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Life cycle assessment of two decentralized water treatment systems combining a constructed wetland and a membrane based drinking water production system

Fida Hussain Lakho [a,*], Asif Qureshi [b], Wouter Igodt [c], Hong Quan Le [a], Veerle Depuydt [d], Diederik P.L. Rousseau [a], Stijn W.H. Van Hulle [a]

[a] Laboratory for Industrial Water and Ecotechnology (LIWET), Department of Green Chemistry and Technology, Ghent University Campus Kortrijk, Sint-Martens-Latemlaan 2B, B-8500 Kortrijk, Belgium

[b] Université du Québec en Abitibi-Témiscamingue (UQAT), 445 Boulevard de l'Université, Rouyn-Noranda (Qc), J9X 5E4, Canada

[c] HelloWater, Kastanjeboomstraat 13, B-8550 Zwevegem, Belgium

[d] Flanders Knowledge Center Water (Vlakwa), Leiestraat 22, B-8500 Kortrijk, Belgium

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ABSTRACT

Decentralized (waste) water treatment technologies (DWTS) are suitable for rural areas that are not connected to conventional (municipal) treatment systems due to the longer transport distances. However, the sustainability of DWTS is still debatable. Therefore, the main goal of this study was to use the Life Cycle Assessment tool to perform the life cycle impact assessment of two different decentralized water treatment systems operated in Belgium. The first one was a mobile constructed wetland treating grey water (Scenario 1) at music festivals, coupled to a membrane based drinking water production system (100 m^3 of potable water production out of 400 m^3 of wastewater generation per festival). The second one was a vertical flow constructed wetland treating black water (Scenario 2) at a restaurant (135 visitors/day), also coupled to a membrane system. Comparison was performed with conventional alternatives (PET bottled water supply and a public drinking water supply, respectively). In most impact categories, Scenarios 1 and 2 had roughly an order of magnitude lower impact than their conventional alternatives. Sensitivity analysis was also performed. In scenario 1, the distance travelled for both the mobile constructed wetland and the PET bottles was varied. In Scenario 2 the distance of the restaurant from a drinking water supply and a sewerage system was varied. These results were also encouraging, showing that the DWTS are still environmentally feasible compared to their conventional alternatives at the shortest distance studied (Scenario 1: 175 km and Scenario 2:75 m). Therefore, DWTS can be considered environmentally beneficial under certain conditions.

1. Introduction

Rapid urbanization leads to large-scale transformations of urban and suburban landscapes with their associated environmental issues such as securing drinking water and disposing of wastewater. The resulting increase in water demand and waste generation requires frequent upgrades to water supply and wastewater treatment systems (Kobayashi et al., 2020). Furthermore, factors such as climate change, global warming, water scarcity and the availability of quality water resources are rapidly becoming issues of concern due to increasing public awareness. Therefore, sustainability in water supply and sanitation is increasingly becoming an important issue to address (Anastaselos et al., 2009).

Centralized water treatment systems (CWTS) are becoming a less popular choice for the treatment of polluted waters due to various factors such as climate change, urbanization, and socio-economic conditions across the globe. Subsequently, new and sustainable alternatives are being sought to treat and reclaim water (Arias et al., 2020). The CWTS' infrastructures have been complemented with decentralized infrastructures to ensure water security by, for example, potable water production by greywater reutilization and rain water harvesting systems (Angrill et al., 2017; Hofman-Caris et al., 2019; Słyś and Stec, 2020; Yan et al., 2018). However, decentralized wastewater treatment systems (DWTS) have also gained special attention due to, for instance, ease of

* Corresponding author.
E-mail address: fidahussain.lakho@ugent.be (F.H. Lakho).

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systems' expandability, no need for long-distance water transportation (Leigh and Lee, 2019), and resource recovery possibilities particularly in domestic wastewater by segregating it into black water (BW) and greywater (GW) along with their individual treatment (Ashok et al., 2018; Kobayashi et al., 2020). Moreover, the discharge and/or the source is closer to the treatment systems, thus reducing the construction and operational costs (Lam et al., 2015).

One of the common DWTS are constructed wetlands (CW). These nature-based solutions are most commonly constructed in communities that are not connected to CWTS due to longer distances and high capital costs (Lakho et al., 2020, 2021; Wu et al., 2015). CWs are considered an environmentally and economically viable option for remote communities worldwide (Dominguez et al., 2018; Vymazal, 2018). Next to this, various membrane filtration technologies (such as microfiltration (MF), ultrafiltration (UF) and reverse osmosis (RO)) have proven to be efficient for (decentralized) production of drinking water (Goh et al., 2016; Mamah et al., 2021). Therefore, coupling of CWs with a membrane filtration systems for water treatment and re-use is also being investigated (Lakho et al., 2021, 2020).

To confirm that such a coupling is more sustainable than a centralized treatment option, it is important to perform a life cycle assessment (LCA). LCA is a tool to assess the environmental and socio-economic consequences of various kinds of products and projects. LCA can be used to improve the technologies and to help stakeholders as well as decision makers in selecting a sustainable and reliable technology (Arias et al., 2020). Therefore, several LCA studies have been performed to assess the socio-economic and environmental impacts of DWTS. For example, Gattringer et al. (2016) evaluated water treatment with CWs using LCA for various factors (emissions and corresponding environmental impacts) along with other environmental factors. In addition, GHG emissions associated with centralized and decentralized systems are also studied through LCA. For example the carbon footprint of a municipal water treatment system for non-potable reuse was compared with a decentralized system (Kavvada et al., 2016). In this context, Strategies et al. (2019) recommended decentralized water treatment systems in order to reduce GHG emissions. Kobayashi et al. (2020) presented a comparison of decentralized greywater management systems (membrane bioreactor (MBR) and CW) at different scales focusing towards global warming potential (GWP), eutrophication potential (EUP) and human health-carcinogenic potential (HHCP), and concluded that the CW scenarios (community and neighbourhood scales) outperformed the MBR and business-as-usual (BAU) scenarios for greywater treatment. Furthermore, the scale of decentralized systems, quantity of water reused and mix of electricity technologies all played important roles in determining GWP, EUP and HHCP values.

In this study, a comparative LCA was conducted to assess the environmental impacts of two different DWTS for treating and reutilizing grey and black water for drinking purposes with their conventional alternatives. The performance of both DWTS has already been monitored and reported effective for the treatment of wastewater and production of potable water (Lakho et al., 2021, 2020). To identify and reduce any discrimination or discrepancies between the DWTS and their conventional alternatives, a sensitivity analysis was also performed by varying the distance (a governing factor in both scenarios) to their sources of potable water supply. So far, there are very few studies focused on the LCA of a fully closed water cycle. Therefore, this study is unique in terms of determining the sustainability of such systems.

2. Methodology

2.1. ISO standards and principles for LCA steps

LCA international standards were published as a part of the series ISO 14000 Environmental Management. AS/NZS ISO 14040 *Life Cycle Assessment-Principles and Framework* and AS/ NZS ISO 14044 *Life Cycle Assessment-Requirements and Guidelines* were introduced in Australia 1997 and subsequently updated in 2006 (ISO, 2006; Zuo et al., 2017). The LCA process is performed in four steps as described in ISO 14040 guidelines (ISO, 2006), namely goal definition and scoping, life cycle inventory analysis, impact assessment and interpretation. Following is the definition of each step.

The goal and scope of the study along with the functional unit and system boundaries are defined in the first step. The major challenge in an LCA study is to develop the model, therefore the definition of goal and scope deals with this problem (Goedkoop et al., 2009). The system boundaries set a certain area of scope that needs to be assessed, and the functional unit is important in terms of material quantification of inputs and outputs for the comparison of two products or systems (Cruz-Diloné, 2014; Goedkoop et al., 2009; Rebitzer et al., 2004). In the second step, the used inputs and outputs materials, energy, waste and emissions into air, water and soil is identified and quantified during the different project's activities (construction, operation and demolition) based on defined functional unit (Cruz-Diloné, 2014; Frances., 2013). In addition, this step involves the characteristics of data collection and calculation procedures. During the third step namely impact assessment step, the magnitude of potential environmental impacts set during inventory analysis are identified in different midpoint and endpoint categories (Curran, 2006). Moreover, the LCIA translates emissions and resource extractions into a limited number of environmental impact scores by means of so-called characterization factors (Milousi et al., 2019). The various environmental impact categories considered for LCIA are abiotic resources, land use, global warming, ozone layer depletion, acidification, eutrophication, ecotoxicological impacts, fossil depletion etc. (ISO, 2006). Interpretation is the fourth and final step of the LCA framework, where findings from the LCI and LCIA are evaluated, interpreted and summarized in order to help decision makers for taking concrete decisions.

2.2. Goal and scope

The aim of this comparative study was to evaluate the environmental performance of the DWTS systems and their conventional alternatives. Two DWTS for the treatment of grey and black water and the production of potable water were used to perform an LCA. The LCA was conducted following ISO 14044 (ISO, 2006) in order to evaluate and quantify the potential environmental impacts of the investigated cases.

2.2.1. Scenarios

Scenario 1 concerns a mobile constructed wetland (MCW) coupled to a membrane based drinking water unit, operated at different festivals in Belgium (Lakho et al., 2020), and substituting (part of the) bottled water use for drinking purposes (the conventional alternative). The MCW was operated at different festivals in Belgium, whereas in this study only one festival was considered as an example. Wastewater generation from showers of 400 m^3 over four days of festival in a year (Van Hulle et al., 2008) was taken into LCA, of which a 100 m^3 was treated for drinking purpose through a combination of MCW and membrane filtration (including LED-UV disinfection) and 300 m^3 of wastewater was discharged to the surface water without treatment (Lakho et al., 2020). As an alternative, the membrane filtration system was replaced with conventional PET bottled waters that supply 100 m^3 of drinking water per festival. Therefore, all the generated wastewater (400 m^3 per festival) was discharged to the surface water.

In Scenario 2, a vertical flow constructed wetland (VFCW) coupled to a membrane based potable water production system and operated with a water flow of 4 $m^3.d^{-1}$ at a restaurant (also in Belgium) (Lakho et al., 2021) was compared with its conventional alternative, i.e. a public water supply and sewage system.

2.2.2. Functional unit and system boundaries

The wastewater discharged and treated water per m^3 per year for Scenario 1 and 2 respectively was used as the functional unit in this

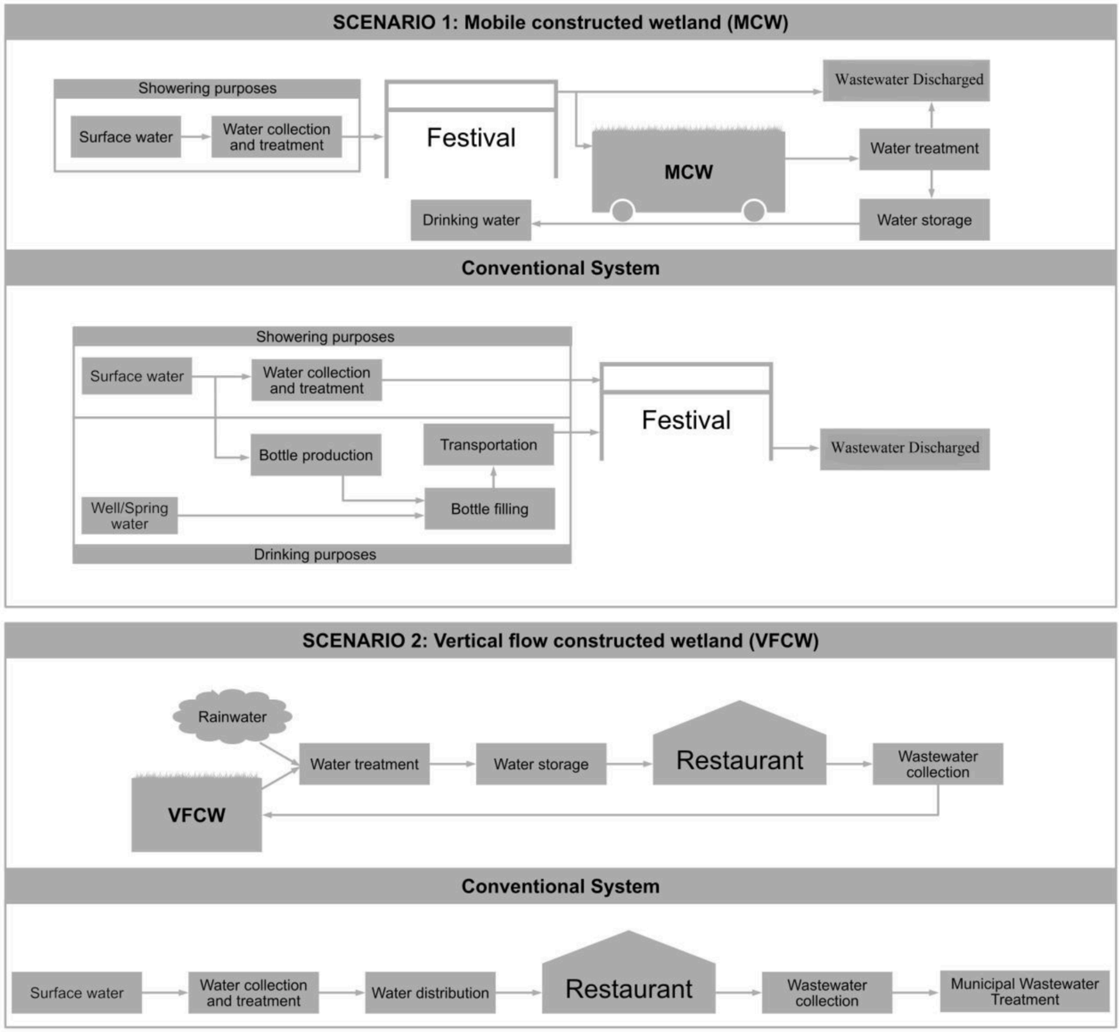

Fig. 1. System boundaries for Scenario 1 and 2.

study for both scenarios because the wastewater treatment and the potable water production are two primary purposes of the studied DWTS. The system boundaries include the collection and treatment of wastewater, and the production of potable water for both DWTS (Fig. 1). However, the system phases were limited to construction and operation of the DWTS and their conventional alternatives, whereas the end of life was not considered as per ReCiPe 2016 midpoint (H) methodology.

2.3. Life cycle inventory analysis

The life cycle inventory (LCI) for this study is shown in Table 1 and 2, and to minimize uncertainties, it was simplified as was also done by other researchers (Opher and Friedler, 2016a, 2016b). The simplification was made in flows and processes that are not already in the database due to the lack of industry grade data, such as complete manufacturing processes of membranes and LED-UV along with the electricity used during their manufacturing. As such, this study was focused to draw attention to the environmental benefits of decentralized water treatment technologies over the conventional water supply and treatment technologies. Therefore, a complete uncertainty and relative sensitive analysis could be future research. In addition, the distribution network for shower water and wastewater collection was not taken into account in both the cases of MCW and PET bottled water supply in Scenario 1. The estimation and assumptions of the designed LCI presented below was reflected in the LCA using the Ecoinvent 3.7 database (Wernet et al., 2016).

The complete data inventory for Scenario 1 and 2 is shown in Tables 1 and 2 respectively. As far as DWTS is concerned, a septic tank as pretreatment, buffer tanks and the main treatment through CW along with collection units were considered (Table 2). In addition, the LCA was performed for a 20 years time period, where the amount of discharged wastewater was estimated as 300 m^3 and 400 m^3 in 4 festival days $year^{-1}$ respectively for MCW and PET bottled water for Scenario 1 (Table 1), and 1040 m^3 $year^{-1}$ of treated water for Scenario 2 as shown in Table 2. Similarly, an activated carbon filter, microfiltration (MF), ultrafiltration (UF) and reverse osmosis (RO) membranes were considered for membrane based potable water production units with the same specification as described in earlier studies (Lakho et al., 2021, 2020). Furthermore, the life span of membranes was assumed to be five years, whereas activated carbon was replaced every 3 years. LED-UV lamps were installed for the disinfection prior potable water supply (Tables 1 & 2).

In order to compare the DWTS for Scenario 1, bottled water (as a conventional alternative) was assumed to be transported over a distance of 500 km, respectively from the supplier to festival (250 km) and transporting of empty bottles to the waste handling site (250 km). For Scenario 2, a concrete sewer line and PVC pipes of 300 m length each were assumed to connect the restaurant with the nearby conventional (public) system (Table 2). The data for building the inventory of both scenarios was taken from the actual studies described previously (Lakho

Table 1
Summary of the life cycle inventory for Scenario 1 (For detailed calculations see Section 1–3 and Table S1–S4 (Supplementary material))

MCW	Input	Quantity	Size	Use
Construction	Cement. Portland	0.3 kg		Formation of mortar for binding purposes within CW
	Cubitainer	1	1000 L	Effluent collection
	Lava rock	3750 kg		CW substrate
	Rockwool	1500 kg		CW substrate
	Silicone product (Pipe)	0.01 kg		CW's hose
	Steel. chromium sheet 18/8	500 kg	2 m x 75 mm x 3 mm	Truck sheet
	Steel rods for support structure	170 kg	7.62× 10.16 cm_sheet-0.32 cm	For supporting truck sheet
	PVC_Pipe	13 m	2.5 inch dia	Pipeline in CW
Operation	Pump (0.81 KW)	1	Running hours 485 hr/20 years	Feed water pumping to CW
Membrane based drinking water unit				
Construction	Microfiltration membrane (MF. HYDRA brand)	4	1.7 m^2	Purification of MCW's effluent
	Activated carbon (brand FA100)	6	Filled 12.6 L	Purification of MCW's effluent
	UF membrane (Polymem. type UF 35 G S2F)	4	4.5 m^2	Purification of MCW's effluent
	RO membrane (DOW FILMTEC BW2530)	4	2.6 m^2	Purification of MCW's effluent
	LED-UV lamps (Aquisense. type Pearl Aqua micro 12 C)	3		Water disinfection
	Framework (Steel)	20 kg		Structure for holding membranes, pumps & other inventory
Operation	Pumps (0.37 KW)	3		For water flow into membranes
	Energy consumption	21 kWh/ year	420 kWh/ 20 years	Pumps and LED-UV lamps
	Wastewater generated	400 m^3/ year	8000 m^3/ 20 years	
	Wastewater treated	100 m^3/ year	2000 m^3/ 20 years	
	Wastewater discharged	300 m^3/ year	6000 m^3/ 20 years	
PET bottle supply				
Construction	PET bottle	2 million	2000 m^3 (water)/20 years	
Operation	Truck Engine	1	3.5–7.5 metric ton. Euro V Engine. Lorry	Transportation
	Wastewater generated	400 m^3/ year	8000 m^3/ 20 years	
	Wastewater discharged	400 m^3/ year	8000 m^3/ 20 years	

et al., 2021, 2020). However, some assumptions were made for the calculation of the amount of construction materials such as cement, PVC (pipes etc.), excavation works and substrate by following previous studies as shown in Table S1 (Supplementary material). Fig S1 and S2 (Supplementary material) show the illustrations of MCW (Scenario 1)

Table 2
Summary of the life cycle inventory for Scenario 2 (For detailed calculations see Section 1–3 and Table S1–S4 (Supplementary material))

VFCW	Input	Quantity	Size	Use
Construction	Septic tank (Concrete)	1	15 m^3	Wastewater sedimentation
	Buffer tank (Concrete)	1	2 m^3	Buffering purpose
	Concrete slab	7.78 m^3		VFCW's walls and bed
	PE Liner	1076 kg		Used at the bottom of VFCW to block the water seepage
	Lava rock	26250 kg		VFCW substrate
	Rockwool	6480 kg		VFCW substrate
	Pipe Joints	6	2.5 inch dia.	For joining the PVC pipes used in VFCW
	Silicone product (Pipe)	0.5 kg		VFCW's hose
	Water Valve	1	2.5 dia	In septic tank
	PVC_Pipe	27 m	2.5 inch	Pipeline within VFCW
Maintenance	Pump (0.81 KW)	2	Running hours 1940 hr/ pump/20 years	Water pumping
Membrane based drinking water unit				
Construction	Microfiltration membrane (MF. HYDRA brand)	4	1.7 m^2	Purification of VFCW's effluent
	Activated carbon (brand FA100)	6	Filled 12.6 L	Purification of VFCW's effluent
	UF membrane (Polymem. type UF 35 G S2F)	4	4.5 m^2	Purification of VFCW's effluent
	RO membrane (DOW FILMTEC BW2530)	4	2.6 m^2	Purification of VFCW's effluent
	LED-UV lamps (Aquisense. type Pearl Aqua micro 12 C)	3		Water disinfection
	Framework (Steel)	80 kg		Structure for holding membranes, pumps & other inventory
Operation	Pumps (0.37 KW)	3		For water flow into membranes
	Energy consumption	218.4 kWh/ year	4368 kWh/20 years	Pumps and LED-UV lamps
Conventional system with water supply and sewerage				
Construction	Concrete pipe	50.8 cm	300 m	Wastewater discharge
	Excavation	900 m^3		For fixing the concrete & PVC pipes
	Backfilling	773 m^3		To cover the buried pipes
	Remaining soil	123 m^3		Disposed to environment
	PVC pipe	50.8 cm	300 m	Potable water supply
	Septic tank (Concrete)	1	15 m^3	wastewater sedimentation
Operation	Wastewater treatment	20800 m^3		

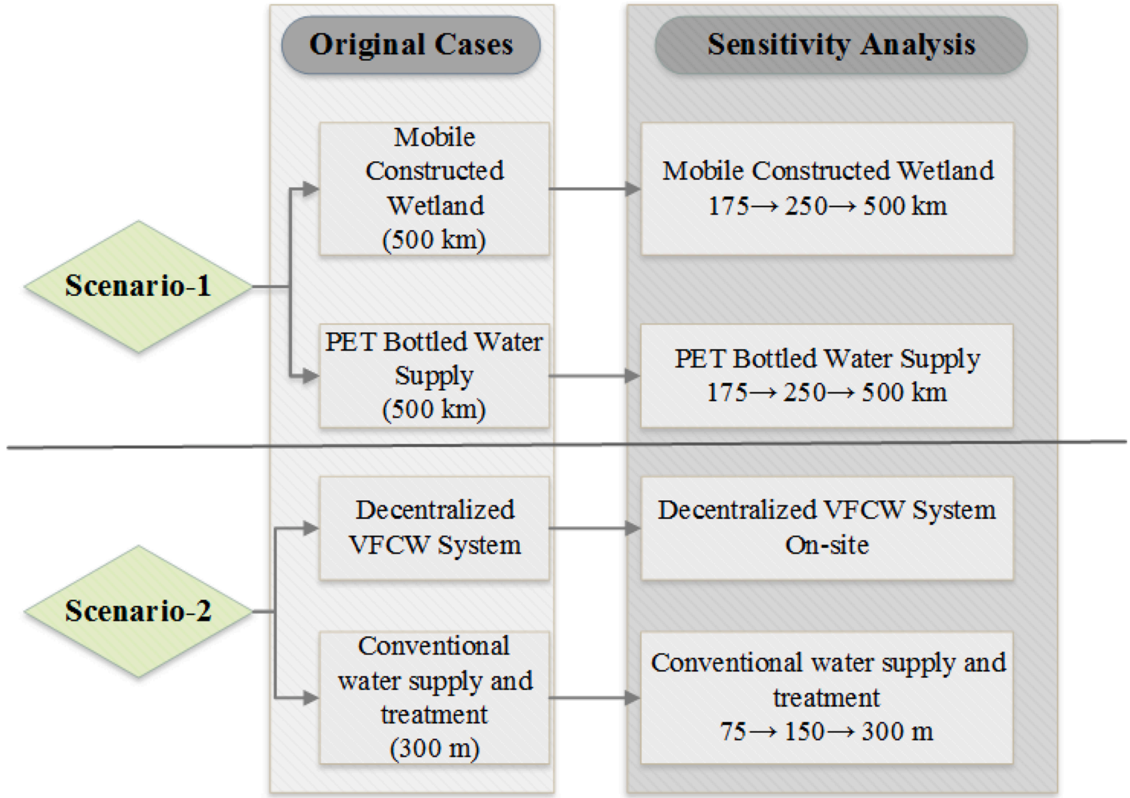

Fig. 2. Flow chart of Scenario 1 and 2 along with their sensitivity analysis.

and VFCW (Scenario 2) along with their various input calculations in Table S2-S4 (Supplementary material). Furthermore, the wastewater quality at festival (Scenario 1) was modelled in order to obtain valid impacts of discharging wastewater into water bodies without treatment as shown in Table S5 (Supplementary material).

2.4. Life cycle impact assessment

The life cycle impact assessment (LCIA) was performed using the software openLCA (Ciroth et al., 2020) and the potential environmental impacts were analysed using the ReCiPe midpoint method 2016 (hierarchist approach) (Goedkoop et al., 2009). Fig. 2 shows a clear flowchart for the various stages taken including the sensitivity analysis for Scenario 1 and 2. The impact categories such as fine particulate matter formation, fossil resource scarcity, aquatic ecotoxicity and eutrophication, global warming, human carcinogenic and non-carcinogenic toxicity, ionizing radiation, land use, mineral resource scarcity, ozone formation, terrestrial ecosystems, stratospheric ozone depletion, terrestrial acidification and water consumption are all included in the ReCiPe method. However, this study focuses on a few selected impact categories including fine particulate matter formation, fossil resource scarcity, aquatic ecotoxicity and eutrophication, global warming, human carcinogenic and non-carcinogenic toxicity, ozone formation and terrestrial acidification, based on previous studies of similar nature (Kobayashi et al., 2020; Opher and Friedler, 2016; Terumi et al., 2018). In order to compare all the impacts at the same scale, the results were

Table 3
Differences between the potential environmental indicators between scenario 1 and 2.

Impact category	Unit	Scenario 1 DWTS	PET Bottle Supply	Scenario 2 DWTS	Conventional water supply and sewerage
Fine particulate matter formation	kg $PM_{2.5}$ eq	$2.43\times10^{+01}$	$3.70\times10^{+02}$	$4.95\times10^{+01}$	$1.27\times10^{+02}$
Fossil resource scarcity	kg oil eq	$2.60\times10^{+03}$	$7.05\times10^{+04}$	$7.18\times10^{+03}$	$2.70\times10^{+04}$
Freshwater ecotoxicity	kg 1.4-DCB	$2.35\times10^{+03}$	$1.66\times10^{+04}$	$3.63\times10^{+03}$	$3.91\times10^{+03}$
Freshwater eutrophication	kg P eq	$3.05\times10^{+01}$	$8.57\times10^{+01}$	$1.14\times10^{+01}$	$3.20\times10^{+01}$
Global warming	kg CO_2 eq	$1.03\times10^{+04}$	$2.49\times10^{+05}$	$2.42\times10^{+04}$	$9.37\times10^{+04}$
Human carcinogenic toxicity	kg 1.4-DCB	$6.72\times10^{+03}$	$2.33\times10^{+04}$	$5.33\times10^{+03}$	$1.86\times10^{+04}$
Human non-carcinogenic toxicity	kg 1.4-DCB	$2.55\times10^{+04}$	$3.47\times10^{+05}$	$3.05\times10^{+04}$	$5.44\times10^{+04}$
Ionizing radiation	kBq Co-60 eq	$6.21\times10^{+02}$	$8.78\times10^{+03}$	$2.26\times10^{+03}$	$5.69\times10^{+03}$
Land use	m^2a crop eq	$4.11\times10^{+02}$	$1.10\times10^{+04}$	$7.73\times10^{+02}$	$2.49\times10^{+03}$
Marine ecotoxicity	kg 1.4-DCB	$3.00\times10^{+03}$	$2.28\times10^{+04}$	$4.55\times10^{+03}$	$5.28\times10^{+03}$
Marine eutrophication	kg N eq	$8.02\times10^{+01}$	$1.22\times10^{+02}$	8.45×10^{-01}	$2.62\times10^{+00}$
Mineral resource scarcity	kg Cu eq	$3.73\times10^{+02}$	$4.65\times10^{+02}$	$2.24\times10^{+02}$	$1.15\times10^{+03}$
Ozone formation, Human health	kg NO_x eq	$2.74\times10^{+01}$	$8.54\times10^{+02}$	$5.63\times10^{+01}$	$2.39\times10^{+02}$
Ozone formation, Terrestrial ecosystems	kg NO_x eq	$2.82\times10^{+01}$	$8.71\times10^{+02}$	$5.92\times10^{+01}$	$2.46\times10^{+02}$
Stratospheric ozone depletion	kg CFC11 eq	1.01×10^{-01}	1.05×10^{-01}	3.24×10^{-02}	3.85×10^{-02}
Terrestrial acidification	kg SO_2 eq	$4.20\times10^{+01}$	$7.80\times10^{+02}$	$1.01\times10^{+02}$	$2.80\times10^{+02}$
Terrestrial ecotoxicity	kg 1.4-DCB	$8.66\times10^{+04}$	$2.27\times10^{+06}$	$4.16\times10^{+04}$	$2.68\times10^{+05}$
Water consumption	m^3	$2.13\times10^{+03}$	$3.10\times10^{+03}$	$2.74\times10^{+02}$	$2.20\times10^{+04}$

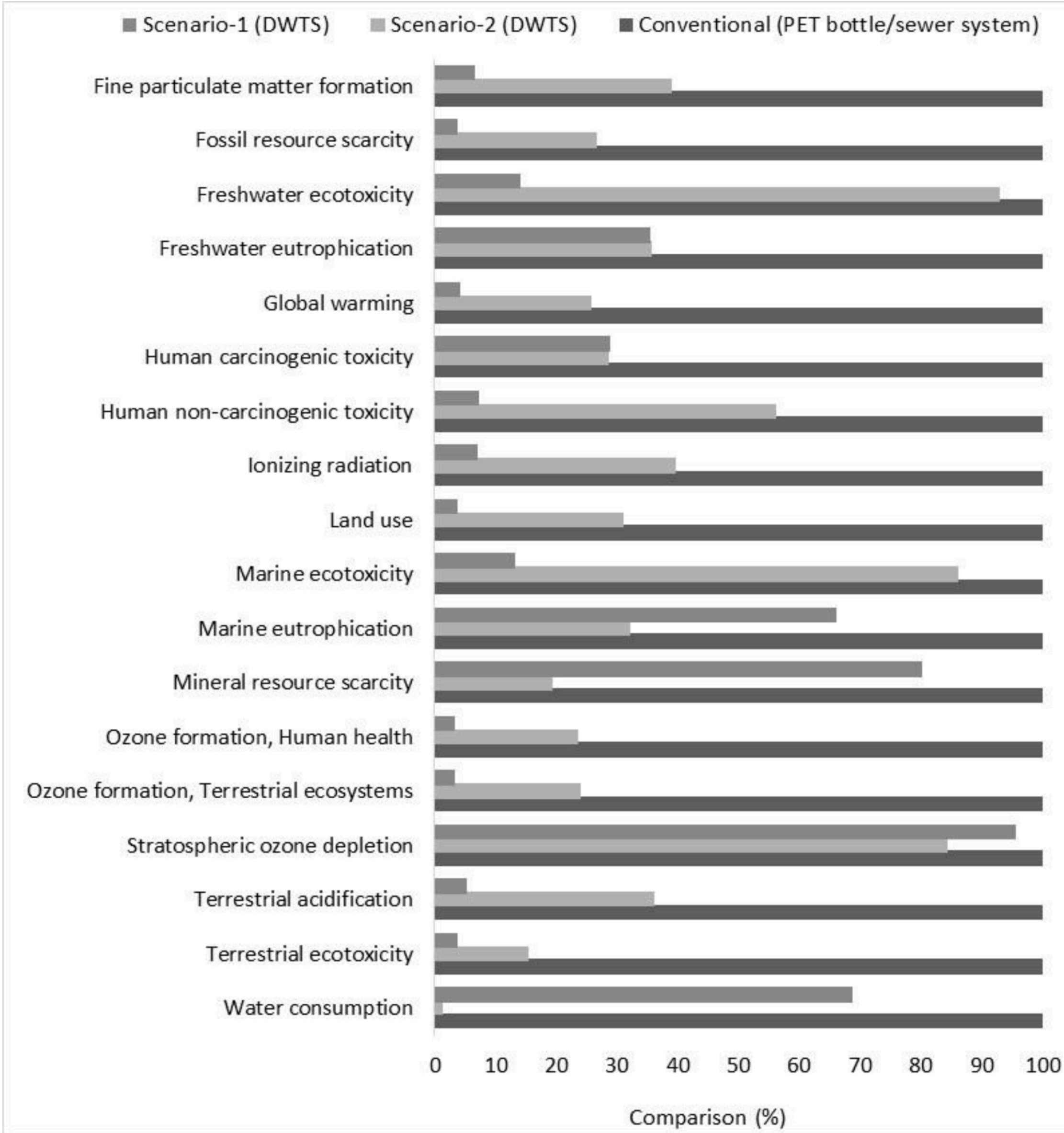

Fig. 3. Contribution of DWTS to each selected impact category for Scenario 1 and 2 against conventional alternatives (both the conventional alternatives are at 100%, that is why only one bar is shown here).

normalized and weighted using ReCiPe.

2.5. Sensitivity analysis

A sensitivity analysis was carried out for Scenarios 1 and 2 by varying the distance from the source of drinking water (bottled water manufacturer and municipal infrastructure, respectively) as shown in Fig. 2. The distance for the sensitivity analysis was selected for Scenario 1 because the construction and operational regime of the MCW were the same and only the distance was varied depending on the location of festival. Therefore, the transportation distance for the bottled water and the MCW in Scenario 1 was changed from actual distance (500 km) to 250 km and 175 km to see the impacts of different travel distances. Regarding Scenario 2, it is obvious that some houses and/or restaurants are far from the conventional facility and vice-versa. Therefore, the distance was selected for sensitivity analysis to check the cut-off distance where conventional system is feasible than the DWTS. Consequently, the distance from the public water supply and wastewater collection was reduced to 150 m and 75 m, thus reducing the required excavation and installation of pipelines. Because DWTS was installed at the restaurant, no further modification was needed in the inventory data. The details of the inventory data can be found in Tables 1 and 2.

3. Results and discussion

3.1. Overall life cycle impact assessment (LCIA)

The results for all impact indicators assessed through ReCiPe 2016 midpoint (H) for 20 years lifetime are shown in Table 3; normalized results are shown in Fig. 3. The results indicate that the DWTS possess much lower impacts (<30%) than their conventional alternative for most of the selected indicators (Fig. 3).

In Scenario 1, the MCW showed overall lower impacts than the PET bottles for almost all the indicators, especially freshwater and marine eutrophication (35–66%), aquatic ecotoxicity (14%), mineral resource scarcity (80%), ozone depletion (95%), water consumption (68%) and toxicities (29%) as shown in Fig. 3. The maximum result is set to 100% of conventional alternatives and the results of the other DWTS are

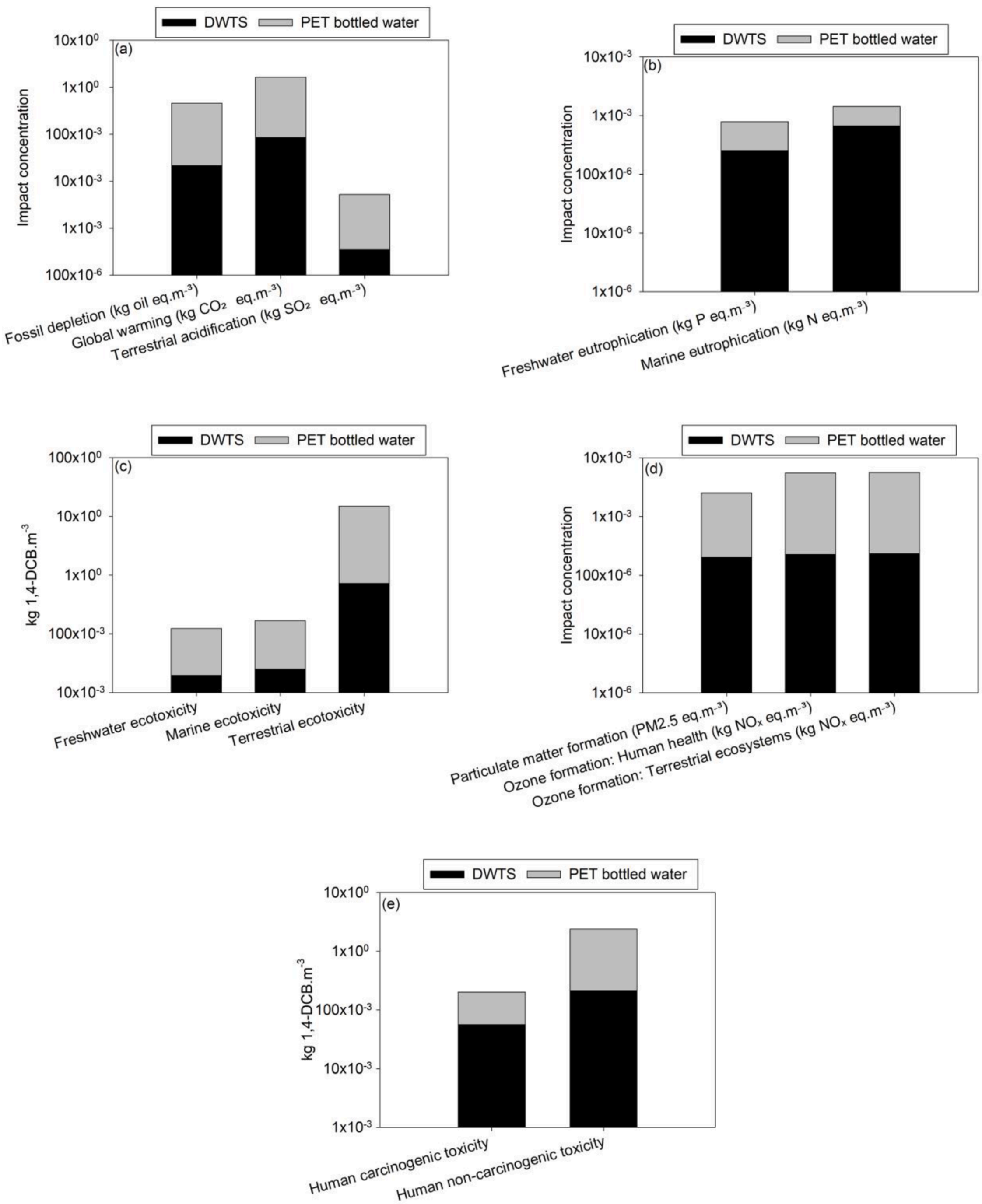

Fig 4. Comparison of fossil depletion, global warming, terrestrial acidification, aquatic eutrophication and ecotoxicity, PM and ozone formation, and human toxicities for DWTS and PET bottled water for Scenario 1 (light grey colour is not the total impact but indicates the extra impact of PET bottled water).

displayed in relation to this result (Fig. 3). The different impacts of aquatic eutrophication and ecotoxicity were mainly due to the different amounts of wastewater discharged into the surface water, and thus different amounts of nutrients and metals (mainly copper and zinc) released. The impacts of ozone depletion were higher due to the various emissions from fossil fuel burning especially for transportation of DWTS and PET bottled water production and transportation.

In Scenario 2, the DWTS has much lower impact as compared to the conventional sewer system, more specifically freshwater and marine ecotoxicity (86–93%), ozone depletion (97%) and non-carcinogenic toxicity (56%) as shown in Fig. 3. The impacts from aquatic ecotoxicity and toxicities in DWTS were rising due to P and N emissions from fossil fuel burning for excavation and construction works. Also the toxicities were increased due to releasing of heavy metals and toxic substances from PVC production and manufacturing of concrete slabs for VFCW along with the energy consumption from fossil based resources.

Each of the impact categories are further discussed in detail in the following sections.

3.2. Scenario 1: sustainability analysis for MCW against PET bottled water

A life cycle impact assessment was carried out to assess the environmental feasibility of MCW producing potable water at the festivals against the same amount of potable water supplied through PET bottled water.

3.2.1. Fossil depletion and global warming potential

The results for fossil depletion and global warming are calculated based on the functional unit (per m^3 of discharged wastewater in a year) and are shown in Fig. 4a. The main contributor to fossil depletion and global warming is PET bottled water supply, since the decentralized system and PET bottled water contributed 8.62×10^{-02} and $1.56\times10^{+00}$ kg CO_2 eq m^{-3}, respectively, for global warming potential, whereas for the fossil depletion the respective contribution was 2.17×10^{-02} and 4.41×10^{-1} kg oil eq m^{-3}. This is in line with another recent study in Flanders, Belgium (Thomassen et al., 2021), where PET bottled water highly contributing towards GWP over a travelling distance of 356 km, of which the main contribution ratio of the obtained GWP impacts was 27%, 25% and 45% for production of PET bottles, transportation of the manufactured bottles to the retailer and distribution (from company to end user), respectively. As far as the fossil depletion is concerned, PET production and the transportation contributed 42% and 47% respectively (Thomassen et al., 2021). Moreover, a higher GWP impact of $6.73\times10^{+02}$ kg CO_2 eq per 12 bottles was found during sustainability analysis of PET bottled water (Horowitz et al., 2018), where the higher impact was attributed to the PET bottle production and large transportation distance of more than three thousand kilometers. In addition, earlier studies showed that the higher GWP impacts of bottled water (either PET or glass bottles) were due to higher carbon dioxide (CO_2) and sulphur dioxide (SO_2) and nitrogen oxides (NO_x) emissions, respectively, during the PET bottle production and transportation of PET bottled water (Lagioia et al., 2012; Papong et al., 2014).

In general, a strong correlation between the fossil depletion and the global warming potential was observed because of the carbon contribution to the atmosphere. Higher differences in the PET bottled water supply could be due to the fuel burnt and electricity used for PET production and its transportation to the destination when compared to the DWTS. However, DWTS were found sustainable and favourable regarding fossil depletion and GWP.

3.2.2. Terrestrial acidification

The conventional alternative of PET bottled water supply contributed more towards terrestrial acidification than the decentralized option as shown in Fig .4a. The main contribution towards acidification from DWTS are the construction of MCW (especially steel sheet and pipes), travelling of truck and energy consumption with the ratio of 62%, 9% and 11% respectively, whereas for PET bottled water this is production (53%) and transportation (46%), probably due to the emissions of substances containing Sulphur dioxides (SO_2) and nitrous oxides (NO_x) (Roy et al., 2012). In this context, Martin et al. (2021) attributed 92% of the acidification impacts of total 11.5 kg SO_2 per ton of PET bottles to the production of the raw material involved in the PET bottles production and transportation. However, Doka (2013) showed that the direct emissions of SO_2 and NO_x has higher impacts of 0.46 kg NO_x and 1.57×10^{-03} kg SO_2 per ton of PET bottles from various processes.

Therefore, the higher terrestrial acidification in this study is due to the PET bottles' production processes. Thus, the DWTS is considered more suitable than the conventional PET bottled water regarding terrestrial acidification.

3.2.3. Aquatic ecosystem impacts: eutrophication and ecotoxicity

The PET bottled water showed more potential impacts as compared to DWTS on the aquatic ecosystem (Fig. 4b). The potential impacts of PET bottled water supply were 5.36×10^{-04} kg P eq m^{-3} and 7.59×10^{-04} kg N eq m^{-3} which was higher than the DWTS (2.54×10^{-04} kg P eq m^{-3} and 6.68×10^{-04} kg N eq m^{-3} respectively). This was due to the phosphorus and nitrogen emissions from discharge of wastewater in surface water and also nitrogen emissions during the energy consumption for transportation and pump operation. The major contributors of freshwater eutrophication for DWTS are due to the discharge of wastewater, construction of the MCW, and the pump (both materials for production and its operation during wastewater treatment at the festivals) that accounted for 84%, 10% and 2% respectively, whereas for marine eutrophication 99% of the contribution was from wastewater discharge into the surface water. Similarly, the higher freshwater eutrophication contributors for the PET bottled water supply were from PET production, discharge of wastewater and transportation accounting for 46%, 40% and 13% respectively, whereas the respective contribution of 11%, 87% and 0.8% was for marine eutrophication. The results of this study are in line with a previous study (Xue et al., 2016) where the researchers attributed the higher eutrophication potential to the high emissions of P and N from sewage treatment system and electricity use during treatment and distribution.

Additionally, the PET bottled water also showed higher impacts of aquatic ecotoxicity than the DWTS (Fig 4c). The major contribution for freshwater ecotoxicity for PET bottled water was from PET production, transportation and wastewater discharged to the surface water with the contribution ratio of 82%, 15% and 1.4%, whereas for marine ecotoxicity the respective contribution was 78%, 19% and 1.5%. The contribution towards freshwater ecotoxicity for DWTS was construction of MCW, wastewater discharge and transportation of truck 76%, 7.7% and 1%, whereas for marine ecotoxicity the respective contribution was 75%, 8% and 1.4%. As far as the ecotoxicity impacts from wastewater discharged are concerned, the major contribution was from metals, especially copper and zinc, that contributed to freshwater ecotoxicity (1.2% and 6.5% respectively) and marine ecotoxicity (1.1% and 7.2% respectively) for DWTS, whereas in PET bottled water supply, copper and zinc contributed for freshwater ecotoxicity (0.3% and 1.3% respectively) and marine ecotoxicity (0.2% and 1.3% respectively) and rest were from PET production and transportation. The outcomes are in line with a previous study (Fang et al., 2016), who attributed ecotoxicity impacts towards heavy metals (primarily zinc and copper) and substances released into the environment from bottle production and distribution processes.

Based on these results, the DWTS is considered an effective alternative to avoid nutrients' and metals' discharge into the environment.

3.2.4. Particulate matter and ozone formation

The DWTS had a lower trend for the PM formation with the respective impact of 2.02×10^{-04} kg $PM_{2.5}$ eq m^{-3} than the PET bottled water supply (2.32×10^{-03} kg $PM_{2.5}$ eq m^{-3}) as shown in Fig 4d. Similarly, the potential impact for ozone formation was higher in the PET bottled water (5.34×10^{-03} kg NO_x eq m^{-3}) than DWTS (2.28×10^{-04} kg NO_x eq m^{-3}) as depicted in Fig. 4d. The stages which contributed to the PM and ozone formation were PET bottle production and transportation to the end users. This means that the large utilization of fuels plays a major role towards PM and ozone formation (Mayer et al., 2021). Because the MCW only involves its transportation to the music festival, DWTS technology had lower potential for particulate matter and ozone formation compared to the PET bottled water supply.

3.2.5. Human toxicity

The differences for human carcinogenic and non-carcinogenic toxicities within DWTS and PET bottled water supply for Scenario 1 are shown in Fig. 4e. Specifically, the PET bottled water has more toxicity with 1.45×10^{-01} kg 1,4-DCB m^{-3} and $2.17\times10^{+00}$ kg 1,4-DCB m^{-3} for carcinogenic and non-carcinogenic toxicities, respectively than the combined MCW and membrane based drinking water system used as decentralized system at the festival (5.60×10^{-02} Kg 1,4-DCB m^{-3} and 2.13×10^{-01} kg 1,4-DCB m^{-3}. The largest contributor towards the human toxicities were PET bottle production and transportation to the storage that accounted for 76% and 21% respectively and only 2% contribution was from wastewater discharged to the surface water. The results are somehow similar with previous studies (Horowitz et al., 2018; Lagioia et al., 2012; Papong et al., 2014), who showed that 91% of the

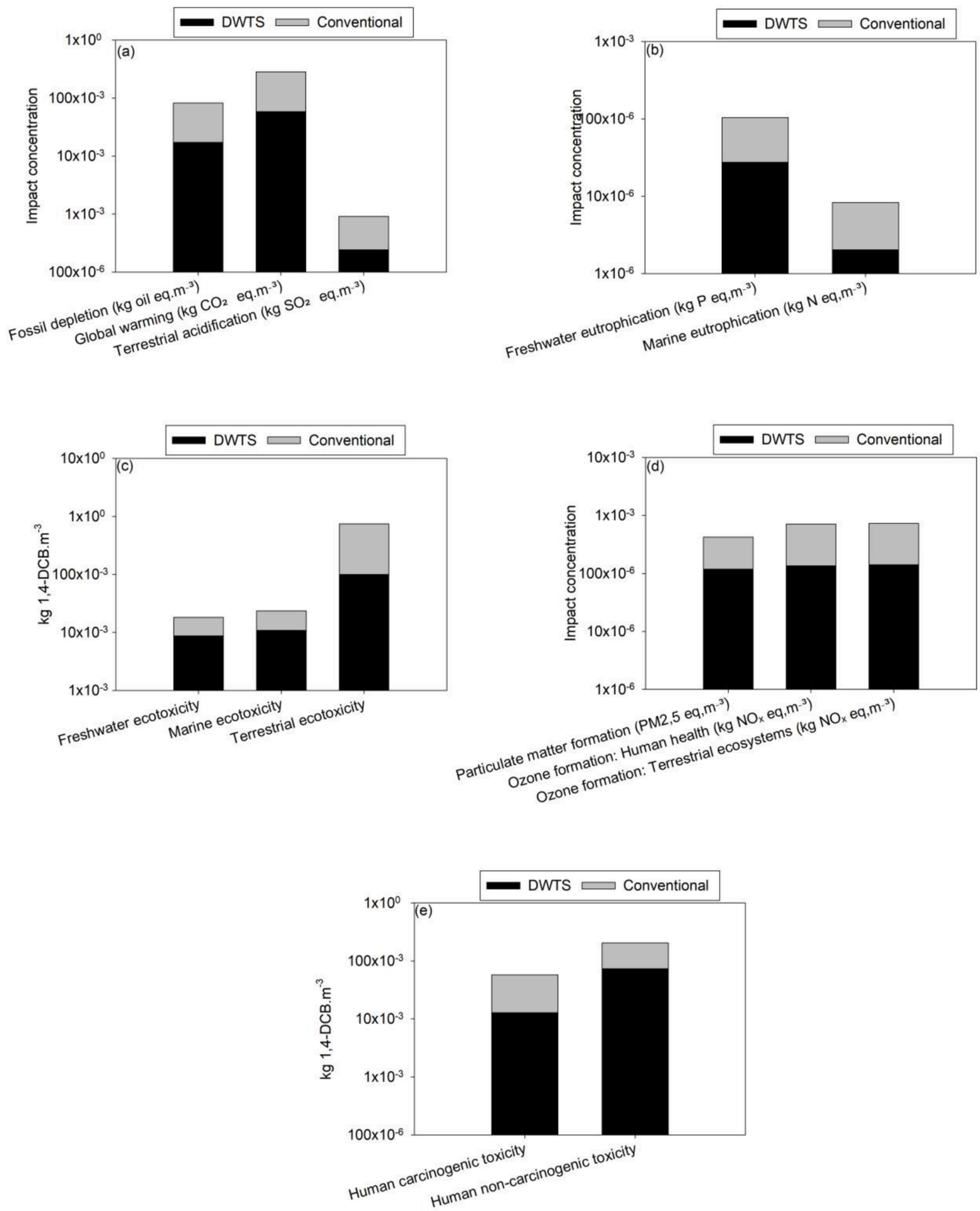

Fig 5. Comparison of fossil depletion, global warming, terrestrial acidification, aquatic eutrophication and ecotoxicity, PM and ozone formation, and human toxicities for DWTS and conventional system for Scenario 2 (light grey colour is not the total impact but indicates the extra impact of Conventional).

contribution towards the carcinogenic and non-carcinogenic impacts was caused by bottle production and transportation. Moreover, 89% of the non-carcinogenic impacts were due to electricity generation and utilization for various purposes such as bottles distribution as well as for refrigerators to store the bottled water (Horowitz et al., 2018). The main contributor of the DWTS in this study for non-carcinogenic toxicity was construction of MCW (steel sheet and pipes production), wastewater discharge and transportation that accounted for 65%, 23% and 2% respectively of the total potential impacts. However, the major contributor specifically from wastewater discharge was zinc that contributed 23% of the total impacts. Despite that, the DWTS showed lower impacts as compared to the conventional option, hence it is considered as a favourable alternative.

It has been observed that the MCW coupled with membrane based drinking water unit for potable water production in Scenario 1 had less potential impacts for all the assessed indicators. Therefore, it possesses a significant potential to treat the greywater generated at the festivals and substitute the PET bottled water supply for drinking purposes.

3.3. Scenario 2: sustainability analysis results for CW against conventional treatment

A remote restaurant connected to a CW to treat its wastewater and fulfil its water demands was studied and was reported to be a viable

Table 4
Sensitivity analysis results for different variants of Scenario 1

Impact category	Unit	DWTS (250 km)	PET Bottle Supply (250 km)	DWTS (175 km)	PET Bottle Supply (175 km)
Fine particulate matter formation	kg $PM_{2.5}$ eq	1.95×10^{-04}	1.75×10^{-03}	1.93×10^{-04}	1.58×10^{-03}
Fossil resource scarcity	kg oil eq	1.97×10^{-02}	2.89×10^{-01}	1.91×10^{-02}	2.43×10^{-01}
Freshwater ecotoxicity	kg 1.4-DCB	1.95×10^{-02}	9.54×10^{-02}	1.94×10^{-02}	9.29×10^{-02}
Freshwater eutrophication	kg P eq	2.54×10^{-04}	5.01×10^{-04}	2.54×10^{-04}	4.90×10^{-04}
Global warming	kg CO_2 eq	8.04×10^{-02}	$1.12\times10^{+00}$	7.86×10^{-02}	9.84×10^{-01}
Human carcinogenic toxicity	kg 1.4-DCB	5.57×10^{-02}	1.20×10^{-01}	5.56×10^{-02}	1.12×10^{-01}
Human non-carcinogenic toxicity	kg 1.4-DCB	2.10×10^{-01}	$1.93\times10^{+00}$	2.09×10^{-01}	$1.86\times10^{+00}$
Ionizing radiation	kBq Co-60 eq	5.08×10^{-03}	4.71×10^{-02}	5.04×10^{-03}	4.47×10^{-02}
Land use	m^2a crop eq	3.11×10^{-03}	4.47×10^{-02}	3.01×10^{-03}	3.75×10^{-02}
Marine ecotoxicity	kg 1.4-DCB	2.48×10^{-02}	1.29×10^{-01}	2.48×10^{-02}	1.24×10^{-01}
Marine eutrophication	kg N eq	6.68×10^{-04}	7.56×10^{-04}	6.68×10^{-04}	7.55×10^{-04}
Mineral resource scarcity	kg Cu eq	3.10×10^{-03}	2.17×10^{-03}	3.09×10^{-03}	1.95×10^{-03}
Ozone formation, Human health	kg NO_x eq	2.02×10^{-04}	3.36×10^{-03}	1.95×10^{-04}	2.76×10^{-03}
Ozone formation, Terrestrial ecosystems	kg NO_x eq	2.09×10^{-04}	3.42×10^{-03}	2.01×10^{-04}	2.82×10^{-03}
Stratospheric ozone depletion	kg CFC11 eq	8.38×10^{-07}	4.58×10^{-07}	8.37×10^{-07}	3.98×10^{-07}
Terrestrial acidification	kg SO_2 eq	3.33×10^{-04}	3.57×10^{-03}	3.27×10^{-04}	3.18×10^{-03}
Terrestrial ecotoxicity	kg 1.4-DCB	6.39×10^{-01}	$7.86\times10^{+00}$	6.14×10^{-01}	$5.96\times10^{+00}$
Water consumption	m^3	1.77×10^{-02}	1.85×10^{-02}	1.77×10^{-02}	1.83×10^{-02}

alternative to the conventional (municipal) water supply and treatment systems (Lakho et al., 2021). This system was compared to a conventional approach where potable water is supplied via the central distribution system and waste water is collected through the municipal sewer system.

3.3.1. *Fossil depletion and global warming potential*

The conventional system contributed 2.25×10^{-01} kg CO_2 eq m^{-3} and 6.49×10^{-02} kg oil eq m^{-3} for global warming and fossil depletion, respectively, which was higher than the decentralized system (5.83×10^{-02} kg CO_2 eq m^{-3} and 1.73×10^{-02} kg oil eq m^{-3}, respectively) as depicted in Fig. 5a. The fossil depletion and GWP from DWTS are related to the construction of VFCW (including concrete slabs and excavation of different tanks) and energy consumption that accounted for 77% and 20% respectively. In contrast, the contribution from the conventional system is mainly due to the concrete pipes (56%), PVC pipes (36%) and excavation for the pipes (1%) respectively, due to the emissions from fossil fuel burning for manufacturing pipes and excavation as well. This can be correlated with the study of Thomassen et al. (2021) who attributed 31 to 43% of the GWP (7.00×10^{-01} kg CO_2 eq m^{-3}) to the fossil fuel burning for energy generation. Especially supplying drinking water by pumping through the distribution network accounts for a large contribution. In addition, the major source of the fossil depletion in the infrastructure and maintenance of the distribution network was due to the excavation and filling for the distribution pipes. Likewise, in another study 77% and 23% of the fossil depletion and GWP were accounted for pumping treated water in the distribution system and maintenance operations, respectively (Barjoveanu et al., 2014). In addition, the previous studies (Amores et al., 2013; Bonton et al., 2012; Lemos et al., 2013; Loubet et al., 2014) analysed the potential environmental impacts of potable water production by conventional treatment systems and concluded that most of the indicators were rising due to critical aspects in water treatment processes, such as using longer pipes and also due to energy usage for water distribution in the supply network.

Similar to Scenario 1, a correlation between the fossil depletion and the global warming potential was observed due to the carbon contribution to the atmosphere. The potential impacts of fossil depletion and GWP were higher in conventional alternatives, which can be attributed to energy consumption for water supply in distribution networks, production of PVC pipes as well as for excavation and filling works. Therefore, the obtained results support the adoption of the decentralized approach over the conventional water supply systems in terms of fossil depletion and global warming potential impact categories.

3.3.2. *Terrestrial acidification*

The sewer system contributed 6.73×10^{-04} kg SO_2 m^{-3} towards terrestrial acidification which was more than the decentralized approach (2.43×10^{-04} kg SO_2 m^{-3}) in Scenario 2 (Fig. 5a). The terrestrial acidification seems correlated to the fossil depletion and global warming potential. Indeed, similar behaviour was also observed by Opher and Friedler (2016) who attributed this trend to the SO_2 and NO_x emissions associated with the fossil fuel burning for excavation works as well as electricity production that was used for pumping water to the distribution network. Additionally, the emissions of SO_2 and NO_x during the production of the PVC pipes used in the distribution network of the conventional alternative also contributed towards terrestrial acidification (Martin et al., 2021).

3.3.3. *Aquatic ecosystem impacts: eutrophication and ecotoxicity*

The conventional technology had higher potential for impacts compared to DWTS for the aquatic eutrophication as shown in Fig. 5b. The major DWTS contributors towards eutrophication were construction works (slab manufacturing, excavation and filling works) for CW (71%) and energy used for pumps (27%) during the operation phase. In contrast, the conventional system discharged nitrogenous and phosphorous compounds from various processes such as construction of concrete pipe (including cement, excavation, filling etc..) (52%) and PVC pipes (35%). Therefore, higher amounts of P and N were released into the environment during these processes, which is in agreement with the findings of Xue et al. (2016). Freshwater, marine and terrestrial ecotoxicity impact potential were less than conventional alternative (Fig. 5c). This could be mainly due to the emissions of heavy metals and sulphuric acids from PVC production and construction of concrete pipes (Opher and Friedler, 2016). Therefore, DWTS alternative for Scenario 2 is more sustainable (having less eutrophication and ecotoxicity potential) than the conventional alternative.

3.3.4. *Particulate matter (PM) and ozone formation*

The conventional alternative showed a higher potential impact for the PM and ozone formation with the respective impacts of 3.05×10^{-04} kg $PM_{2.5}$ eq m^{-3} and 5.76×10^{-04} kg NO_x eq m^{-3} than the DWTS (1.19×10^{-04} kg $PM_{2.5}$ eq m^{-3} and 1.35×10^{-04} kg NO_x eq m^{-3}) in Scenario 2 (Fig. 5d). This could be associated with the emission of NO_x from fuel burnt for the excavation and filling works and electricity consumption for the water supply in conventional system (Mayer et al., 2021). Therefore, the DWTS is also a considerably cleaner alternative regarding PM and ozone formation.

3.3.5. *Human toxicity*

The conventional system in scenario 2 had 4.47×10^{-02} Kg 1,4-DCB

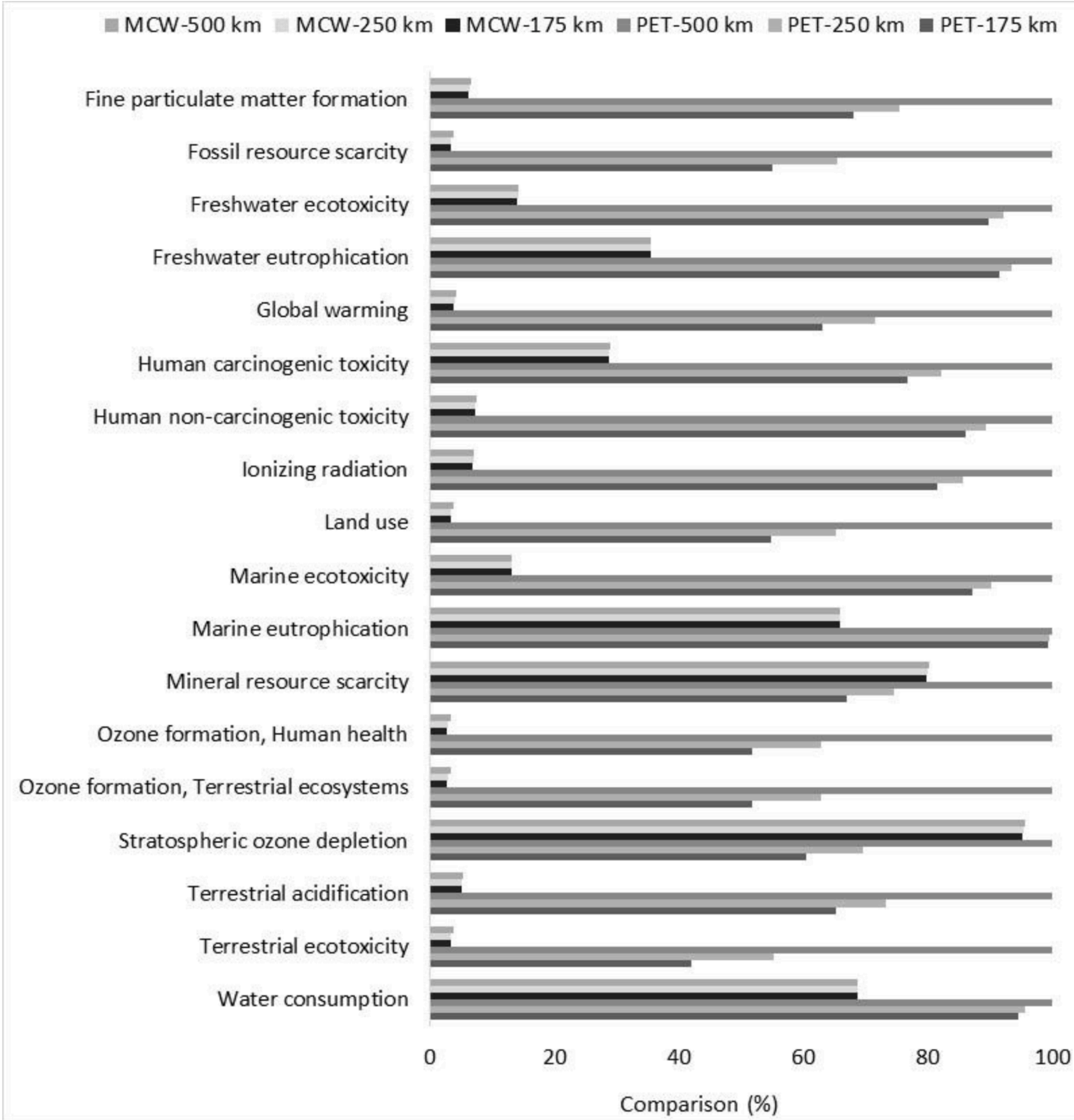

Fig 6. Comparison of potential impacts for all the indicators between different variants of Scenario 1 against PET bottled water at various distance travelled.

m^{-3} and 1.31×10^{-01} Kg 1,4-DCB m^{-3} potential for carcinogenic and non-carcinogenic toxicity, respectively, which was higher than the decentralized technology, amounting to 1.28×10^{-02} Kg 1,4-DCB m^{-3} and 7.33×10^{-02} Kg 1,4-DCB m^{-3}, respectively (Fig. 5e). The largest contributors for human toxicity were due to the release of hazardous substances (including metals like zinc, cobalt and nickel) during energy generation for concrete and PVC pipes and steel production for conventional system as was observed by Risch et al. (2021). Whereas construction works (especially concrete pipes) and pump material along with pump operation contributed 70% and 10% respectively, towards toxicities from DWTS. Thus, potable water production from treatment of black water generates lower risk for human toxicity than the conventional treatment.

3.4. Sensitivity analysis

3.4.1. Scenario 1: MCW against PET bottled water supply at festival

It was noticed during analysis of Scenario 1 that the indicators that caused major impacts were due to fuel burning for transportation of the MCW (DWTS) and PET bottle production as well as its transportation. Therefore, to further assess the impact of travel distance, a sensitivity analysis was carried out by varying the travel distance for the DWTS and PET bottled water. The consumption of fuel was also reduced accordingly. The potential impacts for all the variants for Scenario 1 are shown in Table 4 and their contribution comparison is shown in Fig. 6 where the maximum result is set to 100% and the results of the other variants are displayed in relation to this result.

3.4.1.1. Fossil depletion and global warming. The potential impacts of fossil depletion and global warming for different variants are shown in Table 4 and Fig. S3 (Supplementary material). It is evident that the fossil depletion and global warming potential were directly affected by varying travel distance. For instance, the fossil depletion was reduced from 2.17×10^{-02} kg oil eq m^{-3} for the DWTS (500 km) to 1.97×10^{-02} kg oil eq m^{-3} and 1.91×10^{-02} kg oil eq m^{-3} for the variant distances of 250 km and 175 km, respectively. Similarly, global warming potential was reduced to 8.04×10^{-02} kg CO_2 eq m^{-3} and 7.86×10^{-02} kg CO_2 eq m^{-3}, compared to 8.62×10^{-02} kg CO_2 eq m^{-3} for DWTS. Likewise, PET bottled water supply was found to have higher impacts than the DWTS with the potential of 2.89×10^{-01} kg oil eq m^{-3} and 2.43×10^{-01} kg oil eq m^{-3} for fossil depletion and the global warming potential of $1.12\times10^{+00}$ kg CO_2 eq m^{-3} and 9.84×10^{-01} kg CO_2 eq m^{-3} for 250 km and 175 km, respectively. However, a similar drop in the potential impacts was also observed for the DWTS. The fossil depletion of 2.17×10^{-02}<1.97×10^{-02} <1.91×10^{-02} kg oil eq m^{-3} and the global warming potential of 8.62×10^{-02} <8.04×10^{-02} <7.86×10^{-02} kg CO_2 eq m^{-3} were observed for different distances of 500 km, 250 km, 175 km, respectively (Table 4).

3.4.1.2. Aquatic ecosystem impacts: eutrophication and ecotoxicity. Freshwater and marine eutrophication showed a small reduction with respect to the distance in Scenario 1, as shown in Table 4 and depicted in Fig S3 (Supplementary material). The PET bottled water supply had potential impact of 5.01×10^{-04} kg P eq m^{-3} and 4.90×10^{-04} kg P eq m^{-3} for freshwater eutrophication and 7.56×10^{-04} kg N eq m^{-3} and 7.55×10^{-04} kg N eq m^{-3} for marine eutrophication for the respective distances of 250 km and 175 km, which was higher than the DWTS 2.54×10^{-04} kg P eq m^{-3} and 2.53×10^{-04} kg P eq m^{-3} for freshwater eutrophication and 6.68×10^{-04} kg N eq m^{-3} and 6.67×10^{-04} kg N eq m^{-3} for marine eutrophication, respectively. Furthermore, DWTS showed lower overall aquatic impacts than the PET bottled water supply (Table 4 and Fig S3 (Supplementary material)). These results further support the explanation (Section 3.2.3) that eutrophication and ecotoxicity are mainly caused by the construction of MCW and due to the emissions caused by wastewater discharge. However, in this case, both the major contributors were the same, thus, very slight difference on the outcomes was obtained due to reduction in distance travelled. Additionally, because the amount of PET bottles remained the same, therefore, no major impacts were seen compared to the actual assessed case.

3.4.1.3. Particulate matter and ozone formation. The PM and ozone formation were slightly reduced for DWTS and PET bottled water supply with respect to the distance as mentioned in Table 4 and Fig. S3 (Supplementary material). The potential impacts of PM formation for DWTS were observed as 2.02×10^{-04} kg $PM_{2.5}$ eq m^{-3}, 1.95×10^{-04} kg $PM_{2.5}$ eq m^{-3} and 1.93×10^{-04} kg $PM_{2.5}$ eq m^{-3} for distance travelled of 500 km, 250 km and 175 km respectively, which were still lower than the PET bottled water supply (2.32×10^{-03} kg $PM_{2.5}$ eq m^{-3}, 1.75×10^{-03} kg $PM_{2.5}$ eq m^{-3} and 1.58×10^{-03} kg $PM_{2.5}$ eq m^{-3} respectively). Likewise, the ozone formation for the respective distance of 500 km, 250 km and 175 km for DWTS was within the concentration of 2.28×10^{-04} kg NO_x eq m^{-3}, 2.02×10^{-04} kg NO_x eq m^{-3} and 1.95×10^{-04} kg NO_x eq m^{-3} respectively, again lower than the PET bottled water supply (Table 4). The main contributors for the PM and ozone formation were SO_2 and NO_x emissions (Section 3.2.4), hence the higher reduction in the potential impacts was due to lesser amount of fuel utilized for the transportation.

3.4.1.4. Human toxicity. The DWTS technology had a human toxicity potential of 5.57×10^{-02} kg 1,4-DCB m^{-3} and 5.56×10^{-02} kg 1,4-DCB m^{-3} for carcinogenic, and 2.10×10^{-01} kg 1,4-DCB m^{-3} and 2.09×10^{-01} kg 1,4-DCB m^{-3} for non-carcinogenic toxicity at the travel distances of 250 km and 175 km, respectively, and this was lower than the PET bottled water supply 1.20×10^{-01} kg 1,4-DCB m^{-3} and 1.12×10^{-01} kg 1,4-DCB m^{-3} for carcinogenic, and $1.93\times10^{+00}$ kg 1,4-DCB m^{-3} and $1.86\times10^{+00}$ kg 1,4-DCB m^{-3} for non-carcinogenic toxicity respectively (Table 4 and Fig. S3 (Supplementary material)). The difference for PET bottled water supply was negligible in comparison to the actual distance (500 km) as the major contribution for DWTS was from the system (MCW construction) and wastewater discharge (mainly zinc) and for PET bottled water supply from the PET bottle production and manufacturing (number of bottles unchanged).

In addition, a sensitivity analysis was also performed for MCW and PET bottled water supply considering 5000 km and 0 km (without any distance travelled), respectively as shown in Fig. S4 (Supplementary material). The outcomes showed that despite higher distance travelled by the MCW, it has still much lower potential impacts approximately less than 40% for almost all the indicators as compared to PET bottled water supply at zero distance (Fig. S4), because the number of bottles along with their production had always higher impacts than the MCW system. Meanwhile, to take it even further based on the amount of wastewater treated and discharged, MCW was considered 6 times more than its actual assessed case regarding wastewater treatment and discharge (12,000 m^3 and 36,000 m^3 respectively) against the actual case of PET bottled water supply (2000 m^3 and 8000 m^3 of treated and discharged wastewater respectively) at various distance travelled as shown in Fig. S5 (Supplementary material). Interestingly, the MCW still strongly outperformed the PET bottled water supply for almost all the selected indicators except only aquatic eutrophication and carcinogenic toxicity (Fig. S5). The aquatic eutrophication was increased due to increase in the amount of wastewater discharged into the surface water, whereas carcinogenic toxicity could be increased due to the emissions from construction of MCW (technically, this should not be counted into calculation as the MCW is constructed once) and more energy consumption for pumping water. Anyway, this comparison was with the actual case of PET bottled water and if it would also be increased by 6 times as of MCW, the impacts will be number of times high because by doing so the number of bottles will also be increased and their production could cause higher impacts.

Therefore, it can be concluded that the MCW was found more sustainable than the PET bottled water supply in all the studied variants, and standalone MCW could better perform at 6 scenarios for water treatment and discharge as compared to single PET bottled water supply

Table 5
Sensitivity analysis results for different variants of Scenario 2

Impact category	Unit	DWTS	Conventional (150 m)	Conventional (75 m)
Fine particulate matter formation	kg $PM_{2.5}$ eq	1.19×10^{-04}	1.64×10^{-04}	9.34×10^{-05}
Fossil resource scarcity	kg oil eq	1.73×10^{-02}	3.43×10^{-02}	1.90×10^{-02}
Freshwater ecotoxicity	kg 1.4-DCB	8.73×10^{-03}	5.05×10^{-03}	2.88×10^{-03}
Freshwater eutrophication	kg P eq	2.75×10^{-05}	4.43×10^{-05}	2.79×10^{-05}
Global warming	kg CO_2 eq	5.83×10^{-02}	1.19×10^{-01}	6.63×10^{-02}
Human carcinogenic toxicity	kg 1.4-DCB	1.28×10^{-02}	2.34×10^{-02}	1.27×10^{-02}
Human non-carcinogenic toxicity	kg 1.4-DCB	7.33×10^{-02}	7.13×10^{-02}	4.15×10^{-02}
Ionizing radiation	kBq Co-60 eq	5.42×10^{-03}	9.58×10^{-03}	7.52×10^{-03}
Land use	m^2a crop eq	1.86×10^{-03}	3.22×10^{-03}	1.83×10^{-03}
Marine ecotoxicity	kg 1.4-DCB	1.09×10^{-02}	6.81×10^{-03}	3.87×10^{-03}
Marine eutrophication	kg N eq	2.03×10^{-06}	3.59×10^{-06}	2.24×10^{-06}
Mineral resource scarcity	kg Cu eq	5.37×10^{-04}	1.43×10^{-03}	7.54×10^{-04}
Ozone formation, Human health	kg NO_x eq	1.35×10^{-04}	3.01×10^{-04}	1.64×10^{-04}
Ozone formation, Terrestrial ecosystems	kg NO_x eq	1.42×10^{-04}	3.09×10^{-04}	1.68×10^{-04}
Stratospheric ozone depletion	kg CFC11 eq	7.79×10^{-08}	4.96×10^{-08}	2.82×10^{-08}
Terrestrial acidification	kg SO_2 eq	2.43×10^{-04}	3.66×10^{-04}	2.12×10^{-04}
Terrestrial ecotoxicity	kg 1.4-DCB	1.00×10^{-01}	3.32×10^{-01}	1.75×10^{-01}
Water consumption	m^3	6.58×10^{-04}	5.16×10^{-02}	5.10×10^{-02}

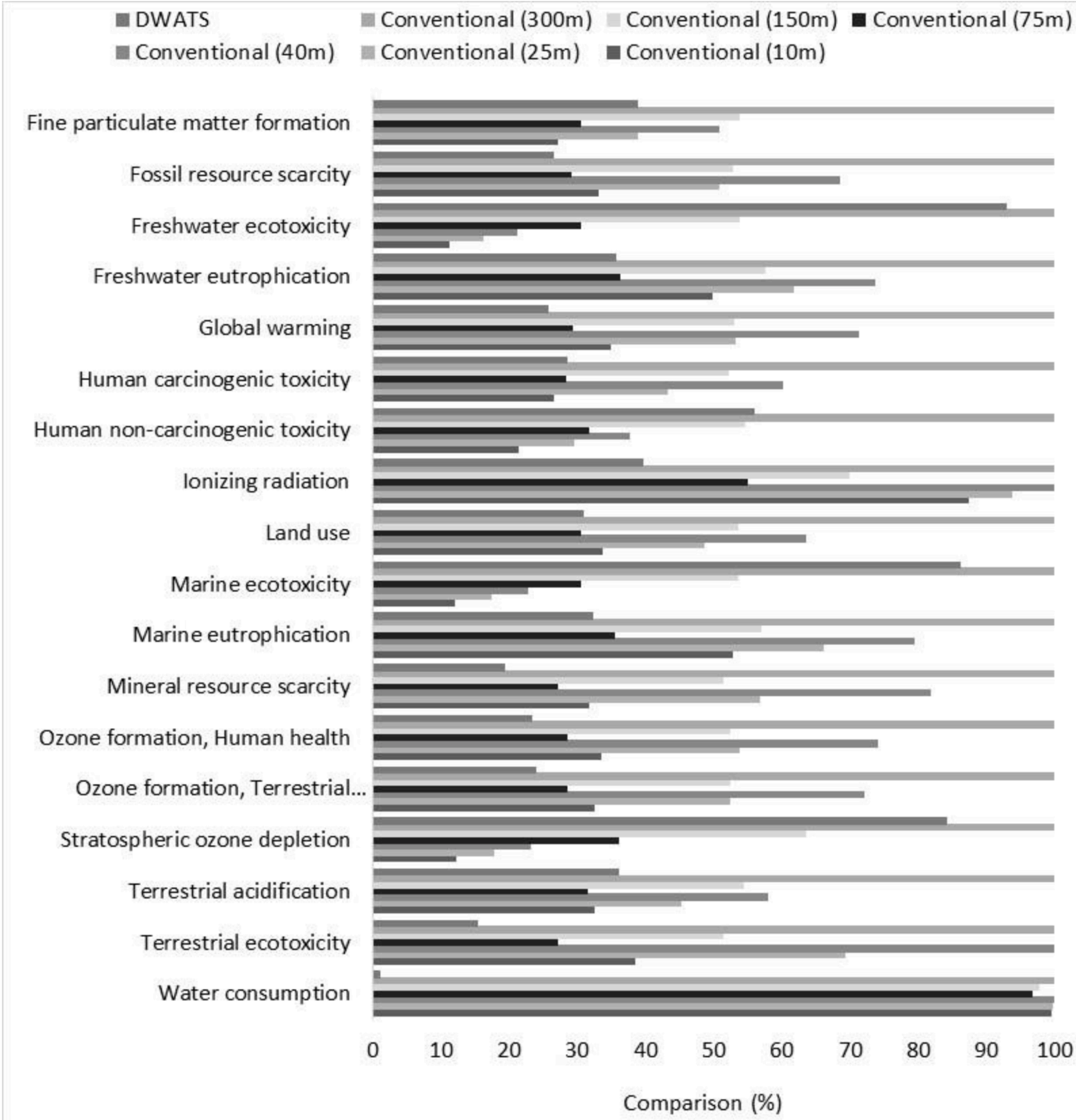

Fig 7. Comparison of potential impacts for all the indicators between different variants of Scenario 2.

at the festival.

3.4.2. Scenario 2: water recycling at the restaurant

In Scenario 2, it was observed that the potential impacts of various indicators were mainly due to fuel burning used in pumping the water to distribution network, production of PVC pipes, construction of concrete pipes and slabs, and also the excavation and filling works. Therefore, to get a clear picture, it was essential to analyse the sensitivity of the compared conventional system by varying the distance. The potential impacts for all the variants (different distances) for Scenario 2 are shown in Table 5 and the maximum result is set to 100% and the results of the other variants are displayed in relation to this result as shown in Fig. 7.

3.4.2.1. Fossil depletion and global warming. The conventional system showed a fossil depletion potential of 3.43×10^{-02} kg oil eq m^{-3} and 1.90×10^{-02} kg oil eq m^{-3} for the respective distances of 150 m and 75 m (Table 5), whereas the fossil depletion potential for the DWTS system remained the same (1.73×10^{-02} kg oil eq m^{-3}; Fig S6 (Supplementary material)), thanks to its stationary installation. Similarly, the conventional system had global warming potential of 1.19×10^{-02} kg CO_2 eq m^{-3} (at 150 m) and 6.63×10^{-02} kg CO_2 eq m^{-3} (at 75 m), which were higher than 5.83×10^{-02} kg CO_2 eq m^{-3} from the DWTS. Even at the lowest studied distance for the conventional water supply and sewer system, the DWTS was still found to be the environmentally most sustainable alternative (Table 5).

3.4.2.2. Aquatic ecosystem impacts: eutrophication and ecotoxicity. The conventional system showed a higher reduction for aquatic eutrophication and ecotoxicity for scenario 2 as depicted in Table 5 and Fig. S6 (Supplementary material), ultimately proving the reduced emissions of P and N because the reduction in fossil fuel burning for the excavation as well as energy consumption for water distribution with respect to distance used closer to the restaurant. In addition, by reducing distance, the excavation and filling works along with the amount of PVC as well as concrete pipes were minimized that greatly contributed to lower potential impacts for 150 and 75 m (Table 5). However, freshwater and marine ecotoxicity impacts were greatly reduced in conventional system at 175 m and 75 m which were lower than the DWTS (Fig. S6 (Supplementary)). This could be due to the variations in the length of PVC and concrete pipes, ultimately lower emissions of heavy metals and acids during their production. But the impacts from DWTS were same due to its stationary installation. Therefore, the standalone DWTS had lower eutrophication impacts and higher ecotoxicity impacts as compared to conventional alternative at 150 m and 75 m.

3.4.2.3. Particulate matter and ozone formation. The conventional system showed a higher reduction for both PM and ozone formation by reducing the distance whereas DWTS had same impacts due to its stationary installation as can be seen in Table 5 and Fig S6 (Supplementary material). The ultimate change in the potential impacts for conventional system was $3.05\times10^{-04} < 1.64\times10^{-04} < 9.34\times10^{-05}$ kg $PM_{2.5}$ eq m^{-3} for PM formation and $5.76\times10^{-04} < 3.01\times10^{-04} < 1.64\times10^{-04}$ kg NO_x eq m^{-3} for ozone formation at the respective distance of 300 m, 150 m and 75 m which were slightly lower than the stationary DWTS for PM formation (1.19×10^{-04} kg $PM_{2.5}$ eq m^{-3}), but still higher for ozone formation (1.35×10^{-04} kg NO_x eq m^{-3}). This also justifies the claim in Section 3.3.4 that the SO_x and NO_x emissions were reduced due to the less amount of fossil fuel burning for the excavation and filling works as well as for PVC pipes production.

3.4.2.4. Human toxicity. The human toxicities were reduced by approximately half of the original impact with concentrations of 2.34×10^{-02} kg 1,4-DCB m^{-3} and 1.27×10^{-02} kg 1,4-DCB m^{-3} for carcinogenic and 7.13×10^{-02} kg.1,4-DCB m^{-3} and 4.15×10^{-02} kg 1,4-DCB m^{-3} for non-carcinogenic toxicities at distances of 150 m and 75 m as shown in Table 5 and Fig S6 (Supplementary material), respectively. The DWTS had higher potential impacts of carcinogenic as compared to conventional 75 m, whereas, it had higher impacts of non-carcinogenic compared to both the conventional variants of 150 m and 75 m. Since the reduction in distance caused huge impact on the outcomes, because of less solid/gaseous fuel burning (leading to less emissions) for excavation and filling works and the major role in minimizing the impacts was due to reduction in the length of the PVC pipes due to shorter distances.

Therefore, the reduction in the distance may cause lesser toxicity, but the DWTS still outperformed the conventional water supply and sewerage disposal systems at the studied distances for almost all other potential indicators. However, to take it even further, a comparative LCIA was also performed to determine the minimum distance at which the conventional systems may outperform the DWTS. In this regard, the distances of 10 m, 25 m and 40 m for conventional water supply and wastewater collection systems would result in better performance compared to the DWTS. Specifically, a distance of 75 m for the conventional systems would make both alternatives equal in almost all the selected impact categories (Fig. 6).

4. Conclusions

The aim of this study was to analyse and compare the sustainability of DWTS technologies with conventional alternatives for wastewater treatment and production of potable water. The results showed that the fossil depletion, global warming, eutrophication and ecotoxicity potentials were significantly lower for the DWTS than their conventional alternatives. The sensitivity analysis revealed that the potential environmental impacts are strongly dependant on the distance travelled in Scenario 1 and from the construction of conventional water supply and sewerage system in Scenario 2. Overall, the studied DWTS technologies are both environmentally sustainable and viable option for the treatment of wastewater and production of potable water at the festivals and the standalone restaurants without municipal water supply and sewerage collection systems.

CRediT authorship contribution statement

Fida Hussain Lakho: Conceptualization, Investigation, Methodology, Writing – original draft. **Asif Qureshi:** Data curation, Writing – review & editing. **Wouter Igodt:** Investigation, Methodology. **Hong Quan Le:** Investigation. **Veerle Depuydt:** Project administration, Funding acquisition. **Diederik P.L. Rousseau:** Conceptualization, Methodology, Writing – review & editing. **Stijn W.H. Van Hulle:** Project administration, Conceptualization, Supervision, Methodology, Writing – review & editing, Funding acquisition.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Supplementary materials

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