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A holistic approach for performance evaluation of wastewater treatment plants: integrating grey water footprint and life cycle impact assessment

Shervin Jamshidi *, Mohammad Farsimadan and Hanieh Mohammadi
Department of Civil Engineering, University of Isfahan, HezarJerib Blvd, Isfahan, Iran
*Corresponding author. E-mail: sh.jamshidi@eng.ui.ac.ir

SJ, 0000-0003-3067-1529

ABSTRACT

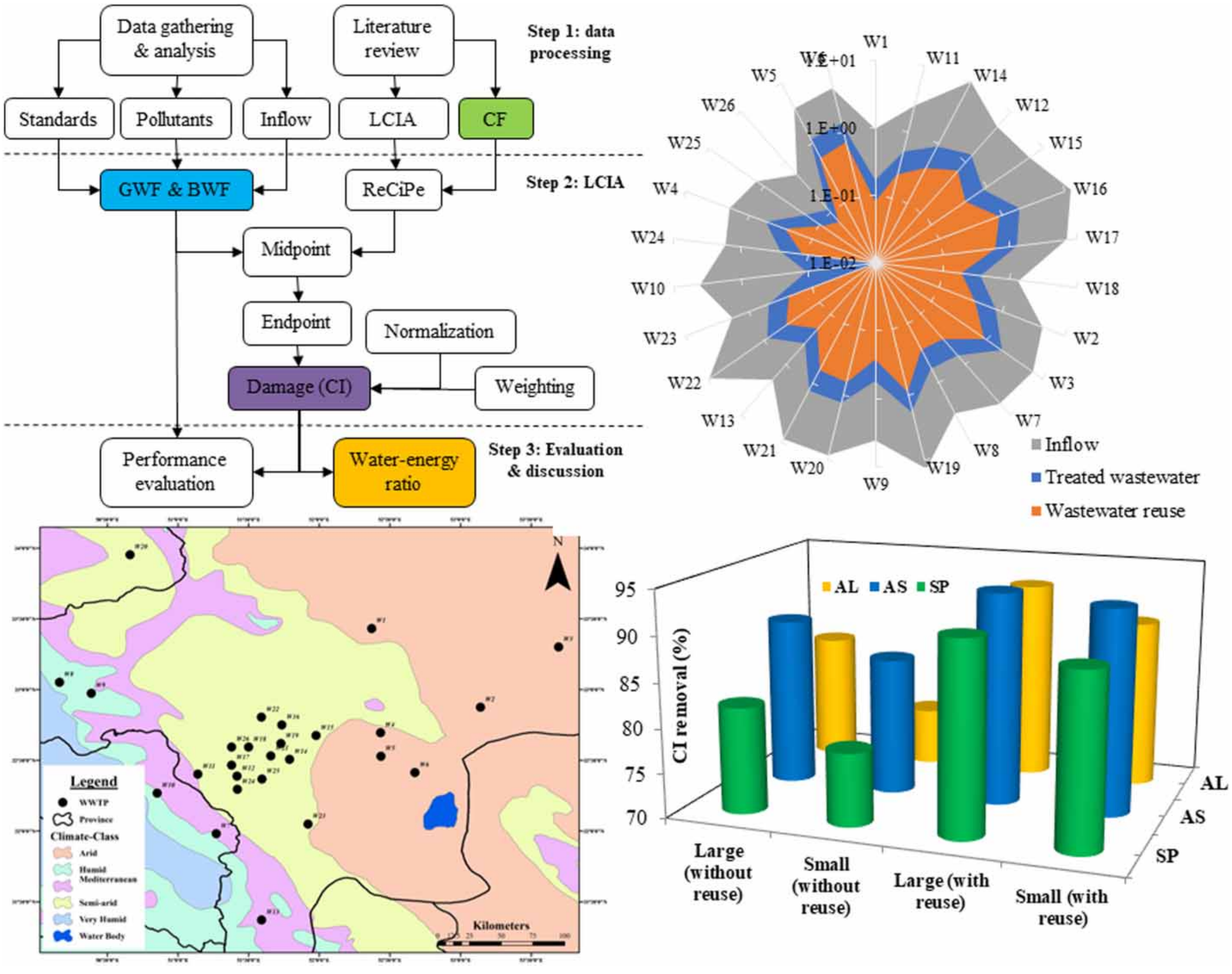
Wastewater treatment plants (WWTPs) have positive and negative impacts on the environment. Therefore, life cycle impact assessment (LCIA) can provide a more holistic framework for performance evaluation than the conventional approach. This study added water footprint (WF) to LCIA and defined ϕ index for accounting for the damage ratio of carbon footprint (CF) to WF. The application of these innovations was verified by comparing the performance of 26 WWTPs. These facilities are located in four different climates in Iran, serve between 1,900 and 980,000 people, and have treatment units like activated sludge, aerated lagoon, and stabilization pond. Here, grey water footprint (GWF) calculated the ecological impacts through typical pollutants. Blue water footprint (BWF) included the productive impacts of wastewater reuse, and CF estimated CO_2 emissions from WWTPs. Results showed that GWF was the leading factor. ϕ was 4–7.5% and the average WF of WWTPs was 0.6 m^3/ca, which reduced 84%, to 0.1 m^3/ca, through wastewater reuse. Here, wastewater treatment and reuse in larger WWTPs, particularly with activated sludge had lower cumulative impacts. Since this method takes more items than the conventional approach, it is recommended for integrated evaluation of WWTPs, mainly in areas where the water–energy nexus is a paradigm for sustainable development.

Key words: activated sludge, greenhouse gas, wastewater reuse, wastewater treatment, water–energy nexus, water footprint

HIGHLIGHTS

- An integrated method was developed for comparing WWTPs' performance.
- Grey and blue water footprints were added to LCIA as environmental indices.
- Method applicability was verified by comparing 26 WWTPs.
- Larger WWTPs with activated sludge comparatively had less environmental impacts.
- Wastewater treatment and reuse reduced 92% of environmental damage.
- A new index for water–energy nexus in WWTPs was introduced.

GRAPHICAL ABSTRACT



INTRODUCTION

Water–energy nexus (WEN) is one perspective under sustainable development that highlights the interrelations between water and energy production in policies or systems for their secure application (Wilson *et al.* 2021). Mini hydropower plants (Comino *et al.* 2020) and growing energy crops (Pacetti *et al.* 2015) are two examples that need water to produce clean energy and limiting greenhouse gas (GHG) emissions. On the opposite, desalination and wastewater treatment plants (WWTPs) require energy and emit GHGs for clean water supply (Chen *et al.* 2020). Accordingly, WWTP is mainly classified as energy consuming unit for water production (Kurian *et al.* 2019; Haitsma Mulier *et al.* 2022), in which its reclaimed water can ultimately reduce water-energy consumption. For example, a survey in the Sahara showed that wastewater reuse for irrigation could decrease 49% groundwater abstraction and 15% energy required for food production by reducing groundwater pumping demands (Ramirez *et al.* 2021). Based on the WEN perspective, in a semi-arid area like Iran, where energy sources are abundant but 80% of renewable water resources have been withdrawn (Madani *et al.* 2016), WWTPs are influential infrastructures with possible added values on a local scale (Jamshidi 2019).

WWTPs are relatively similar to factories that produce water and other products, like fertilizers and recovered phosphorous (Mo & Zhang 2013; Mulchandani & Westerhoff 2016). These facilities use energy for operating electro-mechanical tools, e.g. pumps, aerators, and mixers (Gu *et al.* 2017). Energy consumption in WWTPs indirectly emits GHG, while endogenous decay of carbonaceous and nitrogenous compounds directly releases carbon dioxide (CO_2) and nitrous oxide (N_2O) into the atmosphere (Kampschreur *et al.* 2009; Daelman *et al.* 2015; Huang *et al.* 2020). Karnaningroem & Anggraeni (2021) have argued that chemical residues and energy consumption are important factors in treatment plants. Thus, water quality should not be a single objective for their performance evaluation (Karnaningroem & Anggraeni 2021). The long-term impacts of (1) GHG emissions to the atmosphere, and (2) remaining pollutants in the effluent on the land ecosystem and health can be environmentally critical, particularly when they are dependent on WWTPs' operation (Sabeen *et al.* 2018; Yoshida *et al.* 2018). In other words, sustainable development goals (SDGs) including good health (SDG 3), clean water and sanitation (SDG 6), climate action (SDG 13), and life on land (SDG 15) are equally essential and they should be addressed simultaneously for WWTPs (Delanka-Pedige *et al.* 2021; Obaideen *et al.* 2022). From this point of view, operating WWTPs has both positive and negative environmental impacts (Gómez-Llanos *et al.* 2020). Nevertheless, there is a lack of integrated methods for evaluating the overall performance of WWTPs regarding their total impacts on the environment. WWTPs are currently reviewed for their water pollution removal or meeting regional water quality standards. That is why a new holistic approach is required for their performance evaluation considering possible environmental impacts instead of conventional pollution removal assessments. In the last decade, life cycle assessment (LCA) has been recommended to consider a broader range of impacts and compare the performance of WWTPs (Corominas *et al.* 2020).

LCA is a four step analytical tool that aggregates the estimated direct and indirect environmental impacts of activities or productions from their 'cradle to grave' (Hauschild *et al.* 2018). In its third step, life cycle impact assessment (LCIA), quantitative indices convert pollution loads or resource depletion into equivalent damages under different categories (Rosenbaum *et al.* 2018). For example, in ReCiPe 2016, a developed LCIA database, some affected midpoint categories for WWTPs are eutrophication, water depletion, and global warming (Gallego-Schmid & Tarpani 2019), while endpoints are the ecosystem, human health, and resources (Huijbregts *et al.* 2017). The indicator-based LCIA has some privileges. It firstly reduces the dependency of environmental impact assessments on expert opinions and puts one step toward an unbiased standard framework. Second, it unifies different environmental categories and aggregates their impacts that basically cannot be accumulated due to their different units, effective periods, and receiving bodies (e.g. air, marine, freshwater, agricultural or industrial soil) (Bulle *et al.* 2019). Finally, it has the flexibility to include more indices, e.g. water footprint (WF), in its calculations (Jamshidi & Naderi 2023a). On the contrary, a drawback of LCIA for evaluating WWTPs is that this method mainly quantifies the impacts of hazardous materials (e.g. toxins, heavy metals, and pharmaceuticals) in association with pollutants with clear ecological consequences (e.g. nitrogen and phosphorous in eutrophication). Regular water quality parameters in wastewater treatment like biochemical oxidation demand (BOD), total suspended solids (TSS), and chemical oxidation demand (COD) are not included in LCIA databases (Bai *et al.* 2017). Suryawan *et al.* (2021) recently considered the potential impacts of COD and BOD under eutrophication for the LCA of WWTPs (Suryawan *et al.* 2021). However, it is obvious that these parameters do not lead to eutrophication, as nitrogen and phosphorous are their main cause (Schindler *et al.* 2016; Jamshidi & Naderi 2023b). Another, Malik *et al.* (2015) proposed an environmental performance index (EPI) for comprehensive performance analysis of wastewater treatment (Malik *et al.* 2015). EPI can consider average treatment level, volume, connections etc. on national or regional scale but it is not applicable for specific WWTP evaluation. Therefore, we propose an integrated method to introduce WF into LCIA to fill this gap and enable WWTP evaluation via LCIA based on typical wastewater quality parameters.

WF is the water embedded in a production or service. It consists of blue, green, and grey elements (Hoekstra *et al.* 2011). Grey water footprint (GWF) is the key WF component for the performance evaluation of operating WWTP (Gómez-Llanos *et al.* 2020) because it represents the equivalent volume of freshwater required to assimilate pollutants to standard water quality levels (Franke *et al.* 2013). Moreover, WWTPs can recycle water for reuse. Based on this perspective, WWTPs are a resource for blue water footprint (BWF). These two factors can be adopted as consumed water in the LCIA of WWTPs (Morera *et al.* 2016). Carbon footprint (CF) has also the potential to join GWF and BWF for the evaluation of WWTPs. CF summarizes the total equivalent GHG emitted from a production or service like wastewater treatment. According to 225 WWTPs in China, it has been estimated that the direct and indirect carbon emissions from WWTPs constitute 64 and 36% of total CF, respectively (Chen *et al.* 2023). Here, the type of treatment unit, its design and operation, sludge management, the performance and efficiency of mechanical equipment, technologies used in WWTPs, the type of consumed fuel for energy or equipment production in their value chain, nutrient removal, and wastewater reuse are effective factors on CF (Parravicini *et al.* 2016). For the latter, researchers realized that wastewater treatment for irrigation can save 7% of energy and reduce 3% of the CF of WWTPs (Marinelli *et al.* 2021).

This study develops a joint method that adopts GWF in the LCIA to primarily allow this method to take typical wastewater treatment pollutants (e.g. COD, TSS) in calculations. Second, BWF and CF are added to include the impacts of wastewater reuse and GHG emissions in WWTPs evaluation. Finally, for verification, the developed method quantitatively evaluates and compares the overall performances of 26 WWTPs in Iran during operation. The research scope is confined to biological wastewater treatment and excludes the possible impacts of construction, sludge management, chemical additives, and wastewater collection. Therefore, this research is basically different in methodology and basic assumptions from the common LCA studies about WWTPs. It aims to develop a holistic framework for the performance evaluation of WWTPs. This method estimates the equivalent environmental damages, instead of the conventional approach that compares pollution removals. This study also discusses how to include typical pollution loads (BOD, COD, and TSS), quality standards, wastewater reuse, and possible carbon emission within GWF, BWF and CF for accounting for the related damages in LCIA. Consequently, the method puts one step toward simplifying further integrated WWTP evaluations on the basis of WEN thinking.

METHODS

This study used a three-step method in the calculation. At first, water quality data in the inlet and outlet of WWTPs was analyzed and used for accounting for the environmental footprints (GWF, BWF, and CF). Second, midpoint and endpoint indices (based on ReCiPe database) converted pollution loads and footprints to equivalent cumulative damages in LCIA. Finally, the performance of WWTPs was evaluated and compared based on the developed water-energy index and quantified damages. Figure 1 shows the overall methodology and its steps.

Study area

According to the official report of the Iranian National Water and Wastewater Company, more than 35.2 million inhabitants in cities and rural are connected to the engineered sewage systems with about 250 operating WWTPs (NWWC 2021). These facilities mostly use activated sludge (AS), waste stabilization ponds (SP), and aerated lagoon (AL) for BOD, COD, and TSS removal from domestic wastewater. The aggregate operating capacities of these three processes in Iran consist of about 65, 15, and 13% of the collected wastewater, respectively. The remaining capacity is attributed to other systems, like sequencing batch reactors (WRI 2020).

This research uses the average water quality data (2015–2021) of the influent and effluent of 26 WWTPs located in central Iran, in Esfahan (Yazdian & Jamshidi 2021), Chaharmahal and Bakhtiary, and Markazi provinces (Jamshidi *et al.* 2023).

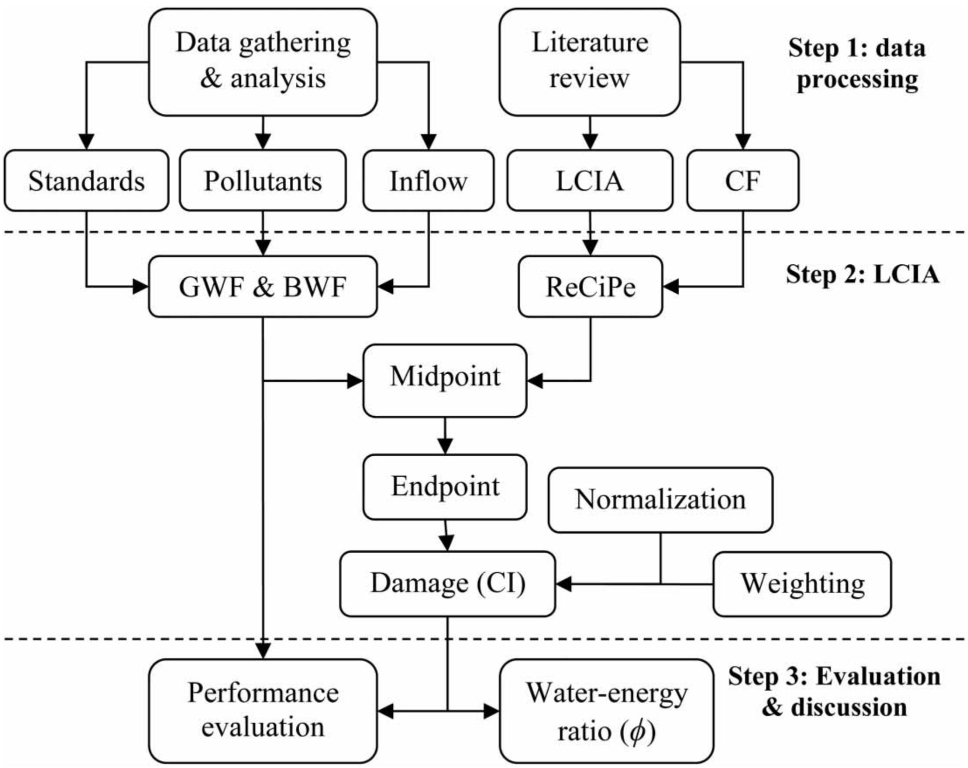


Figure 1 | The overall method of this research with its steps.

These WWTPs are spatially distributed among four climates (Figure 2) and currently treat the municipal wastewater of 23 cities with 1,900–980,000 population-equivalents (PE). Three types of biological treatment units, AS, AL and SP are used and there are different regional demands for recycling their treated wastewater (Table 1). We also classified the capacity of WWTPs in accordance with Goliopoulos *et al.* (2022), in which small WWTPs receive an inflow of PE $\leq$50,000, whereas large WWTPs serve a higher ($>$50,000 PE) population (Goliopoulos *et al.* 2022). For comparing pollution removals in studied WWTP, we also used box plots and analysis of variance by Minitab 19 software.

Water footprint

Since GWF is the embodied water for pollution assimilation, it should be accounted for based on the net pollution discharged from a manufacturer, e.g. industry or agriculture (Roudbari *et al.* 2023). Nevertheless, WWTP is normally recognized as a service that inherently reduces the pollution of wastewater and its performance assessment is reliant on its pollution removal. Regarding the new perspective, we can consider WWTP as a water factory. However, its net GWF from the inlet (raw wastewater) to the outlet (treated wastewater) is still negative ($GWF_{in} > GWF_{out}$). In order to solve the problem, we recommend accounting the GWFs twice based on the pollution loads in the inlet (GWF_{in}) and outlet (GWF_{out}). Here, the performance equals the GWF reduction (%) from the inlet to the outlet (Equation (1)). In addition, their corresponding environmental damages (see Equations (7) and (8)) should also be calculated twice in accordance with GWF_{in} and GWF_{out}. For multiple pollutants, GWF is the maximum value among water quality parameters (Equation (2)). It means that in WWTPs, GWF_{in} can be reliant on a quality parameter (e.g. BOD), while GWF_{out} can be dependent on another parameter (e.g. TSS). It does not necessarily require using identical water quality parameters for GWFs in the inlet and outlet. Equation (2) defines GWF factors (Tahar *et al.* 2018; Varol & Balcı 2020).

$$\text{Removal (\%)} = \frac{\text{inlet} - \text{outlet}}{\text{inlet}} \times 100 \tag{1}$$

$$\text{GWF} = \frac{1}{\text{PE}} \max\left(\frac{C \times Q}{C_{max} - C_{nat}}\right) i \tag{2}$$

Here, GWF is the grey water footprint (m^3/ca.) of a WWTP, C is the pollutant (i) concentration in inlet or outlet (mg/L), Q is the average flow rate (m^3/yr) of wastewater, PE is the population-equivalent that represents the capacity of WWTPs (person), C_{max} and C_{nat} are the allowable and natural concentration of pollutants (mg/L), respectively in t receiving environment as assumed in Table 2. For wastewater reuse (e.g. industries or irrigation), C_{max} is different with discharge conditions. The consumer of reclaimed water is now responsible for its quality. This difference in C_{max} departs the GWF of discharged or reused wastewater.

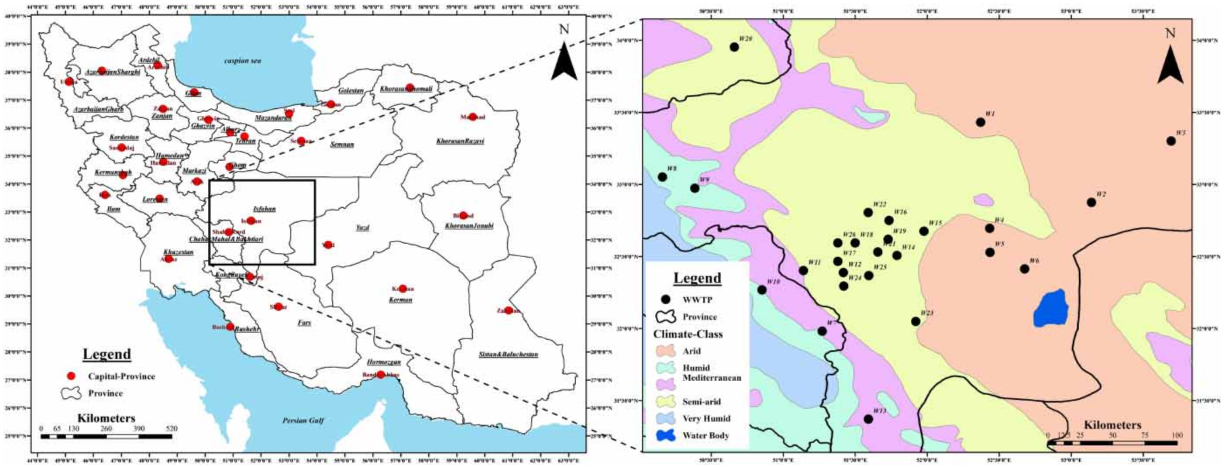


Figure 2 | Location and spatial distribution of studied WWTPs in different climates.

Table 1 | The specifications of studied WWTPs

WWTPs	ID	Process	Population	Flow (m^3/d)	Climate	Reuse alternative
Ardestan	W1	SP	42,105	955	Arid	Irrigation /aquifer recharge
Naeen	W2	SP	39,261	3,045	Arid	Irrigation /aquifer recharge
Anarak	W3	SP	1,903	274	Arid	Irrigation /aquifer recharge
Kuhpayeh	W4	SP	23,674	1,026	Arid	Irrigation /aquifer recharge
Harand	W5	AL	8,455	1,140	Arid	Irrigation /aquifer recharge
Varzaneh	W6	SP	29,718	2,493	Arid	Irrigation /river rehabilitation
Borujen	W7	AS	57,071	7,370	Humid	Irrigation /industrial application
Buyin	W8	SP	24,163	1,432	Humid	Irrigation /aquifer recharge
Daran	W9	AL	20,078	1,563	Humid	Irrigation /aquifer recharge
Shahrekord	W10	AS	190,441	27,700	Humid	Irrigation /industrial application
Baghbahadoran	W11	AL	10,279	1,144	Mediterranean	Irrigation
ZarrinShahr	W12	AL	55,817	9,638	Semi-arid	Irrigation
Semirom	W13	AL	74,109	2,636	Mediterranean	Irrigation
Baharestan	W14	AS	79,023	13,046	Semi-arid	Irrigation
E. Isfahan	W15	SP	490,315	54,238	Semi-arid	Irrigation /river rehabilitation
N. Isfahan	W16	AS	980,630	181,028	Semi-arid	Irrigation /aquifer recharge
FooladShahr	W17	SP	88,426	12,523	Semi-arid	Irrigation /industrial application
Ghahderijan	W18	AL	34,226	1,139	Semi-arid	Irrigation
S. Isfahan	W19	AS	653,753	106,399	Semi-arid	Irrigation /river rehabilitation
Delijan	W20	SP	40,902	8,400	Semi-arid	Irrigation /industrial application
SepahanShahr	W21	AL	70,557	12,088	Semi-arid	Irrigation /aquifer recharge
ShahinShahr	W22	AS	352,001	42,994	Semi-arid	Irrigation /industrial application
Shahreza	W23	SP	159,797	7,773	Semi-arid	Irrigation
Safayieh	W24	AS	50,137	1,846	Semi-arid	Irrigation /industrial application
Mobarakeh	W25	AL	150,411	4,458	Semi-arid	Irrigation /industrial application
Najafabad	W26	AL	319,205	4,436	Semi-arid	Irrigation /industrial application

Table 2 | Assumed C_{max} and C_{nat} for GWF accounting in two scenarios

	C_{max} (mg/L)			
Parameters	Discharge	Reuse	C_{nat} (mg/L)	Reference
Total suspended solids (TSS)	50	100	10	DOE (2016)
Chemical oxidation demand (COD)	20	60	5	
Biochemical oxidation demand (BOD)	10	30	2	

It is noteworthy that the effluent of WWTPs is fundamentally classified as blue water (Hoekstra *et al.* 2011). It means that WWTPs produce blue water (Jamshidi 2019) for a purposeful reuse or release to the environment. In both scenarios, the WFs should be calculated by Equation (3) which is not different to the main WF definition (GWF + BWF) (Hoekstra *et al.* 2011). It only emphasizes that BWF (Equation (4)) acts as a water resource or a replacement for water abstracts (m^3/ca.).

$$\mathrm{WF} = \mathrm{GWF} - \mathrm{BWF} \tag{3}$$

$$\mathrm{BWF} = \frac{Q}{\mathrm{PE}} \tag{4}$$

In some cases, the WF of WWTPs may become negative (WF <0) where the effluent is well treated so that $C < (C_{\max} - C_{\text{nat}})$. On this condition, the WWTP does not embed water, it works as a water supply instead.

Carbon footprint

This study briefly reviewed the literature to estimate the average CF of WWTPs with respect to their conventional secondary treatment units. Accordingly, we took two assumptions about the main source of CF in WWTP, and the approximate CF of WWTP.

Energy utilization and GHG emissions from mechanically aerated (e.g. AS) or natural-based (e.g. SP) units are different. Here, we assumed that the CF of WWTPs is mainly dependent on the type of biological treatment units (Ramachandra & Mahapatra 2015; Wu *et al.* 2022). Maktabifard *et al.* (2020) based on 6 full-scale WWTPs concluded that energy source affects less than 50% of the total CF of WWTPs (Maktabifard *et al.* 2020), while direct GHG emission from biological treatment units has higher impacts. Later, Goliopoulos *et al.* (2022) concluded that aeration with secondary treatment units consumes more than 70% of energy in WWTPs (indirect GHG), whereas pretreatment uses only 10% (Goliopoulos *et al.* 2022). These two studies imply that if we exclude the related CF of sludge management, chemical additives, and construction materials, the majority of direct and indirect GHG emissions from operating WWTPs are attributed to the aeration and biological treatment unit. In Poland, India and England, less than 50% of the CF of WWTPs is due to its construction materials (Singh *et al.* 2016; Zawartka *et al.* 2020), but it is also noted that this ratio is reducing in operating WWTPs due to electricity consumption and direct CO_2 emissions (Zawartka *et al.* 2020). According to a recent study on large Italian WWTPs, the endogenous decay of wastewater in secondary treatment units includes 52% of direct GHG emissions from WWTPs, whereas energy consumption takes 47% of indirect emissions (Riccardo *et al.* 2023).

According to the literature, the CFs of two WWTPs in Poland having AS with 250,000 and 60,000 PE range between 17 and 39 grCO_2e/PE (Maktabifard *et al.* 2019). However, they recommended multiplying CF with 2–3 wherever N_2O is also included for estimating GHG emissions. Hence, a relatively large WWTP (>50,000 PE) with conventional AS has CF >60 grCO_2/PE. In Scotland, the annual average GHG emissions of 16 WWTPs were about 7–108 grCO_2e/PE. It was mainly due to the pumps, excess sludge, and additives used for denitrification (Gustavsson & Tumlin 2013). In Spain, the average CF of 4 WWTPs was about 280 grCO_2e/m^3 including sludge management (Gómez-Llanos *et al.* 2020). In Greece, aeration for pretreatment and biological treatment has CF of about 120 grCO_2e/PE (small WWTPs) and 60 grCO_2e/PE (large WWTPs) with an average ±15% (10–20 grCO_2e/PE) variation for pumps (Mamais *et al.* 2015). In Iran, it was also estimated that WWTPs with mechanical aeration have CF of about 30–36 grCO_2e/PE (Aghabalaei *et al.* 2023). Therefore, we assumed that the average CF of secondary treatment units like AL, AS and SP are 110, 80 and 20 grCO_2e/PE, respectively. These assumptions are in the range of the reviewed studies.

Life cycle impact assessment

ReCiPe 2016 is one LCIA model with a recently revised database (Huijbregts *et al.* 2017). Here, midpoint coefficients estimate the equivalent impacts in different categories, while endpoint coefficients convert these impacts into the major categories of human health and ecosystem (k). The damaged human health is based on disability-adjusted years (DALY), while the ecological impairments are based on the probable number of affected (PAF) species per year (Huijbregts *et al.* 2017). Table 3 shows the details of the applied midpoint and endpoint impact categories in this study. It should be noted that all impact categories of ReCiPe are not necessarily applicable to the performance evaluation of WWTPs. For example, Ionizing radiation and ozone depletion are not related to WWTPs and particulate matter ($PM_{2.5}$) is not a typical measured pollutant. Yet, we categorized GWF as water consumption, while BWF is a water production index for WWTPs.

ReCiPe has three perspectives to evaluate LCIA indices (Tamburini *et al.* 2019). This study used the hierarchical perspective for both midpoint and endpoint indices as it is common for modeling and policy-making in developing countries (Jamshidi & Naderi 2023a). Accordingly, the studied environmental factors of WWTPs are initially converted into equivalent damages by Equation (5) (Jamshidi & Naderi 2023a).

$$U_j = (T \times M \times PE)_j \tag{5}$$

where U is the midpoint index, T is the pollution load (kg/yr) or consumed water (m^3/yr), M is midpoint conversion coefficient, and j is the affected category media (e.g. marine or terrestrial ecosystem). Afterwards, ReCiPe uses endpoint coefficients

Table 3 | Impact categories and the conversion coefficients of ReCiPe (Huijbregts *et al.* 2017)

	Midpoint			Endpoint		
Impact category	**Influencing factor**	**Coefficient (*M*)**	**Unit**	**Coefficient (*E*)**	**Unit**	**Normalization factor (*N*)**
Human health	GWF& BWF	0.5	m^3	2.22×10^{-6}	DALY/m^3	1.96×10^{-4}
	GHG	20-80-110[a]	CO_2-eq/ca.	9.28×10^{-7}	DALY/CO_2-eq.	7.42×10^{-3}
Terrestrial ecosystem	GWF& BWF	0.5	m^3	1.35×10^{-8}	Species.year/m^3	3.48×10^{-6}
	GHG	20-80-110	CO_2-eq/ca.	2.80×10^{-9}	Species.year/ CO_2-eq.	2.24×10^{-5}
Aquatic ecosystem	GWF& BWF	0.5	m^3	6.04×10^{-13}	Species.year/m^3	6.16×10^{-10}
	GHG	20-80-110	CO_2-eq/ca.	7.65×10^{-14}	Species.year/ CO_2-eq.	6.11×10^{-10}

[a]Assumed as section carbon footprint.

(E) regarding the corresponding human health or ecosystem damages (Table 3) to convert midpoint indices into endpoint index (V) by Equation (6). Thus, the environmental impacts with different units are now aggregated under a unified index as DALY (health) or PAF (ecological).

$$V_j = (U \times E)_j \tag{6}$$

Endpoints (V) are normalized (Z_k) per capita as Equation (7) for each major impact category (k) of human health or ecological damage.

$$Z_k = \left(\sum \frac{V_j}{N_j}\right)_k \tag{7}$$

$$S = \frac{\sum [Z \times W]_k}{PE} \tag{8}$$

Here, S is dimensionless which represents the cumulative impact (CI), N is the normalization factor that converts endpoint damages to equivalent impairment per capita, and W is the weight of major impact categories (Equation (8)). In this study, the weights (W) of human health and ecological damages are assumed 0.4 and 0.6, respectively, similar to the EPI method (Wendling *et al.* 2018).

By means of LCIA framework, we could define a state-of-the-art index (ϕ) that can account for the ratio of equivalent damages of GHG emission from WWTPs (S_{GHG}) to the total impairments of WF (S_{WF}) in these facilities (Equation (9)). Here, S is the CI of GHG and WF.

$$\phi = \frac{S_{GHG}}{S_{WF}} \tag{9}$$

ϕ is dimensionless and represents the hidden damages of GHG emission from WWTPs on the basis of WF damages. For example, $\phi > 0.5$ implies that the WWTP is more risky with its global warming than remained pollutants in wastewater. Conversely, $\phi < 0.5$ highlights the prevailing consequences of wastewater pollution. $\phi < 0$ shows that WWTP has synergy with the environment by recycling water ($S_{WF} < 0$) or clean energy production ($S_{GHG} < 0$). As noted before, the positive and negative impacts of wastewater pollution (GWF) and reuse (BWF) are already considered within WF.

RESULTS AND DISCUSSION

Water footprint

According to the comparative performances of 26 WWTPs on pollution abatement derived from Minitab software (Figure 3), it can be concluded that AS was significantly (P-value < 0.05) more effective on BOD, COD, and TSS removal with 87.1 ± 2% ($P = 0.014$), 87.1 ± 1.4% ($P = 0.002$) and 86.3 ± 1.7% ($P = 0.00$) efficiency, respectively. The average BOD removal for AL and SP were 80.4 ± 2% and 79.5 ± 1.3%, respectively, and their COD removal were 76.7 ± 2.5% (AL) and 75.8 ± 1.9% (SP). However, AL and SP were less effective on TSS removal about 55.1 ± 7% and 54 ± 3.4%, respectively. In

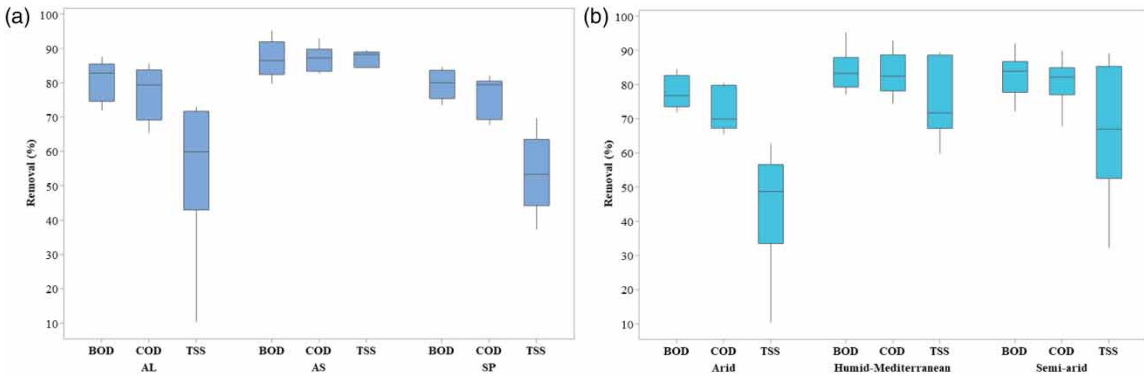


Figure 3 | Pollution removal (%) of WWTPs based on treatment units (a) and climates (b).

addition, WWTPs in arid climates were significantly less effective in COD ($P = 0.022$) and TSS ($P = 0.014$) removal in comparison with other climates, whereas BOD removal was not significantly different ($P > 0.05$).

Reporting pollution removals follows the conventional method for the performance evaluation of WWTPs. GWF can instead take multiple pollutants simultaneously in evaluation combined with the size (*PE*) of WWTPs and the regional conditions (C_{max}) for treated wastewater. For example, Figure 4(a) illustrates the calculated GWF of three pollutants (m^3/ca) in

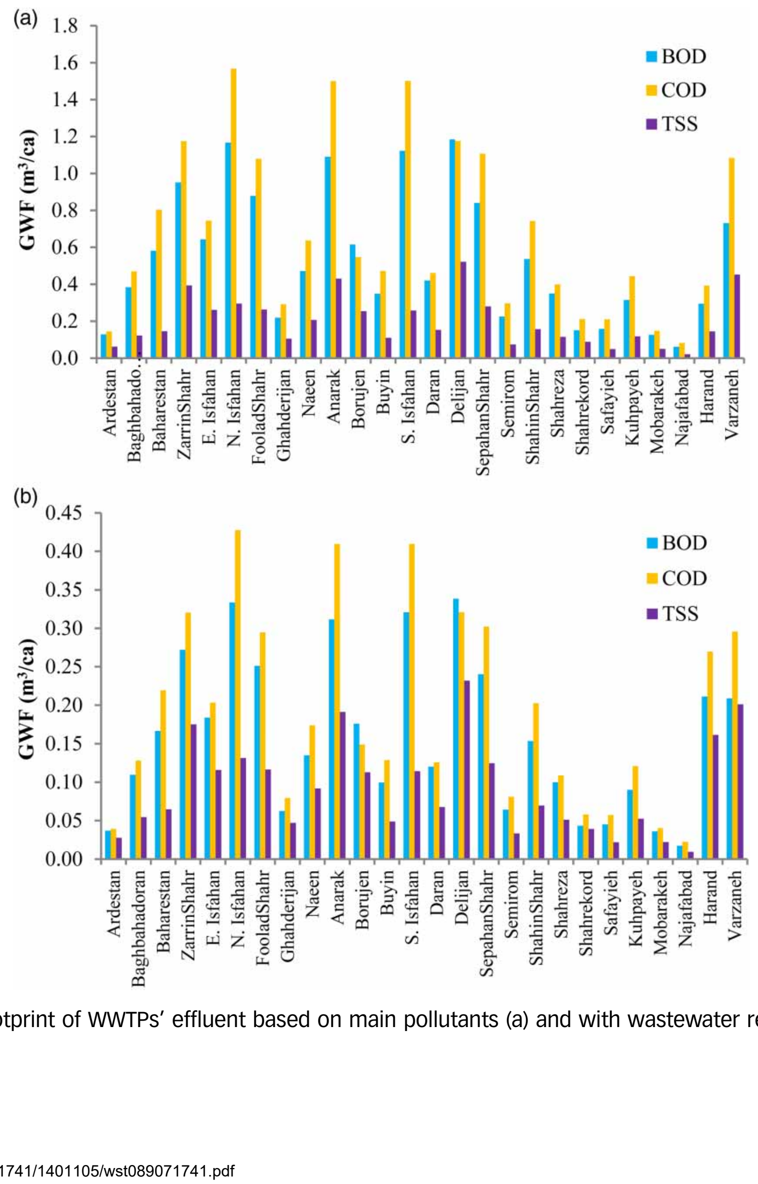


Figure 4 | The grey water footprint of WWTPs' effluent based on main pollutants (a) and with wastewater reuse (b).

the effluent of 26 WWTPs (Equation (2)). Figure 4(b) also re-evaluates these GWFs by wastewater reuse. These figures show that COD is mostly the leading factor in GWF in both scenarios; except for Borujen and Delijan WWTPs. Higher COD concentration (C) in the effluent and the range of defined variables (C_{max} and C_{nat}) in Equation (2) are effective factors that highlighted COD as the leading pollutant in this study. On the contrary, TSS is not a significant pollutant (Figure 4(a)). In this approach, *PE* affects GWFs (m^3/ca), wastewater reuse changes GWF by C_{max} (see Table 2), and biological treatment units (AS, AL and SP) have different removals (see Figure 3). Thus, a multi-pollutant GWF is a more inclusive index for the performance evaluation of WWTPs than the conventional pollution removal assessments.

Figure 5 demonstrates the overall GWF of each WWTP that equals the maximum GWF among 3 pollutants (see Equation (2)). The average GWF of all studied WWTPs equals 0.71 ± 0.09 m^3/ca with the range of 0.08–1.57 m^3/ca. In addition, these facilities supply blue water for irrigation or other purposes (see Table 1). The average produced BWF is 0.1 ± 0.01 m^3/ca which ranges between 0.01 and 0.21 m^3/ca. It is noteworthy that the quality of reused wastewater was previously included in GWF (see Table 2). Accordingly, wastewater reuse reduces the average GWF from 0.71 ± 0.09 m^3/ca to 0.19 ± 0.02 m^3/ca (73%) with the range of 0.02–0.43 m^3/ca (Figure 5). This is due to the fact that treated wastewater is reused with more lenient water quality standards in comparison with direct discharge to the environment. As a consequence, the average ratio of embedded GWF to the produced BWF (GWF/BWF) by reuse decreases by 71% from 7.07 ± 0.45 to 2.05 ± 0.17 for 26 WWTPs (Figure 5). It implies that discharging treated wastewater from a moderate WWTP requires the blue water of at least seven identical WWTPs for its pollution assimilation, whereas, this rate reduces to 2 by reuse. The average WF of WWTPs reduces from 0.6 ± 0.08 m^3/ca (discharge) to 0.094 ± 0.017 m^3/ca (84%) for these facilities on the condition of continuous wastewater reuse.

It should be noted that zero wastewater discharge into the environment is one recommended strategy (Tong & Elimelech 2016). However, LCA needs some reinforcements to quantify the impacts of this strategy. GWF, combined with BWF and CF, has the potential for bridging the analysis above with the environmental impacts and interpreting the cumulative environmental impairments.

Life cycle impact assessment

Figure 6 illustrates the ratio of equivalent environmental impairments of WWTPs on the basis of their WF and GHG emissions by the application of ReCiPe indices. It shows that WF is mostly the leading index. On average, the damages of GHG emissions from biological treatment units include $3.5\% \pm 0.95$ of their accumulated environmental impairments. This ratio ranges between 0.21% (Anarak) and 19.2% (NajafAbad) among the WWTPs. If the benefits of wastewater reuse are included, the average increases to $6.2\% \pm 1.58$. It implies that the ecological or health impacts of pollutants remaining in the treated wastewater are relatively higher than the impacts of GHG emissions from WWTPs.

According to Equation (9), ϕ is the damage ratio of CF per WF in WWTPs (not total impairments). It equals $4\% \pm 1.14$ for discharged treated wastewater and increases to $7.5\% \pm 2.29$ by reuse. Here, the equivalent environmental damage of GHG

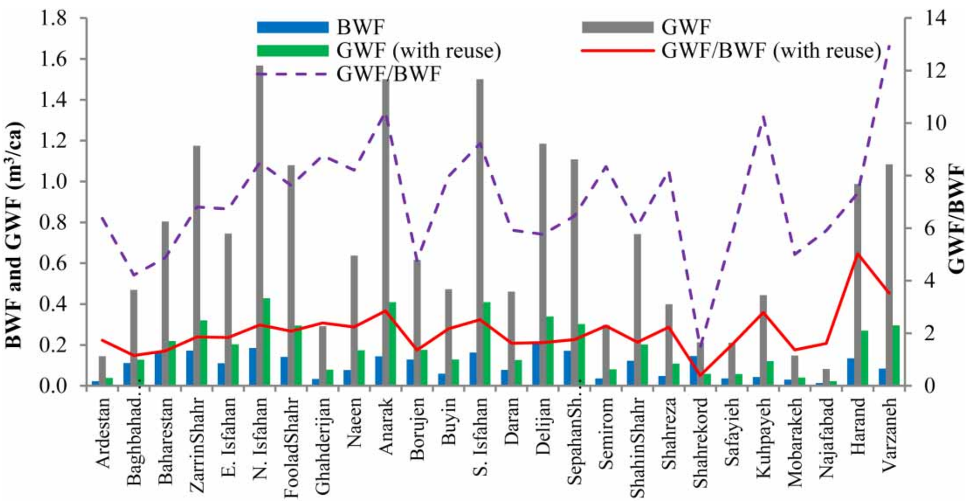


Figure 5 | The GWF and BWF of WWTPs and their ratios with and without wastewater reuse.

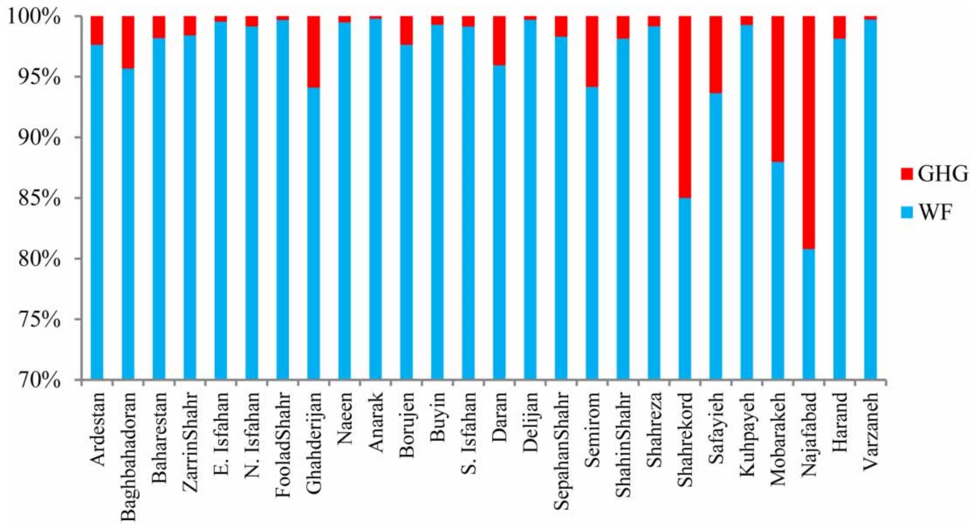


Figure 6 | The ratio of GHG and WF environmental impairment of WWTPs.

emission per WF in WWTPs (ϕ) ranges between 0.2% (Anarak) and 23.8% (NajafAbad) due to the factors like the capacity of WWTPs (large/small), wastewater reuse, and type of treatment units (AS, AL, SP). For example, ϕ for systems with mechanical aeration (AS, AL) is seven folds greater than SP (Figure 7). ϕ is also comparatively 1.4 times greater for smaller WWTPs (<50,000 PE) than larger WWTPs (>50,000 PE). The last result, that smaller WWTP has higher CF, is due to the proportion of energy consumption per treated wastewater. Obviously, larger WWTPs require more energy, for their pumps or sludge aeration, but this consumption would not be necessarily doubled by doubling PE. Energy consumption in WWTPs is elevated step-by-step, by adding equipment in line, when PE linearly increases. Thus, the CF of a small WWTP is relatively significant as similarly verified by the research on 30 WWTPs in Greece (Goliopoulos *et al.* 2022). The estimated ratio of 1.4 was also obtained by their research as the ratio of GHG emission (CO_2/ca.) of small to large WWTPs. Singh *et al.* (2016) concluded that the energy used in smaller WWTPs (KWh/m^3) may even exceed 12 times that of large WWTPs (Singh *et al.* 2016).

ϕ index implies that although aeration has advantages in improving treatment performance (see Figure 3 for AS), it has a secondary impact as GHG emission. In other words, WWTP uses energy to reduce pollution damage to human health and the ecosystem, whereas it adds some damage through indirect GHG emission. Hence, ϕ is an index for WWTPs to compare the impacts of energy consumption respecting the impacts of remaining pollution for a sustainable performance. In other words, ϕ is applicable in WEN studies for WWTPs, or similar facilities. However, its optimal value still requires more case

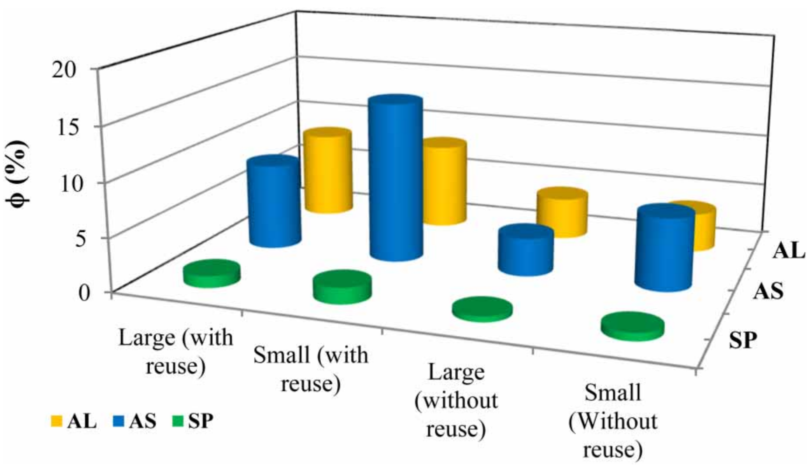


Figure 7 | Equivalent GHG damages per WF (ϕ) regarding WWTPs' units, reuse, and sizes. Greater ϕ shows higher impacts of GHG emission than water pollutants on the environment.

studies. Currently, researchers refer to the water used for energy production for joining CF with WF. For instance, in China, researchers recently estimated that WWTPs require 41grCO_2/m^3 GWF reduced (Gu *et al.* 2016). However, our study joins CF and WF differently through ϕ and its average among 26 WWTPs is 4–7.5%. However, it can be increased, by about 2–10 folds (15–30%), if the CFs of sludge management, construction, nutrient removal, chemical additives, and collection systems are added (Parravicini *et al.* 2016; Zawartka *et al.* 2020).

ReCiPe by its coefficients provides a quantifiable tool to account for the cumulative environmental damages of wastewater discharge. Figure 8 shows the difference between the overall impairment of untreated, treated and reused wastewater for all studied WWTPs. Wastewater treatment and its reuse can reduce the average inlet CI from 5.05 ± 0.7 to 0.83 ± 0.11 (84%) and 0.42 ± 0.05 (92%), respectively. Previously, researchers claimed that wastewater treatment and reuse can yield a 33% reduction in the LCIA of WWTPs regarding its GHG emissions (Miller-Robbie *et al.* 2017).

Combining WF with CF in LCIA can present a more holistic approach for comparing WWTPs' performance. Here, BOD, COD, and TSS removals are no longer separated evaluators. As an alternative, the likely impacts of multi-pollutants and treatment strategies determine the overall performance. According to Figure 9, it is realized that the wastewater reuse strategy

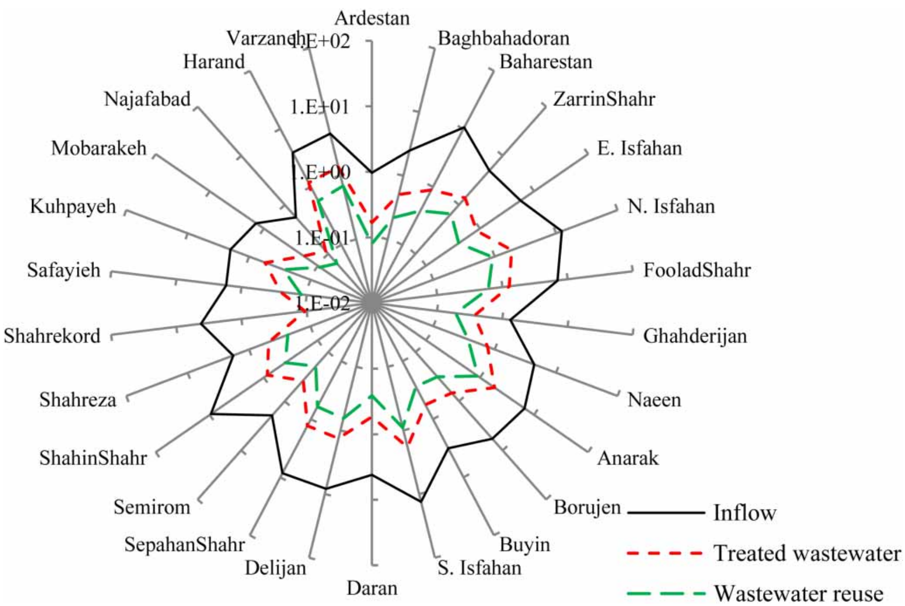


Figure 8 | The impacts of wastewater treatment and reuse on the cumulative environmental damages (CI).

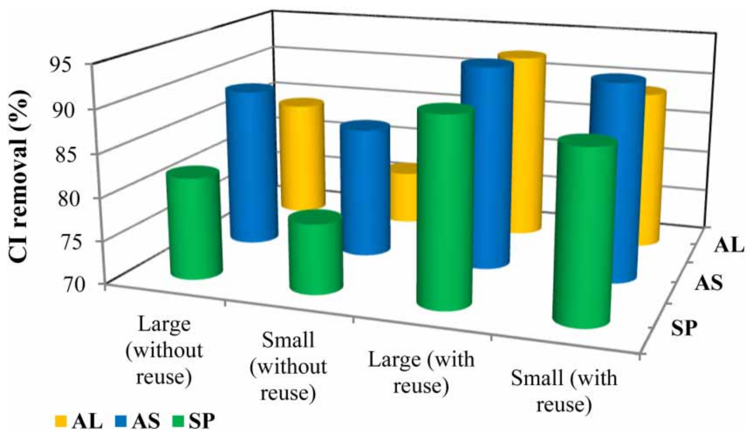


Figure 9 | Overall performance of WWTPs based on their units, reuse, and sizes. More CI removal shows higher performance by the developed method.

(with/without reuse) is more effective on CI removal (%) than the capacity of WWTPs (small/large). Activate sludge has also comparatively higher performance, based on CI removal (%), than SP and AL.

Finally, we should argue that the accuracy of CI is its main weakness and having access to the background databases of LCA may increase this accuracy for integrated performance evaluation of WWTPs in developing countries (Gallego-Schmid & Tarpani 2019). Other LCIA models, such as the Ecoinvent database, can be comparatively used to estimate the overall GHG emission and environmental impacts of WWTPs (Lahmouri *et al.* 2019). According to Yoshida *et al.* (2014), LCA highlights the impact categories responsible for total impairments. Eutrophication, ecotoxicity, and global warming are three main classifications (Yoshida *et al.* 2014). A recent study in Iran showed that ecotoxicity with global warming are critical impact category of a WWTP (Tayyebi *et al.* 2023). It should be noted that this study excluded reporting the related impacts of sludge, nutrients, construction, collection systems, and chemical additives due to their details. These factors can surely improve the developed method with a holistic perspective. Thus, we expect future case studies, in different locations, to follow the same method for evaluating the performance of WWTPs regardless of their spatial specifications. However, due to the details of their operation and impact categories (LCIA), the final results would be different. For example, it is probable that the performance of SP exceeds AS in other locations, or nitrogen becomes a pivotal pollutant for GWF. Yet, we mostly recommend this approach for application in areas where: (1) the government has concerns about WWTPs' sustainability, (2) the monitoring organization, like the Department of Environment, requires a holistic quantifiable index for reporting, and (3) engineers and companies seek for greener options rather than cost-effective alternatives.

CONCLUSIONS

This study primarily developed a quantitative holistic method by introducing WF (GWF–BWF), as a water consumption factor, and CF, as a GHG emission index, in the LCIA of WWTPs. Its application was also verified for comparing the overall performances of 26 WWTPs, in different climates with dissimilar sizes and secondary treatment units. The developed method considers more items with a broader impact assessment perspective for performance evaluation of WWTPs, rather than the conventional pollution removal assessments. Its advantages are: (1) including wastewater reuse, (2) simultaneous multi-pollutant assessment, and (3) considering the net impacts from the inlet to the outlet. These advantages are obtained by adding BWF and GWF in LCIA for cumulative impact assessment. Here, GWF combines multiple pollutants and facilitates complex pollution removal assessments. It also has a water consumption theme that can be easily used in LCA, like BWF. Therefore, this method is recommended for future evaluations, particularly for WWTPs in areas where policy-makers or engineers are seeking sustainable options. Its application is not confined by pollutants, treatment systems, operation, climate conditions or locations. These factors affect indices or midpoints which eventually convert to cumulative normalized damages. Therefore, the developed method can support evaluators with quantified indices for a more inclusive comparison of different WWTPs.

This study also concludes the following:

- For studied WWTPs, GWF played a critical role in the cumulative impacts under LCIA and COD was the major pollutant. The related damages of GHG emissions, from endogenous decay and energy consumption in secondary treatment units, constituted 3.5–6.2% of total impairments. In addition, the average ratio of GWF to BWF was from 2 to 7. It implied how much embedded reclaimed water was required for assimilating the remained pollutants in treated wastewater. GWF/BWF ratio is applicable for reporting the impact of wastewater reuse with respect to its remained pollution content.
- Wastewater treatment and reuse could reduce the average cumulative impacts (CI) of raw wastewater by 92% for studied WWTPs from 5 to less than 1. These CIs implied that, on average, each citizen deteriorated the environment more than 5 folds of his share (footprint) by discharging raw wastewater. WWTPs with reuse could reduce this rate to about $0.4 < 1$.
- Among studied WWTPs, AS was a relatively cleaner treatment unit compared to AL and SP. Larger WWTPs had moderately less CI than smaller WWTPs. Here, reclamation and reuse played a critical role in CI reduction in comparison with treatment units and capacity. Accordingly, policy-makers in the studied area can now plan for upgrading WWTPs if sustainability and nexus thinking become mandatory. It is also noteworthy that the proposed method can have different performance results for WWTPs in other areas, as a matter of operation, pollutants, reuse strategies, climatic conditions, standardization, and LCIA impact categories. However, it does not reject the applicability or validation of the proposed method.
- ϕ was a new index calculated by the developed method. It demonstrated the equivalent GHG damages of WWTPs in proportion to the WF impairments about 4–7.5%. However, further case studies are required to elaborately determine this ratio

by considering other items in WWTPs. Sludge management, construction materials, chemical additives, nutrient removal, wastewater collection systems, etc. can change this ratio.

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AUTHOR CONTRIBUTIONS

S.J. was involved in conceptualization, visualization, supervision, methodology, formal analysis, investigation, and writing. M.F. performed methodology and did formal analysis. H.M. performed methodology and did the investigation.

DATA AVAILABILITY STATEMENT

All relevant data are included in the paper or its Supplementary Information.

CONFLICT OF INTEREST

The authors declare there is no conflict.

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