simulation should have $D_x = D_y = 0$. We choose the simulation with minimum distance to the ideal point $D = \sqrt{D_x^{\ 2} + D_y^{\ 2}}$. We consider simulations with $\boldsymbol{n}=1$ and $\boldsymbol{\rho}_0 = 1$, 10, 20, 30, 40 and 50, in addition to simulations with $\boldsymbol{\rho}_0 = 10$ and $\boldsymbol{n}=0$, 2 and 5. Their performance in the (D_x, D_y) space is shown in Fig. 7.

While the simulations are reasonably realistic, they obviously present discrepancies compared to the reported data (Fig. 8).

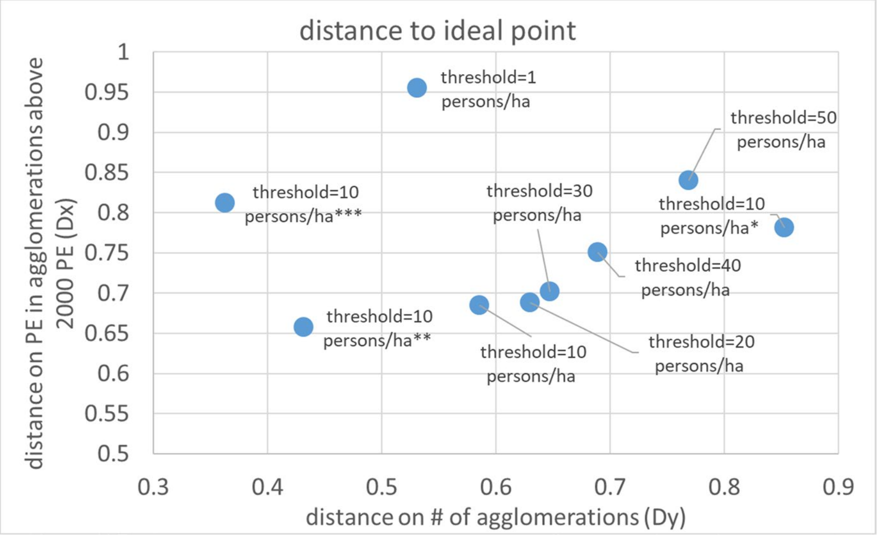

Fig. 7 Performance of different combinations of parameters $\boldsymbol{n}$ and $\boldsymbol{\rho}_0$. The latter is the threshold density. Parameter $\boldsymbol{n}$ is always 1 except (*) $\boldsymbol{n}=0$, (**) $\boldsymbol{n}=2$, (***) $\boldsymbol{n}=5$

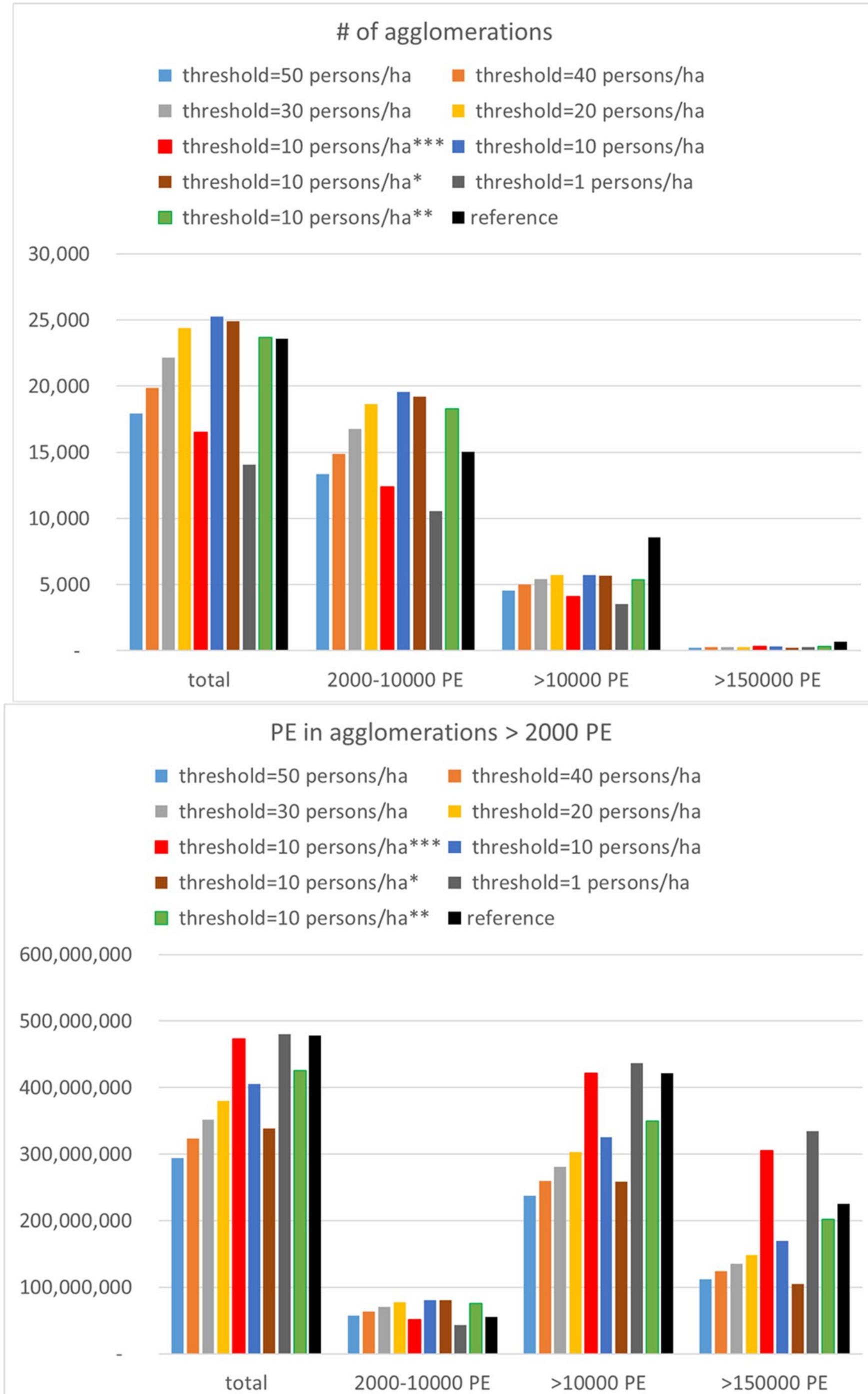

Fig. 8 Comparison of simulated and reported (reference) served PEs and number of agglomerations, for different combinations of $\boldsymbol{n}$ and ρ_0. Parameter n is always 1 except (*) $n=0$, (**) $n=2$, (***) $n=5$

Annex 2. Details of the cost model

We estimate the cost entailed by providing a collection system and a wastewater treatment system for all small agglomerations below 2000 PE under the following assumptions.

1. The cost of a collection system (Euro/PE) depends on the population in the agglomeration, *P*, according to the FEASIBLE model (COWI, 2010; OECD, 2004) expenditure function:

$$C_{Network} = 2828.8 - 190.3\ \mathrm{In}(P)$$

We assume an operation and maintenance (O&M) cost of 1% of the investment per year. The lifetime of the network is supposed to be 100 years, and we use a discount rate of 2.5% for the investment.

2. The cost of a septic tank, according to OECD, 2004 and update in COWI, 2010, is:

$$C_{Septic} = 1.7\ (835 - 98\ \log\ (P))$$

We assume an O&M cost of 3%, a lifetime of 30 years and a discount rate of 2.5%.

3. The cost of an improved wastewater treatment system is given by the average of costs of various typologies of treatment wetlands according to Pistocchi et al. (2020). The cost of investment (Euro/PE) and operation (Euro/PE/year) is given in the form cost = $a\ P^b$, with a (coefficient) and b (exponent) given in Table 5. For the plants, we also assume a lifetime of 30 years and a discount rate of 2.5%. With these parameters, we compute an annualised cost function for each typology of treatment wetland (see Fig. 9), and we consider the average of the ensemble.

Table 5 Cost parameters for the typologies of treatment wetlands outlined in Pistocchi et al. (2020) (FBA, forced bed aeration; FRB, French reed beds; HF, horizontal flow; VF, vertical flow; Hybrid denotes a combination of VF and HF; "reuse" or "no reuse" denote typologies where effluents are/are not required to meet standards for water reuse)

Type of treatment wetland	Capex		Opex	
	Coefficient	Exponent	Coefficient	Exponent
FBA	629.57	0.144	74.634	0.229
Hybrid	784.26	0.147	44.394	0.164
FRB	903.79	0.19	99.155	0.375
HF (reuse)	4226.3	0.471	86.041	0.198
HF (no reuse)	2383.8	0.41	61.471	0.122
VF	3746.4	0.391	60.623	0.101

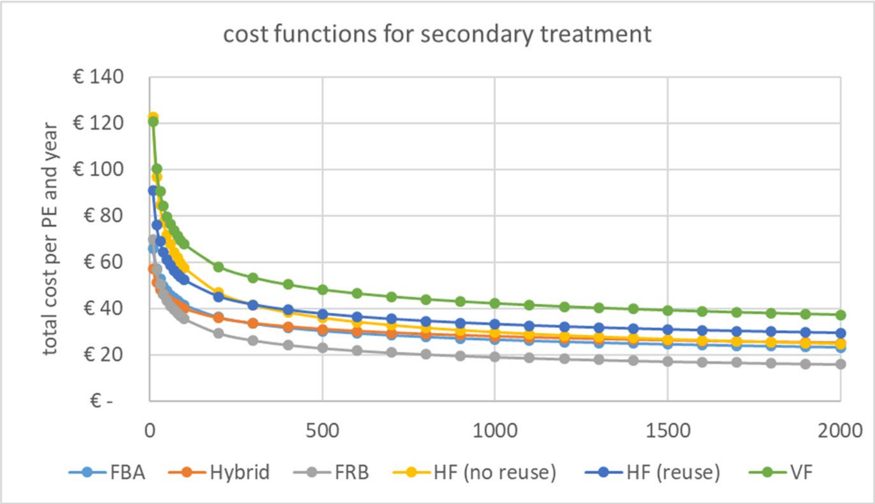

Fig. 9 Annualised costs for the typologies of treatment wetlands in Table 5 (FBA, forced bed aeration; FRB, French reed beds; HF, horizontal flow; VF, vertical flow; Hybrid denotes a combination of VF and HF; "reuse" or "no reuse" denote typologies where effluents are/are not required to meet standards for water reuse)

4. The additional cost of collection and treatment, compared to the cost of septic tanks without collection, is given by the sum of collection network costs and the above average, minus the cost of a septic tank system. This can be approximated by the function:

$$\text{Annualised cost} = 218.36 * P^{-0.37}(\text{Euro/PE/year})$$

This is equal to an estimated cost of a treatment wetland system, plus the estimated cost of a small collection network servicing the agglomeration, minus the cost of a septic tank sized for the whole agglomeration.

The cost of N removal in excess of simple carbon removal (secondary treatment) is computed as the cost of a vertical flow treatment wetland as per Table 5, minus the average of costs assumed for secondary treatment. This is well approximated by the function:

$$\text{Annualised additiona lcost of N removal} = 61.44 * P^{-0.14}(\text{Euro/PE/year})$$

For P removal, we assume the difference of costs between secondary plants and tertiary plants with P removal only according to OECD, 2004, and COWI, 2010. This is well approximated by the function:

$$\text{Annualised additional cost of P removal} = 102.1 * P^{-0.315}(\text{Euro/PE/year})$$

The cost functions of the above additional costs are plotted in Fig. 10.

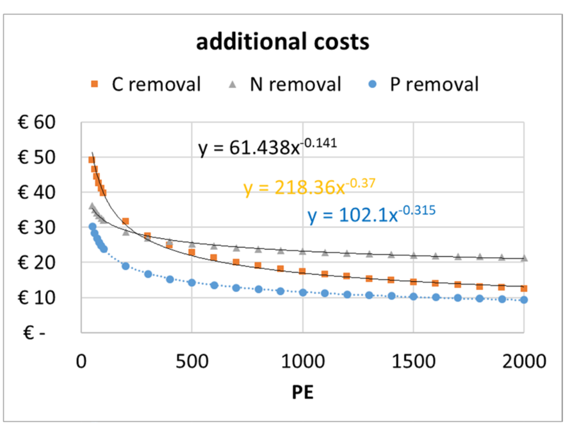

Fig. 10 Cost functions used in the assessment

Annex 3. Greenhouse gas emissions

We conducted an analysis of greenhouse gas emissions (GHG) using a life cycle assessment approach in line with the methods outlined in Parravicini et al. (2022). This allowed defining emission factors for various typologies of wastewater treatment plants for small agglomerations. For technical plants down to a capacity of 500 PE, we refer to typologies discussed in Parravicini et al. (2022) (typologies 4.*x* in Fig. 1 therein). For plants below 500 PE, we consider the additional typologies listed in Table 6.

For each typology, we compute a PE-specific GHG emission load in kg CO_2e per PE per year, including direct and indirect GHG emission sources. The balance boundaries of the assessment comprise the building and the operation phases of the wastewater treatment system (assumed life time of 30 years). Nitrous oxide and methane emitted in the water bodies receiving the treated effluents were also included. Sludge stabilisation options at the technical plants were aerobic simultaneous sludge stabilisation (SASS), post-aerobic sludge stabilisation (ASS) and reed beds (RB). Sludge transport and further treatment and/or disposal were out of the assessment boundaries. In the option with primary sludge being transferred to a WWTP for stabilisation, CO_2e emitted by transportation and treatment in the WWTP were accounted for. For transportation, we applied an emission factor of 0.134 kgCO_2e per ton per kilometre (Ecoinvent 3.7, market group for transport, freight, lorry, unspecified, Global, 2020). For the sludge stabilisation in the WWTP, we apply the average total CO_2e emission load estimated for WWTP typologies 5000–30,000 PE with SASS or ASS or RB in Parravicini et al. (2022).

GHG emissions from the construction phase of sewer network down to 50 PE were also included in the assessment and estimated to about 38 g CO_2e/PE/year, as described in Parravicini et al. (2022). The estimation of methane emissions from sewer networks is challenging due to their geographical extension and variability of conditions (Ye et al., 2022). Of the methane produced and emitted in the sewer network, only the load remaining dissolved in the sewage and then stripped in aerated treatment systems was considered, applying the approach described in IPCC, 2019. The emitted methane load was estimated at 4 to 6 g CO_2e/PE/year.

Tailored COD, N and P mass-balances for each plant typology, capacity and treatment performance provided the inventory data for the GHG assessment,

Table 6 Treatment typologies considered to treat the wastewater from small agglomerations. *SASS*, simultaneous sludge stabilisation; *post-ASS*, aerobic sludge post-stabilisation; *RB*, reed beds

Code	Description	Pollutants addressed	Treatment options
5.a.3	Small technical aerobic systems	COD/NH_4^+	Sludge stabilisation: SASS
5.a.4	Small technical aerobic systems	$COD/NH_4^+/P$	Sludge stabilisation: SASS
5.a.5	Small technical aerobic systems	COD/N	Sludge stabilisation: SASS
5.a.6	Small technical aerobic systems	COD/N/P	Sludge stabilisation: SASS
5.b.1	Small technical aerobic systems	COD	Sludge stabilisation: post-ASS
5.b.2	Small technical aerobic systems	COD/P	Sludge stabilisation: post-ASS
5.b.3	Small technical aerobic systems	COD/NH_4^+	Sludge stabilisation: post-ASS
5.b.4	Small technical aerobic systems	$COD/NH_4^+/P$	Sludge stabilisation: post-ASS
5.b.5	Small technical aerobic systems	COD/N	Sludge stabilisation: post-ASS
5.b.6	Small technical aerobic systems	COD/N/P	Sludge stabilisation: post-ASS
5.b.1r	Small technical aerobic systems	COD	Sludge stabilisation: RB
5.b.2r	Small technical aerobic systems	COD/P	Sludge stabilisation: RB
5.b.3r	Small technical aerobic systems	COD/NH_4^+	Sludge stabilisation: RB
5.b.4r	Small technical aerobic systems	$COD/NH_4^+/P$	Sludge stabilisation: RB
5.b.5r	Small technical aerobic systems	COD/N	Sludge stabilisation: RB
5.b.6r	Small technical aerobic systems	COD/N/P	Sludge stabilisation: RB
6.a.1	Treatment wetlands, HF	COD	Sludge stabilisation: RB, combined with primary settler
6.a.2	Treatment wetlands, HF	COD/P	Sludge stabilisation: RB, combined with primary settler
6.a.1 s	Treatment wetlands, HF	COD	Combined with septic tank, sludge transferred and treated at a WWTP
6.b.1	Treatment wetlands, VF	COD/NH_4^+	Sludge stabilisation: RB, combined with primary settler
6.b.2	Treatment wetlands, VF	$COD/NH_4^+/P$	Sludge stabilisation: RB, combined with primary settler
6.b.1 s	Treatment wetlands, VF	COD/NH_4^+	Combined with septic tank, sludge transferred and treated at a WWTP
7.a.1	Septic tank with overflow and ground/filter infiltration	COD	Sludge stabilisation: RB
7.a.2	Septic tank with overflow and ground/filter infiltration	COD/P	Sludge stabilisation: RB
7.b.1	Septic tank with overflow and ground/filter infiltration	COD	Sludge stabilisation: transported to WWTP
7.b.2	Septic tank with overflow and ground/filter infiltration	COD/P	Sludge stabilisation: transported to WWTP

based on sound literature values and expert knowledge. Electricity demand for the operation of the technical aerobic systems was derived from Ganora et al. (2019) (see also Parravicini et al., 2022). Electricity demand for non-aerated TW (e.g. for wastewater pumping) was also estimated, although being comparatively small. In all options featuring P removal, iron chloride was used per default as flocculant/precipitation agent.

Emission factors for the impact assessment were mainly derived from the IPCC Guideline Refinement 2019 (IPCC, 2019) or other literature sources presented in details in Parravicini et al. (2022).

In addition to the methodological approach described in in Parravicini et al. (2022), we made the following additional assumptions:

- *Small technical aerobic systems (50–500 PE)*

For typology 5.*x*.*y* (*x*=*a*,*b*; *y*=1 to 6), we set the same assumptions as for the larger wastewater treatment plants 500–5000 PE in Parravicini et al. (2022). An exception is the electricity demand, which was assumed higher for the smaller plants. We assumed, based on literature values, that plants striving N removal exhibit lower direct N_2O emissions than plants without targeted N removal (detailed information in Parravicini et al., 2022).

- *Treatment wetlands (<2.000 PE)*

For typologies 6.*x*.*y* (*x*=*a*,*b*; *y*=1,2) were modelled as treatment wetlands, in the options vertical flow (VF) and horizontal flow (HF) wetlands. The removal performance of the two systems was derived from the IWA books on treatment wetlands Dotro et al. (2017) and Langergraber et al. (2019): The removal of COD, NH4+, N and P for VF wetlands are 90%, 90%, 20% and 15% respectively; for HF wetland they are 80%, 25%, 40%, 15% respectively. Applying hybrid multistage systems combining different wetland typologies would allow for higher treatment performance, especially in term of N removal. This option was not evaluated separately, and the results can be extrapolated from the ones for VF and HF wetlands. Enhanced P removal was achieved through flocculation/precipitation with iron chloride. We assumed that the wastewater is treated first mechanically in a primary settler (50% COD removal) and the primary sludge is stabilised and dewatered in a RB. In an additional option, a septic tank was used for primary treatment. In this case, sludge is transported to a WWTP in order to undergo stabilization. We assume a transport distance of 40 km both ways. The additional CO_2e emissions released at the WWTP were added to the CO_2e emissions directly related to the TW. Electricity demand for wastewater pumping (calculated assuming a flow of 150 L/PE/day, and a pumping head of 2 metres) is small compared to the consumption for aeration in technical aerobic systems. Emission embedded in the infrastructure were estimated considering resource use for excavation works (4–5 m^2/PE, 1–1.5-m deep according to Mander et al., 2014 and Dotro et al., 2017).

The emission factor for direct N_2O and CH_4 emissions were derived from the 2013 Supplement on Wetlands to the 2006 IPCC Guidelines for

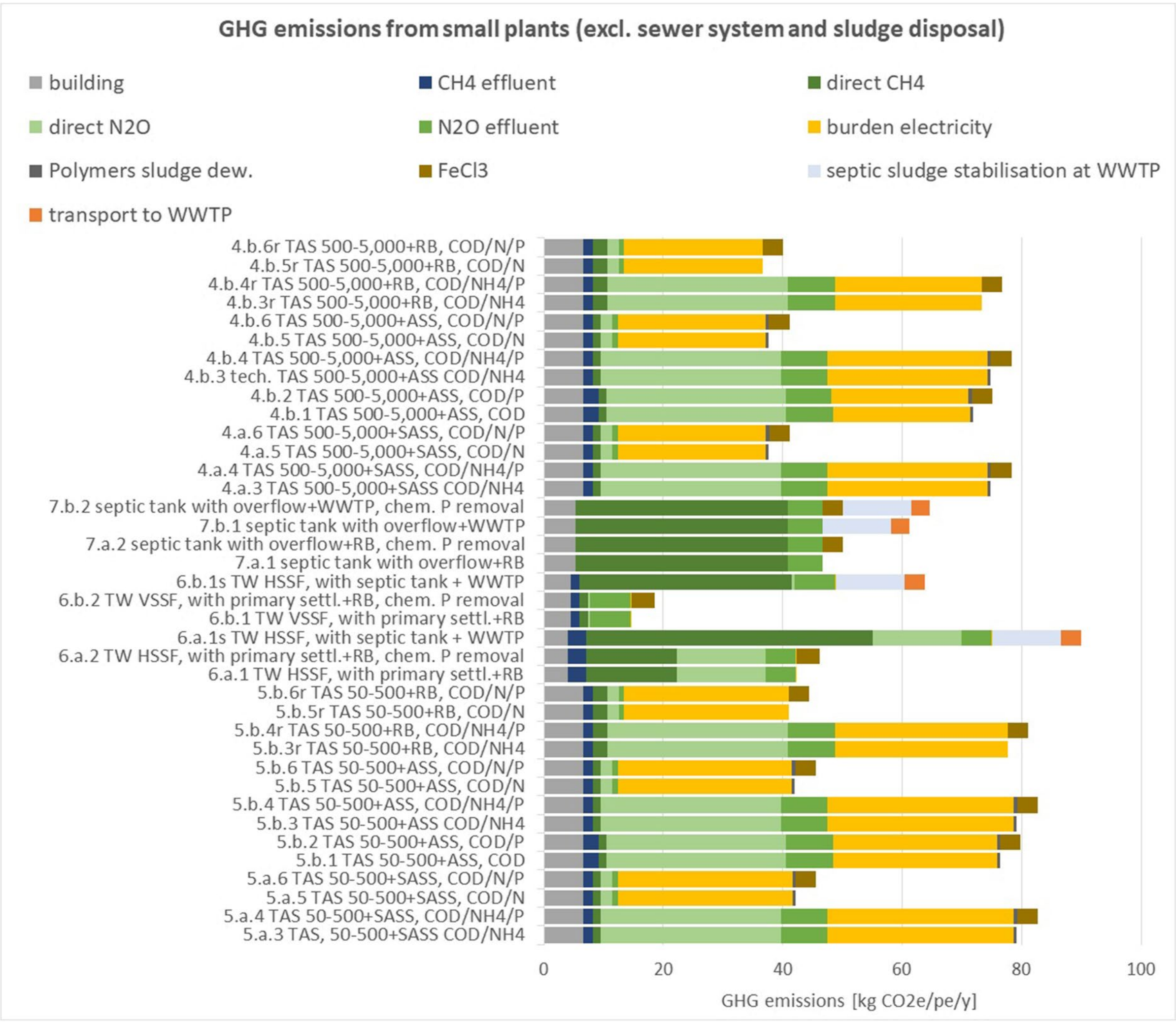

Fig. 11 Population-specific CO_2e emission loads of the wastewater treatment typologies considered

National Greenhouse Gas Inventories (IPCC, 2014, Chapter 6, table 6.4/equation6.2 and table 6.7): for VSSF 0.0025 kg CH_4/kg $COD_{influent}$ and 0.00023 kg N_2O-N/kg $N_{influent}$ and for HSSF 0.025 kg CH_4/kg $COD_{influent}$ and 0.0079 kg N_2O-N/kg $N_{influent}$.

Interestingly, rather aerobic systems as VSSF, providing ammonium oxidation but lower N removal, exhibit so far lower N_2O emission than systems with pronounced anoxic/anaerobic milieu (HSSF). This is not in line with experiences at centralised WWTP, where denitrification can act as a sink for the generated N_2O (Valkova et al., 2021).

To predict N_2O and CH_4 emissions in passive systems as treatment wetlands is even more challenging than in centralised WWTPs, where operating conditions can be described and controlled. The IPCC (2014) gives, e.g. for VSSF an uncertainty range of ± 56% for CH_4 and of ±70% for N_2O. It is pointed out that emission factors measured so far in TW are lower than for centralised WWTPs, so far the plants are properly designed and managed. This can be explained perhaps by the low loading conditions typically applied in these systems.

- *Septic tanks with overflow and ground/filter infiltration*

For typologies 7.*x*.*y* (*x*=*a*,*b*; *y*=1,2), removal performance was set to 50%, 33% and 33% for COD and N and P removal respectively. The sludge production was estimated to 60 g COD/PE.d, 42 g VS/PE.d (COD/VS = 1.42, DWA, 2016), 48 g TS/PE.d (assuming organic content being ~90% of TS), and 1600 g sludge/PE.d (3%TS). Two options with sludge stabilisation in RB or transport and treatment at a WWTP (assumption: 40 km both ways) were computed. The emission factor for direct methane emissions from septic tank was derived from IPCC, 2019 (table 6.3). GHG emissions embedded in the infrastructure were evaluated on the basis of concrete and stainless steel demand estimated for fictive septic tank (0.25 m^3/PE, Dotro et al., 2017).

Based on the above assumptions, we obtain the population-specific CO_2e emission loads shown in Fig. 11.

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