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Micropollutant abatement with UV/H_2O_2 oxidation or low-pressure reverse osmosis? A comparative life cycle assessment for drinking water production

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ABSTRACT

Micropollutants (MP) are undesired in drinking water. Advanced oxidation processes (AOP) or low-pressure reverse osmosis membrane filtrations (LPRO) can be used to remove them during the water purification process. For a specific case, two treatment scenarios were compared with a life cycle assessment (LCA), using three impact assessment methods (Ecological Scarcity 2013, ILCD 2011, EDIP 2003). Scenario 1 (AOP-based) was a UV/H_2O_2 oxidation with a subsequent granular activated carbon (GAC) filter to remove excess H_2O_2 before soil infiltration. Scenario 2 (LPRO-based) was a side-stream treatment with an ultrafiltration (UF) and low-pressure reverse osmosis (LPRO) filtration before soil infiltration and the LPRO retentate was treated with O_3/H_2O_2 and subsequent granular activated carbon (GAC) filter before discharge back into Rhine. Sensitivity analyses were performed on the relevant contributors to evaluate the robustness of the results. LCA results showed that in the base-line scenario (electricity from renewable energy sources) the LPRO-based treatment had notably fewer environmental impacts than the AOP-based treatment, which was confirmed with three impact assessment methods. Key contributors to the impacts were mostly operating resources, i.e., electricity, H_2O_2, liquid O_2 for ozone generation and GAC, but also construction resources in the LPRO process. The electrical energy source was decisive for the results: with a share of renewable energy sources <80%, the AOP-based treatment was the better option due to its lower specific energy demand. The optimization of treatment conditions, such as lower H_2O_2 concentration at an increased UV fluence; different H_2O_2:O_3 molar ratios; or extended GAC utilization time could influence the environmental impact within a range of ±10–30%. Environmental benefits, i.e. the reduction of potential hazardous effects of 21 MPs, were determined with EDIP 2003 and USEtox for both treatment scenarios. The estimated benefits were negligible in comparison to the environmental burden caused by the treatments, thus would not be justified from a global LCA impact-benefit perspective. However, because of several uncertainties and lack of data, the inclusion of treatment benefits in LCAs for drinking water purification requires further research.

1. Introduction

Surface waters are important resources for drinking water production, but are increasingly polluted by micropollutants (MPs), such as industrial chemicals residues, food additives, pharmaceuticals, or pesticides (Schwarzenbach et al., 2006; Benotti et al., 2009). Effects of MP residuals on ecosystems and human health are not yet fully understood (Alharbi et al., 2018; Wee and Aris, 2017; Guo et al., 2019). To avoid long-term impacts, they must be prevented from entering the drinking water system. For many MPs, simple drinking water treatment processes such as coagulation/flocculation, sedimentation or sand filtration only have limited elimination effects on MPs (Mompelat et al., 2009; Simazaki et al., 2015; Kiefer et al., 2020; Couto et al., 2019). The adsorption of MPs in a granulated activated carbon (GAC) filter can also be insufficient depending on the MP (Stackelberg et al., 2007; Huerta-Fontela et al., 2011; Kiefer et al., 2020). Additional treatment approaches can be applied, such as partial oxidation in ozonation (Camel and Bermond, 1998; Westerhoff et al., 2005), advanced oxidation processes (AOP, Camel and Bermond, 1998; Von Gunten, 2018; Wünsch et al., 2021) or

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Abbreviations

ADI	Acceptable daily intake
AOP	Advanced oxidation process
BAC	Biological activated carbon
BV	Bed volumes
CF	Characterization factor
DWS	Drinking water supplier
EBCT	Empty bed contact time
EQS	Environmental quality standard
DOC	Dissolved organic carbon
GAC	Granular activated carbon
LCA	Life cycle assessment
LPRO	Low-pressure reverse osmosis
MAR	Managed aquifer recharge
MP	Micropollutant
NF	Nanofiltration
UF	Ultrafiltration
UV	Ultraviolet

separation from water by dense membranes (Nghiem and Schäfer, 2004; Plakas and Karabelas, 2012; Taheran et al., 2016). However, the use of these technologies also has impacts to the environment over their complete life cycle, e.g., by their production, installation, operation, and end-of-life disposal (Mohapatra et al., 2002; Bonton et al., 2012; Garfi et al., 2016). This should be taken into consideration and should be justified by the positive effects achieved. To analyze environmental impacts of technologies, the life cycle assessment (LCA) is today the most comprehensive method and guide for decision-makers. An LCA takes into account a variety of environmental impacts over the whole life cycle of a process, such as raw material extraction, transport, commissioning, operation, de-commissioning, waste processing and/or recycling of materials. The framework of the LCA methodology is covered in International Organization for Standardization (ISO) 14040 and 14044 (ISO, 2006a; 2006b).

An existing drinking water treatment plant in Basel (Switzerland) consists of a rapid sand filter, a managed aquifer recharge (MAR) followed by a granulated activated carbon filter (GAC) and disinfection step with UV radiation. Not all MPs are fully attenuated after MAR and GAC filtration. Hence, a multi-barrier approach with an additional treatment to target MPs appears promising (Grünheid et al., 2005; Maeng et al., 2011; Lekkerkerker-Teunissen et al., 2012). In order to sustain adequate treatment results even with increasing and fluctuating levels of MPs, lower threshold concentrations or concerns about highly mobile MPs, additional treatment approaches are being investigated by the drinking water supplier (DWS). Potential synergistic effects from the additional treatment and MAR would be desirable. This study compares the environmental performance of two considered treatment scenarios with a LCA. The investigated treatment scenarios are described in more detail below.

LCAs of drinking water treatment approaches have been previously applied in a few studies with similar treatment approaches such as GAC, nanofiltration (NF) and reverse osmosis (RO) but no AOPs have been assessed so far (Mohapatra et al., 2002; Barrios et al., 2006; Bonton et al., 2012; Manda et al., 2014; Ribera et al., 2014; Garfi et al., 2016). An overview of LCAs on MP abatement in drinking water and wastewater treatments is presented in the supporting information (SI, Table S1). Environmental burdens were mainly generated during operation of the technologies, but construction can also have impacts. Key parameters that influenced the impacts were electricity (Mohapatra et al., 2002; Garfi et al., 2016), GAC (Barrios et al., 2006; Bonton et al., 2012), construction (Bonton et al., 2012; Ribera et al., 2014), or process auxiliaries, such as cleaning chemicals (Barrios et al., 2006; Bonton et al., 2012). The electrical energy source was shown to be another key parameter and environmental impacts decreased notably with increasing fractions of renewable energy sources (Bonton et al., 2012; Manda et al., 2014). Sensitivity analyses of treatment parameters have only been included in a few studies (Manda et al., 2014; Garfi et al., 2016). Moreover, mostly only one impact assessment method was generally used, even though methods can show different results (Renou et al., 2008). Therefore, three different impact assessment methods were used to evaluate the robustness and validity of the results: Ecological Scarcity 2013, EDIP 2003 and ILCD 2011. Ecological Scarcity 2013 and ILCD 2011 have not previous been used in the reviewed literature. In contrast to the ISO 14040/14044 (ISO, 2006a; 2006b), the results were weighted and fully aggregated to a single score.

Some LCA studies for wastewater treatment included benefits from MP abatement, resulting in reduction of the water's toxicity impacts. In general, the benefits of MP abatement determined by the LCA methods were negligible compared to the environmental burdens caused by the technologies applied (Köhler et al., 2012; Igos et al., 2012, 2020; Zepon Tarpani and Azapagic, 2018; Arzate et al., 2019). It was shown that an increasing number of MPs considered in the LCA resulted in an increase of environmental benefits upon MP abatement, because the sum of reduced toxicities from abated MPs become more significant (Türk et al., 2013; Li et al., 2019). However, in the studies reviewed, these approaches have not yet been applied for drinking water treatment.

Overall, this research aims to answer (i) which of the investigated treatment scenarios against MP has a lower environmental impact with comparable performance in the specific case study, and (ii) how robust the results are according to the applied LCA methods and a sensitivity analysis. It may thus allow us to draw generalizable conclusions. Furthermore, (iii) this study investigates whether the additional environmental burdens caused by deployment of an additional treatment can be justified by environmental benefits from lower toxicities as a result of the abatement of MPs.

2. Method

A framework of the LCA study presented is shown in Fig. 1. A description of each step is given in the following sections.

2.1. Description of treatment scenarios

The selected treatment scenarios are shown in Fig. 2 and are described in the following. The additional treatments would take place after the rapid sand filtration and before the MAR as the DWS would like to protect the soil and aquifer from potential MP accumulation.

Scenario 1 based on full-stream treatment with UV/H_2O_2, consisting of a hydrogen peroxide (H_2O_2) dosage and subsequent low-pressure mercury-vapor lamp radiation (radiating almost mono-chromatically at 254 nm). UV/H_2O_2 AOPs are known to partially oxidise MPs based on photolysis and hydroxyl radicals generated *in situ* by H_2O_2 photolysis (Glaze et al., 1987; Legrini et al., 1993; Stefan, 2018). To quench residual H_2O_2 before MAR, a granular activated carbon (GAC) filter is considered (Huang et al., 2018). It was important for the DWS not to deploy ozone-based treatment for drinking water in order to eliminate the risk of bromate formation. This is because bromide concentrations in the source water (river Rhine in north-western Switzerland) can peak above 100 μg/L, and if ozone is applied without bromate mitigation strategies, such concentrations can result in the formation of bromate concentrations above 10 μg/L (the threshold concentration for drinking water in Switzerland, recommendation by the world health organization, WHO) (Von Gunten, 2003).

Scenario 2 based on side-stream treatment with a low-pressure reverse osmosis (LPRO) membrane. Dense membranes can produce a retentate with a significantly higher MPs concentration than before filtration (Van der Bruggen et al., 2008; Solley et al., 2010). It is assumed that, after adequate treatment, this retentate can be discharged to the

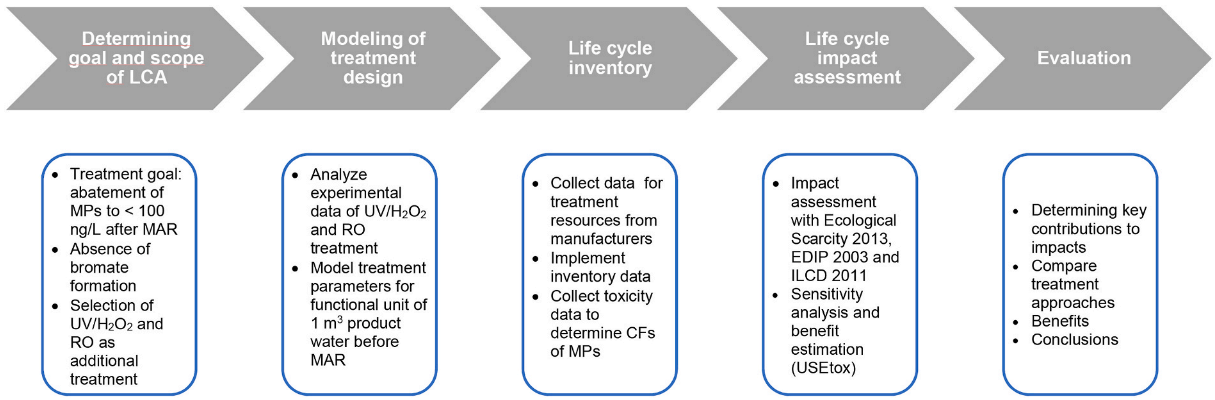

Fig. 1. Framework of the presented LCA case study.

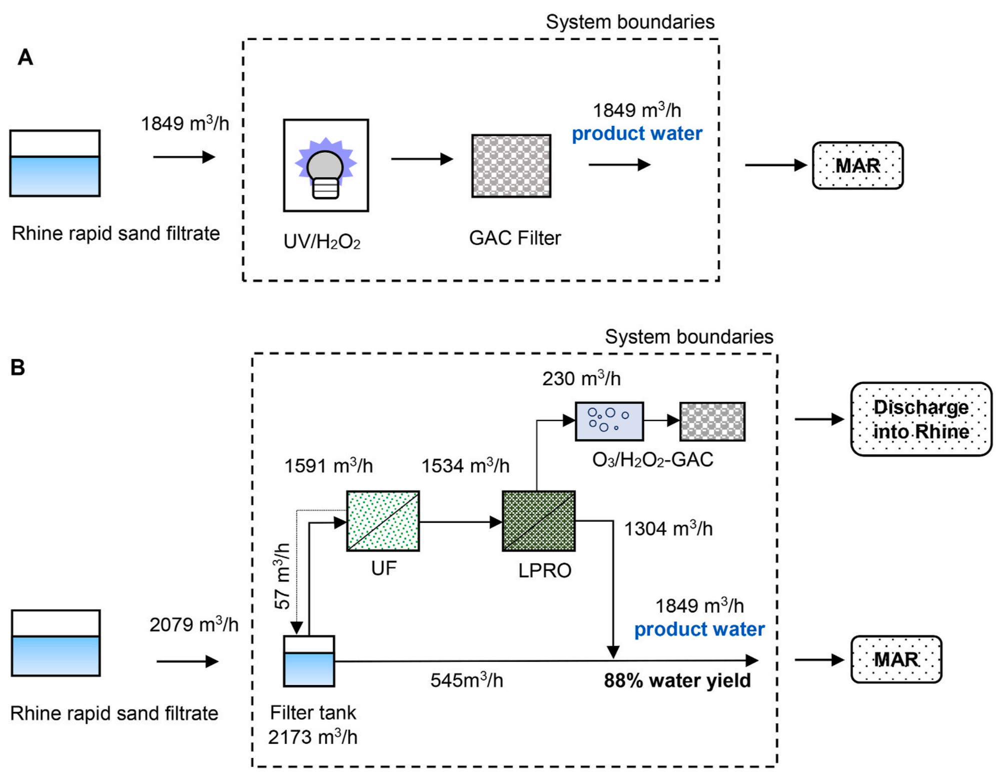

Fig. 2. A: Treatment design and system boundary of scenario 1 (AOP based). **B**: Treatment design and system boundary of scenario 2 (LPRO based). UV/H_2O_2 = Ultraviolet radiation with hydrogen peroxide, GAC = Granulated activated carbon, UF = Ultrafiltration, LPRO = Low-pressure reverse osmosis, O_3/H_2O_2 = Ozonation with hydrogen peroxide, MAR = Managed aquifer recharge.

river Rhine. It must therefore be properly managed under Swiss law, which was adapted in 2014 with regard to the issue of MPs in the river Rhine and came into effect in 2016 (Eggen et al., 2014; WPO, 2021). A promising treatment to abate MPs in a retentate is the combination of an AOP process with a subsequent GAC filter (Justo et al., 2015; Cai et al., 2020). For the retentate produced by the LPRO process, ozonation appears to be an adequate treatment (Van der Bruggen et al., 2008; Pérez-Gonzáles et al., 2012; Deng, 2020). Upon ozonation of bromide-containing waters, bromate can be formed (Von Gunten, 2003). The recommended threshold value for bromate concentration in surface waters is 50 μg/L (Swiss Centre for Applied Ecotoxicology Eawag-EPFL; Soltermann et al., 2017). The addition of H_2O_2 before ozone dosage – a known strategy for bromate mitigation (Katsoyiannis et al., 2011; Von Sonntag and von Gunten, 2012) – was considered to limit bromate formation during concentrate ozonation. The ozonated retentate is further treated with a granular activated carbon (GAC) filter to adsorb possible toxic by-products (Çeçen and Aktaş, 2011; Hübner et al., 2015; Von Gunten, 2018) and improve MP abatement (Knopp et al., 2016; Müller et al., 2019; Cai et al., 2020) before discharge into the river.

2.2. Goal and scope

The goal of this study was to compare the environmental impacts of two treatment approaches as additional barrier for MPs abatement in the drinking water production of a specific drinking water supplier (DWS) located in Basel (Switzerland). The life cycle assessment was carried out based on the ISO 14040 and 14044 (ISO, 2006a; 2006b). The functional unit is defined as the production of 1 m^3 water before MAR (product water). In the system boundaries, only processes required for the respective treatment scenario were included, i.e., raw materials, principal infrastructure, chemicals, energy consumption, building, transportation, and end of life treatment. Processes that were similar for both treatment scenarios such as installations, e.g., pipes for distribution were not included. The operational size of the treatment scenarios was selected based on the existing DWTP treatment capacity. The annual water volume flow infiltrated at the MAR site was 16.2 Mio m^3 per year (average over the last 5 years, internal data by DWS). Hence, the average product water flow of both treatments was set to 1849 m^3/h.

The treatment systems of the scenarios were designed to achieve the same defined treatment target, i.e. all MPs abated to a maximum concentration of 100 ng/L after MAR (corresponding to the limit value for pesticides in Switzerland, WPO, 2021). Previous studies provided the data required for the system designs (Prahtel, 2019; Wünsch et al., 2019, 2020). For the system designs, MPs were considered of which the 80th percentile concentration exceeded the defined treatment goal (see Table S2). This included benzotriazole, metformin, guanyl urea and acesulfame. EDTA was also observed in concentrations > 100 ng/L, but it was excluded for the design due to its limit value in drinking water of 0.2 mg/L (TBDV, 2021). The MP abatement during MAR was assumed as the 20th percentile of its empirical abatement efficiency in previous column experiments (Wünsch et al., 2019, Text S1, Table S3, Fig. S1). Hence, the required abatement efficacy of the investigated treatment designs was estimated conservatively.

The MP that was decisive for the respective system design differed for the investigated scenarios. For scenario 1 (UV/H_2O_2) it was metformin, while it was acesulfame for scenario 2 (LPRO). Table 1 summarizes the treatment design parameters used for both treatment scenarios. More details about the system design and treatment parameters are in the SI (scenario 1: Text S2, Table S4, scenario 2: Text S3, Tables S6–8).

2.3. Life cycle inventory (LCI)

According to the goal and scope definition, the manufacturers of the treatment technologies investigated established the treatment designs and provided the inventory data (Tables S4 and S6-8). Ecoinvent (v. 3.3), a large international database, was used for the background data. Where necessary, the manufacturers' data was adapted according to the determined treatment design. For both treatment scenarios, the input electricity mix was modelled on that of the local energy provider (Industrielle Werke Basel, 2019) which provided electrical energy from 100% renewable sources (94% hydro-based, Table S9). The inventories for the production of 1 m^3 product water of both scenarios are presented in Tables 2 and 3. Pieces are abbreviated to "p" and "tkm" means tonkilometers. For more details on calculation approaches and ecoinvent processes used, see Text S4 and Tables S10–S15.

2.4. Life cycle impact assessment (LCIA)

The environmental impacts of the scenarios were assessed using the three impact assessment methods: Ecological Scarcity 2013 (Frischknecht and Büsser, Knöpfel, 2013), EDIP 2003 (Hausschild and Wenzel, 1998; Hauschild and Potting, 2005) and ILCD 2011 (European Commission, 2011). Ecological scarcity was selected as the most suitable method as it represents the political environmental targets of

Table 1
System parameters of investigated treatment approaches, i.e., scenario 1 (UV/H_2O_2 + GAC) and scenario 2 (UF-LPRO + O_3/H_2O_2-GAC).

Parameters	Scenario 1	Scenario 2
Share of treated rapid sand filtrate	100%	76.5%
UV fluence at 254 nm [J/m^2]	6700	–
H_2O_2 dose [mg/L]	6.5	–
GAC type	New	Regenerated
GAC bed volumes [BV]	80'000	80'000
Empty bed contact time (EBCT) [min]	6.6	15
Specific ozone dose [mg O_3/mg DOC]	–	0.6
Specific H_2O_2 dose ratio [mol H_2O_2/mol O_3]	–	4 : 1

Table 2
Inventory data of scenario 1 (UV/H_2O_2 + GAC) to produce 1 m^3 of product water.

Component	Lifespan	Unit	Value for functional unit	Data source
UV lamps	14'000 h	p/m^3	2.32E-05	Xylem
UV reactors	20 y	p/m^3	6.17E-09	Estimate
Pumps	20 y	p/m^3	1.23E-07	Xylem
Hydrogen peroxide (50%)		g/m^3	1.30E+01	Xylem
Steel waste treatment		kg/m^3	1.28E-07	Xylem
Glass treatment		kg/m^3	6.40E-08	Xylem
Underground deposit		kg/m^3	1.09E-06	Xylem
Electricity		kWh/m^3	8.10E-02	Xylem
GAC, new	80'000 BV	kg/m^3	6.25E-03	Estimate
Transport road		tkm/m^3	1.50E-05	Estimate
Building	50 y	m^2/m^3	6.54E-07	Estimate

Switzerland where the case study site was located. In addition, to determine the environmental benefits of the abatements of micropollutants, methods with (freshwater) toxicity impact categories were required. The Ecological Scarcity 2013 method provides a toxicity category for pesticides, but no other trace organic chemicals. ILCD 2011 is a current method for LCA with a Europe-wide application and contains toxicity impact categories. New characterization factors were calculated with USEtox, as described in section 3.4.1. A similar approach was taken with the EDIP methodology, but the calculation of characterization factors is based on different calculation approaches and input data. Since the calculation of characterization factors for toxicity categories differs between the ILCD and EDIP methods, the respective results facilitate a comparison of the two methods.

According to the ISO 14044 methodology the assessment of the LCA inventory was carried out with characterization, normalization, and weighting steps (ISO, 2006a; 2006b). Long-term emissions were excluded from the assessment. Regarding weighting, this study is not in line with the requirements of the ISO standard, where weighting for comparisons disclosed to the public is not allowed, but rather it deliberately goes beyond them. The results of all impact categories were weighted and fully aggregated to a single score. For effective decision-making, fully aggregated methods that are also based on political goals are widely accepted, useful, and can be essential in complex situations (Kägi et al., 2016). Actual data for normalization and weighting factors for EDIP 2003 and ILCD 2011 were applied (Hischier et al. (2010), Benini et al. (2014), Laurent et al. (2011), Sala et al. (2018)). In the Ecological Scarcity 2013 method, the factors are already included in the assessment and are not an optional step (Frischknecht and Büsser, Knöpfel, 2013). For the environmental benefits, USEtox v. 2.0 (Fantke, 2015, 2018) was used additionally. USEtox is the recommended approach in the ILCD 2011 methodology for toxicity impacts and can be aggregated to the single score. With EDIP 2003, the following impact categories were used: Ecotoxicity water acute, ecotoxicity water chronic, ecotoxicity soil chronic and human toxicity water. With ILCD 2011 the following impact categories were used: Human cancer toxicity, human non-cancer toxicity, freshwater aquatic ecotoxicity. With the previous determined CFs and the reduced numbers of MPs, the environmental impact avoided was then calculated, normalized, and weighted.

2.5. Sensitivity analysis

To determine the sensitivity of the key parameters and their influence on the LCA results, analyses were carried out. Only the Ecological Scarcity 2013 method was used, which has most appropriate for the specific study site. In the reviewed literature on LCA of drinking water

Table 3
Inventory data for scenario 2 (UF-LPRO + O_3/H_2O_2-GAC) to produce 1 m^3 product water.

	Component	Lifespan	Unit	Value for functional unit	Data source
Ultrafiltration (UF)	UF modules	10 y	p/m^3	1.5E-06	Toray
	Pumps	20 y	p/m^3	3.4E-08	Toray
	Steel	20 y	kg/m^3	4.6E-05	Toray
	Cleaning solutions	–	kg/m^3	6.2E-08	Toray
	Electricity (product water)	–	kWh/m^3	6.7E-02	Toray
	Building	50 y	m^2/m^3	4.4E-07	Estimate
	Waste plastic, mixture, incineration	–	kg/m^3	4.0E-05	Toray
	Waste bulk iron	–	kg/m^3	4.6E-05	Toray
	Transport road	–	km/m^3	1.6E-04	Estimate
	Transport sea	–	km/m^3	1.2E-03	Estimate
Low-pressure reverse osmosis (LPRO)	RO modules	5 y	p/m^3	6.8E-04	Toray
	Pumps	20 y	p/m^3	4.3E-08	Toray
	Steel	20 y	kg/m^3	4.6E-05	Toray
	Acidic cleaning solution (hydrochlorid acid)	–	l/m^3	3.5E-04	Toray
	Alkaline cleaning solution (sodium hydroxide)	–	l/m^3	1.4E-03	Toray
	Electricity	–	kWh/m^3	3.0E-01	Toray
	Building	50 y	m^2/m^3	8.9E-07	Toray
	Waste plastic, mixture, incineration	–	kg/m^3	2.2E-04	Toray
	Waste bulk iron	–	kg/m^3	4.6E-05	Toray
	Transport road	–	km/m^3	8.8E-04	Toray
	Transport sea	–	km/m^3	6.6E-03	Toray
O_3/H_2O_2-GAC	Hydrogen peroxide (50%) solution	–	g/m^3	4.0E+00	Xylem
	Ozone reactor	20 y	p/m^3	3.1E-09	Estimate
	Ozone electricity	–	kWh/m^3	6.6E-03	Xylem
	GAC, regenerated	80’000 BV	kg/m^3	7.8E-04	Estimate
	Liquid oxygen (LOX)	–	kg/m^3	7.0E-03	Xylem
	Cooling water	–	m^3/m^3	9.6E-04	Xylem
	Building	50 y	m^2/m^3	1.2E-07	Estimate
	Additional electricity		kWh/m^3	3.40E-03	Estimate

treatment, sensitivity analyses were also carried out, e.g., with electrical energy consumption in ozonation (Manda et al., 2014; Garfí et al., 2016) or lifetime of membranes (Manda et al., 2014). In wastewater treatment, analyses were also performed for activated carbon regeneration (Li et al., 2019; Igos et al., 2020) or the source of the electrical energy (Türk et al., 2013; Igos et al., 2020). Moreover, Köhler et al. (2012) did a scenario analysis with different H_2O_2 doses and Igos et al. (2012) with LOX consumption. However, this has not yet been done for treatment parameters such as bed volumes for GAC or different DOC-specific O_3 doses in combination with different H_2O_2 doses, regardless of the type of water. Thus, for the following input parameters sensitivity analyses were conducted: electrical energy source, utilization time of GAC (bed volumes), UV fluence, O_3-specific H_2O_2 dose and DOC-specific O_3 dose. It must be pointed out that in this analysis the focus is on the environmental impacts, i.e., to identify the breakthrough points and whether the treatment design could be improved from an LCA perspective. However, experimental data with the treatment parameters used is not available and the actual MP abatement and optimal treatment conditions would have to be proven with data as the chosen values were more hypothetical. Also what has not been considered here is that a different setting of the retentate treatment (O_3/H_2O_2 and GAC) would also potentially change the parameters of the other process (e.g. other bed volumes required in the GAC filter) in order to have a similar treatment goal. The analyses of the retentate treatment were carried out based on the assumption that the other process parameters do not change. A comparison of how the parameters interact was not done.

The electrical energy sources and treatment parameters used for the sensitivity analysis are described in Text S5 and Tables S16–17.

2.5.1. *Environmental benefits of micropollutant abatement*

The avoided environmental impacts as a result of the MP abatement have been calculated based on the acute and chronic toxicity effects of MPs on ecosystems and humans. The abatement refers to the product water after the scenarios but before MAR. The abatement of MPs in the O_3/H_2O_2-GAC process, i.e., the retentate treatment in scenario 2, was not included due to lack of experimental data. In total, 57 MPs in the water samples were measured (Table S2). In other LCAs that included benefits from MP abatement, diclofenac was often found to be the MP that had a significant impact (Li et al., 2019; Igos et al., 2020). Diclofenac’s 80^{th} percentile concentration in the river rapid sand filtrate water was 32 ng/L (Table S2, SI). Therefore, the concentration of 30 ng/L was selected as the next lower full tens concentration as a cut-off concentration what included 21 MPs for the benefit estimation. Of the excluded MPs is the herbicide diuron potentially the most hazardous one with a threshold concentration of 70 ng/L compared to diclofenac with 50 ng/L (WPO, 2021). However, the measured concentrations of diuron were very low with 1–2 ng/L. The 80^{th} percentile concentration was selected to account for varying concentrations of MPs in the water source without considering potential outlier concentrations (maximum concentrations). We argue that the cut-off value likely does not lead to an underprediction of benefits because potential benefits of less concentrated MPs are likely negligible as confirmed in section 3.2.5. This is supported by the fact that metolachlor-ESA and metolachlor-OXA – two pesticide metabolites relevant for drinking water producers – were considered despite their 80^{th} percentile concentrations were <30 ng/L.

For the mass of the reduced MP, the respective median concentrations were used. Table S5 provides the MPs’ relative abatements of each scenario. To determine the avoided environmental impact in the LCA, characterization factors (CFs) are required. As there were very few CFs for the included MPs available in the methods, new CFs were calculated according to the EDIP 2003 and USEtox methodology (Hausschild and Wenzel, 1998; Fantke, 2015, 2018; Hauschild and Potting, 2005). Data for eco- and human toxicity were taken in the following order: Environmental quality standards (EQS, proposed by the Swiss Centre for Applied Ecotoxicology Eawag-EPFL in Switzerland) or acceptable daily intake (ADI), experimental data from literature and if no data were available, a quantitative structure-activity relationship (QSAR) based tool to predict effect concentrations such as ECOSAR and QSAR Toolbox was used. To predict missing data for chemical and physical properties of the included MPs, the “Estimation Programs Interface SuiteTM” (EPI Suite) was used. Text S6, Tables S20–22 (chemical properties) and Tables S23–24 (toxicity data) provide more details about the approaches and data used.

3. Results and discussion

3.1. Environmental impact of the base-case scenarios

The total environmental impacts of the base-case scenarios according to the impact assessment methods are shown in Table 4. Table 5 shows the percentage shares of the sub-systems, i.e. UV/H_2O_2 and GAC for scenario 1, and UF, LPRO, O_3/H_2O_2 and GAC for scenario 2, relating to the overall impacts on the scenario. Fig. 3 shows the contributions of the individual processes (construction, H_2O_2, GAC, LOX, cleaning chemicals and electricity) in scenario 1 and scenario 2 to the total environmental impact categories. The results for each scenario are described below and the treatments are compared.

The impacts in scenario 1 (AOP based) are mainly due to the operation (89–97%) according to all impact assessment methods. GAC had more impact than the UV/H_2O_2 process in the treatment scenario, within a range of 54–70%. H_2O_2 alone counted for 27–39% of the environmental impacts. The impacts of the constructions (reactor, UV lamp and building) were generally low (3–11%) depending on the method, as were those of electricity with a negligible contribution of 3–4%. The main impact categories of the total scenario with shares over 10% were: global warming and main air pollutants for Ecological Scarcity 2013; acidification, radioactive waste and human toxicity water for EDIP 2003; mineral, fossil & renewable resource depletion, climate change, human toxicity cancer effects, particulate matter and fossil resource depletion for ILCD 2011.

The impacts in scenario 2 (LPRO-based) are distributed over 24–45% for constructions and 55–76% for the operation, depending on the method. With Ecological Scarcity 2013 and EDIP 2003, the LPRO and O_3/H_2O_2 sub-systems were the main contributors to the total impacts, with similar shares of 40–44% each, followed by UF (10–12%) and GAC (5–6%). Electricity had the highest contribution with 27–33%, followed by construction with 24% (of which are 60% due to membrane resources), LOX (18–25%) and H_2O_2 (17–19%) according to Ecological Scarcity 2013 and EDIP 2003. With ILCD 2011, the share of the LPRO process was 48% followed by O_3/H_2O_2-GAC (32%) and UF (20%). Construction counted for 45% in this treatment scenario, mainly due to building resources (60%) followed by membrane resources (24%). Electricity counted for 26% of the total impacts, followed by H_2O_2 (17%) and LOX (7%). GAC only had 3% contribution. Cleaning chemicals showed negligible impacts in all assessment methods. The main impact categories of the total scenario with shares over 10% were: global warming and main air pollutants for Ecological Scarcity 2013; radioactive waste, ozone depletion and acidification for EDIP 2003; mineral, fossil & renewable resource depletion, human toxicity cancer effects and climate change for ILCD 2011.

Within the impact assessment methods applied, differences were observed for the share of each key factor and the impact categories. While Ecological Scarcity 2013 and EDIP 2003 methods showed similar contributions of the processes, slight differences were observed with the ILCD 2011 method, especially in scenario 2, with higher shares of constructions (due to building resources) and H_2O_2 of the impacts that resulted in lower contributions of the other parameters (i.e., GAC in scenario 1 and electricity, LOX and GAC in scenario 2). These differences are explained by different calculation and weighting approaches used by the impact assessment methods and this influences the results. Nevertheless, the assessed methods showed comparable results and pointed out the same key contributors to the environmental impacts. This confirmed the validity as well as robustness of the results and the fundamental statement was independent of the LCA method applied. In addition, the results could also be applicable for other regions since EDIP 2003 and ILCD are European methods.

Table 5
Percentage share of treatment sub-systems of total environmental impacts in scenario 1 (AOP treatment) and scenario 2 (LPRO treatment).

Method	Share of sub-systems					
	Scenario 1		Scenario 2			
	UV/ H_2O_2	GAC	UF	LPRO	O_3/ H_2O_2	GAC
Ecological Scarcity 2013	34%	66%	12%	43%	40%	5%
EDIP 2003	36%	64%	10%	40%	44%	6%
ILCD 2011	46%	54%	20%	48%	29%	3%

Even though other impact assessment methods were used as in the reviewed literature (section 1), similar results were obtained in this study. In LCA's for wastewater treatment it was seen that also H_2O_2 contributed to the impacts in AOP treatments (Köhler et al., 2012; Arzate et al., 2019) and ozonation in general had an impact due to its electrical energy consumption (Zepon Tarpani and Azapagic, 2018). In contrast to this study, the electrical energy for filtration processes had in general higher contributions to the impacts in literature (80% contribution in Garfi et al., 2016; 50% in Mohapatra et al., 2002). However, Bonton et al. (2012) showed also that impacts could vary significantly if electrical energy from renewable sources would be used for the operation. This explains the higher contribution of retentate treatment in scenario 2, and showed that the high electrical energy demand of the LPRO was not decisive for the impact in the baseline scenario. Cleaning solutions had no impact in this study, whereas in literature this has been observed (Zhou et al., 2011; Tarnacki et al., 2012). It should be mentioned again that the antiscalant solution was not included in this LCA due to lack of data in the utilzed database. It was shown that antiscalants can make a notable contribution to the impacts because of the phosphoric contents used in its production (Zhou et al., 2011). However, another chemical used as an antiscalant (polycarboxylates) can show impacts in photochemical oxidation but was negligible in comparison to the total impacts of a desalination process (Tarnacki et al., 2012). The antiscalant considered for scenario 2 was a poly-acrylate based, phosphor free and environmentally friendly antiscalant (Ropur, 2019). Therefore, the impacts were considered to be insignificant, but this assumption needs further verification.

When comparing the scenarios, it is clearly seen that in the base-case scenario 2 would be the preferable option from a LCA perspective with significantly less environmental impacts per 1 m^3 product water, regardless of the impact assessment method used. The differences ranged from +50 to 120% higher environmental impacts of scenario 1 compared to scenario 2, depending on the impact assessment method. In both scenarios, the main cause of environmental impacts were the

Table 4
Environmental impacts of scenario 1 (UV/H_2O_2 + GAC) and scenario 2 (UF-LPRO + O_3/H_2O_2-GAC) according to Ecological Scarcity 2013, EDIP 2003 and ILCD 2011 with electrical energy from 100% renewable sources (base case).

Method	Unit	Total environmental impact		Share of construction (C) and operation (O)			
		Scenario 1	Scenario 2	Scenario 1		Scenario 2	
		AOP	LPRO	C	O	C	O
Ecological Scarcity 2013	Ecopoints	59	27	3%	97%	24%	76%
EDIP 2003	10^{-6} Points	96	47	5%	95%	24%	76%
ILCD 2011	10^{-6} Points	9	6	11%	89%	45%	55%

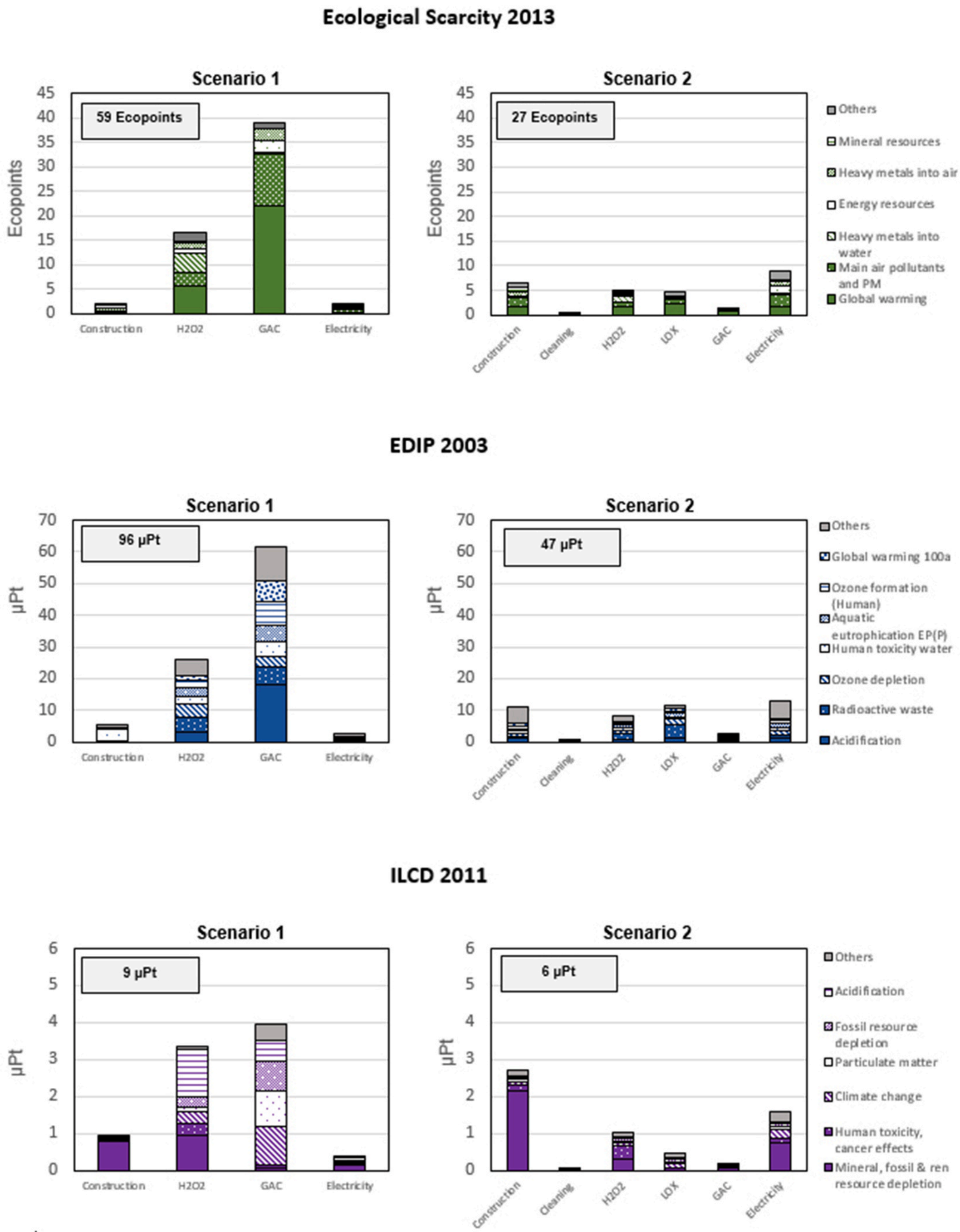

Fig. 3. Absolute contribution to environmental impact categories of individual processes (construction, H_2O_2, GAC, LOX, cleaning chemicals, electricity) in scenario 1 (UV/H_2O_2 + GAC) and scenario 2 (UF-LPRO + O_3/H_2O_2-GAC) according to Ecological Scarcity 2013, EDIP 2003 and ILCD 2011 for the base case (electrical energy from 100% renewable sources).

operational processes. The construction had a low contribution in scenario 1 but were notable in scenario 2. This is due to membrane resources and can also be explained by higher space consumptions for the building required (Text S4). It can be seen, at least for the investigated UV/H_2O_2-based treatment, that robust estimates of the LCA result can be made if only the operational inputs are considered in the model, i.e., electrical energy, H_2O_2 and GAC. However, the following sensitivity analysis shows which parameters have a decisive influence on the results and whether these results are stable in order to draw the same conclusions.

3.2. *Sensitivity analyses*

An overview of all sensitivity analyses is given in the supporting information (Table S18).

3.2.1. *Electrical energy source*

It was observed that the electrical energy source had a strong influence on the total environmental impact (Fig. 4). In comparison to the base-case scenario with 100% renewables (94% hydro-based), the electrical mixes of CH-2015 and ENTSO-2015 resulted in overall increased environmental impacts for both scenarios. In these cases, the LPRO scenario caused higher impacts than the AOP scenario in contrast

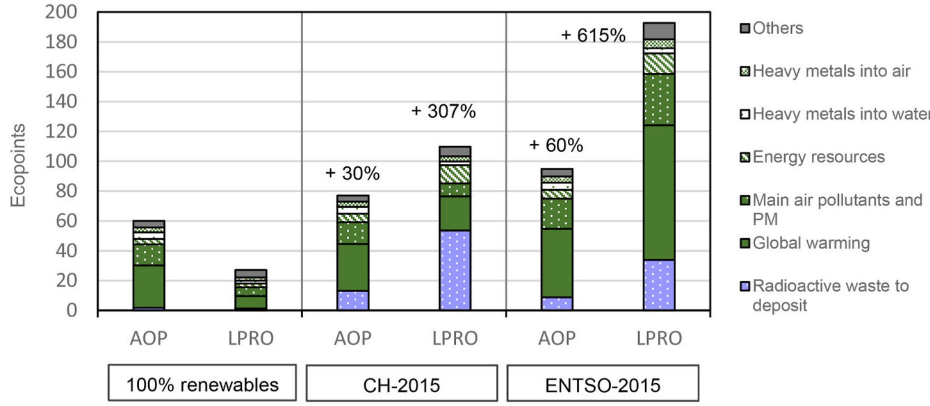

Fig. 4. Absolut environmental impacts of scenario 1 (UV/H_2O_2 + GAC, i.e., AOP) and scenario 2 (UF-LPRO + O_3/H_2O_2-GAC, .i.e., LPRO) according to Ecological Scarcity 2013 with different sources of the electrical energy mix: 100% renewables (base-case), CH-2015 (country mix for Switzerland as in 2015) and ENTSO-2015 (country mix for ENTSO as in 2015).

to the above-discussed baseline scenario with electrical energy from 100 % renewable sources. This is explained by the significantly higher specific electrical energy demand of the LPRO scenario compared to the AOP scenario, i.e., 0.38 to 0.08 kWh/m^3 product water, respectively. The electrical energy mixes considered included electricity from nuclear power plants and fossil resources, which give rise to radioactive waste and global warming.

Furthermore, an analysis with different shares of electricial energy from renewable sources in the CH-2015 electrical energy mix indicated that scenario 2 (LPRO based) only causes less environmental impacts than scenario 1 if the electrical energy is produced by approximately >80% from renewable sources (Fig. 5A). However, the result is only valid for renewable electricity from similar sources as in this study (i.e., mainly hydropower). To compare the impact with other renewable electrical energy sources, Fig. 5B shows the environmental impact of the treatment scenarios if the electrical energy source was only hydropower, wind power, orsolar energy. In general, hydro power had the least environmental impacts followed by wind power. For solar power, the impacts increased significantly, especially for LPRO treatment, indicating a higher environmental impact of solar power compared to the other renewable sources assessed. However, the used process for solar energy in the data base is from the year 2015. Other studies showed that the environmental impacts are decreasing for solar energy due to technological advances, whereas the impacts were relatively constant over time for hydro power (Krebs and Frischknecht, 2018; Messmer and Frischknecht, 2014). Despite this, the electrical energy sources used for the water treatment is crucial; it has a great impact on the LCA results and should be considered for the application. It is concluded that the LPRO-based treatment can only be the preferred option over the UV/H_2O_2-based treatment from an LCA perspective if the renewable energy is hydro- or wind-based, but not solar-based.

3.2.2. *UV/H_2O_2 full-stream treatment*

In principle, the UV/H_2O_2 operational set point to reach a certain treatment goal (in this case: abatement of 25% metformin, see Text S2) can be selected according to Equation S2 (SI) with one degree of freedom (UV as function of H_2O_2 or *vice versa*). With this, the set point of UV fluence and H_2O_2 dose can be considered an optimization problem, as discussed below. Results for a UV fluence range of 4′000 to 10′000 J/m^2 are shown in Table 6.

With the DWS's standard electricity mix (100% renewables), the overall environmental impact was lower with higher UV fluence, i.e., higher electrical energy consumption, and lower H_2O_2 dosages. Higher UV fluence can sometimes be necessary for efficient MPs abatement in AOP, e.g., 7′200 J/m^2 for diatrizoate or 7′500 J/m^2 for *N*-nitrosodimethylamine (NDMA) (Sha et al., 2012; Kovalova et al., 2013). This highlights that higher H_2O_2 concentrations had a greater environmental

Table 6
Sensitivity analysis of total environmental impacts of scenario 1 (UV/H_2O_2 + GAC, i.e., AOP) with different UV fluences and H_2O_2 doses compared to the treatment in the baseline scenario (UV fluence: 6′700 J/m^2, H_2O_2 dose: 6.5 mg/L), assessed with the Ecological Scarcity 2013 method.

UV fluence [J/m^2]	H_2O_2 dose [mg/L]	Total Ecopoints in scenario 1 (AOP)	
		100% renewables	CH-electricity
4′000	11.4	71 (+20%)	82 (+6%)
5′000	9.0	65 (+10%)	78 (+1%)
6′000	7.3	61 (+3%)	77 (−1%)
6′700	6.5	59	78
7′000	6.1	59 (−2%)	77 (−1%)
8′000	5.3	56 (−5%)	77 (−1%)
9′000	4.6	55 (−7%)	78
10′000	4.0	54 (- 9%)	80 (+3%)

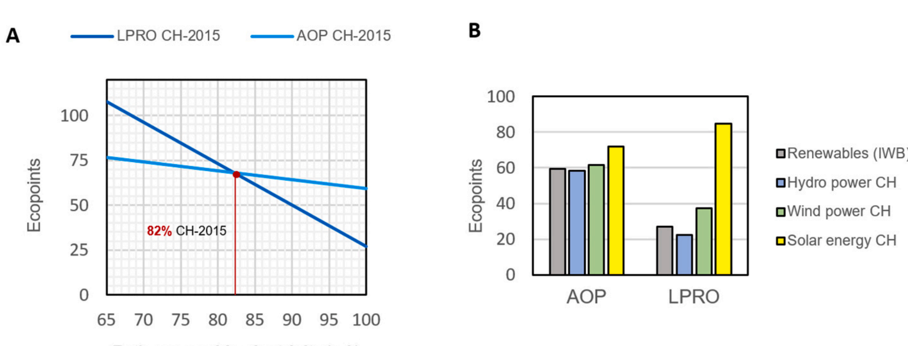

Fig. 5. A: Environmental impacts of scenario 1 (UV/H_2O_2 + GAC, i.e., AOP) and scenario 2 (UF-LPRO + O_3/H_2O_2-GAC, .i.e., LPRO) according to Ecological Scarcity 2013 with different ratios of electrical energy from renewable sources in the country mix for Switzerland as of 2015 (CH-2015). **B**: Total environmental impacts of scenarios 1 (AOP) and 2 (LPRO) according to Ecological Scarcity 2013 with the standard electrical energy mix (100% renewables, mainly hydro-based), 100% hydro power, 100% wind power and 100% solar energy (all from Switzerland).

impact than higher electricity consumption. Apart from this, higher H_2O_2 concentrations could be used for the same impact with higher UV fluence. In contrast, with the CH-2015 electricity mix, higher UV fluence did not lead to less environmental impacts; rather, they increased compared to the standard UV fluence. Thus, the increased electricity consumption created more environmental impacts than the environmental impacts saved due to lower H_2O_2.

Overall, the optimization potential of the UV/H_2O_2 process must be considered small (i.e., <10%) in the relevant range of 5′000 to 10′000 J/m^2 with 9 to 4 mg H $_2O_2$/L, respectively, for both electricity mixes considered here. Besides, a UV fluence of 10′000 J/m^2 is rather unrealistic in typical drinking water treatment applications. Note that the standard operational set point was selected from a strongly simplified model based on the assumption that environmental impacts are only caused from the operation of the UV/H_2O_2, as described in detail in Text S2. It is concluded that with this approach is feasible to determine an operational set point close to the optimum set point determined in a complex model and, thus, it allows for rapid assessments in the process design and layout phase of a process.

3.2.3. O_3/H_2O_2 retentate treatment

Table 7 shows the results of the sensitivity analysis with different DOC-specific O_3 doses and H_2O_2:O_3 ratios. Overall, a variation of treatment parameters in the O_3/H_2O_2 process influenced the total environmental impacts in a range of approximately ±30%. In literature, H_2O_2 addition in a 2:1 to 4:1 M ratio to O_3 proved effective to partially mitigate bromate formation in wastewater treatment with similar bromide concentrations as expected in the concentrate (Soltermann et al., 2017). Furthermore, it appears useful to utilize as few O_3 as possible to reach the treatment goal for two reasons: first, because the environmental burdens from the use of LOX and electrical energy are lowered; second, because the risk of bromate formation is lowered, which could additionally translate into less H_2O_2 use and hence even less environmental impacts. Due to the potential bromate formation, MP elimination by GAC filtration rather than by O_3/H_2O_2 might be necessary, which also could be a suitable solution (Ruhl et al., 2014; Kennedy et al., 2015; Sperlich et al., 2017), although this might increase the environmental impact of the GAC filter due to the likelihood of shortened lifespan. However, it should be emphasized once again that different parameters in this process could also require a change in the parameters in the GAC for a similar treatment goal of the retentate (e.g., BV and EBCT).

3.2.4. GAC filter utilization times

The results of the analysis with higher and lower GAC filter bed volumes (BV) in scenario 1 (AOP based) and scenario 2 (LPRO based) are presented in Table 8. The default value of both scenarios was 80′000 BV, equivalent to a lifespan of 1 year scenario 1, and approximately 2.3 years in scenario 2. In scenario 1, doubling the utilization time (BV = 160′000) decreases the environmental impacts of the whole treatment by 33%. Higher bed volumes beyond 100′000 for quenching residual H_2O_2 in drinking water treatment were reported in literature as effective (Li et al., 2016). For this reason, the prolongated use of GAC for H_2O_2 quenching is considered a realistic option and appears advantageous from an ecological (and economic) point of view.

In scenario 2, the increase in BVs showed no significant effect on environmental impacts. This is explained by the relatively small filter bed necessary to treat the concentrate, hence the GAC's overall contribution to the environmental burdens is relatively low, even in the standard scenario. With a lower BV of GAC, the impacts increased only by 8%. Regenerated activated carbon is already more environmentally friendly than the fresh GAC of scenario 1. Despite this, if MPs abatement for the retentate treatment is considered with GAC rather than with O_3/H_2O_2, the overall environmental impacts might significantly decrease. In reality this might be necessary, because the efficient abatement of potential transformation by-products of the MPs considered would probably be more efficient with a treated BV of < 50′000 (Knopp et al., 2016). However, the GAC treatment is closely coupled with the O_3/H_2O_2 process if the same treatment goal is to be achieved. Depending on the type of activated carbon used and the utilization time of the GAC filter, and therefore potentially other O_3/DOC dosages the impacts could vary significantly. From a LCA perspective, it could be more environmental friendly to reduce the utilization time of GAC and reduce ozone concentrations at the same time, because this could lead to fewer environmental impacts than the opposite. Thus, further research and experimental data for MPs adsorption is required to determine the most suitable adjustment for the retentate treatment.

Table 7
Sensitivity analysis of total environmental impacts with different DOC-specific O_3 doses and H_2O_2:O_3 molar ratios in scenario 2 (UF-LPRO + O_3/H_2O_2-GAC, . i.e., LPRO) compared to base-case scenario (0.6 mg O_3/mg DOC and H_2O_2:O_3 ratio 4), assessed with the Ecological Scarcity 2013 method.

[mg O_3/mg DOC]	Total Ecopoints in scenario 2 (LPRO)		
	H_2O_2:O_3 ratio 2	H_2O_2:O_3 ratio 4	H_2O_2:O_3 ratio 6
0.1	18 (−33%)	18 (- 31%)	19 (−30%)
0.2	19 (−28%)	20 (−25%)	21 (- 22%)
0.3	21 (−24%)	22 (−19%)	23 (14%)
0.4	22 (−19%)	24 (- 13%)	25 (−6%)
0.5	23 (−14%)	25 (−6)%	27 (2%)
0.6	24 (−10%)	27	29 (+9%)
0.7	26 (- 5%)	29 (+6%)	32 (+17%)
0.8	27 (±0%)	30 (+12%)	34 (+25%)
0.9	28 (+4%)	32 (+19%)	36 + (33%)
1.0	29 (+9%)	34 (+25%)	38 (+41%)

Table 8
Sensitivity analysis of total environmental impact with different bed volume (BV) of the GAC filters compared to the respective baseline case (BV 80′000) in scenario 1 (UV/H_2O_2 + GAC, i.e., AOP) and scenario 2 (UF-LPRO + O_3/H_2O_2-GAC, .i.e., LPRO), assessed with the Ecological scarcity 2013 method. n.a.: not assessed.

Bedvolumes [BV]	Total Ecopoints	
	Scenario 1 (AOP)	Scenario 2 (LPRO)
30′000	n.a.	29 (+8%)
60′000	n.a.	27 (+2%)
80′000	59	27
120′000	46 (−22%)	26 (−2%)
160′000	40 (- 33%)	26 (−3%)

3.2.5. Environmental benefits from micropollutant abatement

According to the EDIP 2003 and USEtox methods, for all 21 MPs included new CFs were calculated and are described in Text S7 and shown in Tables S26 and S28. Although the values for the benefits are very different for the two methods studied, the relative order of magnitude is the same for both and negligible compared to the environmental impacts. At most, only approximately 2% (Table S14) of the environmental burdens from the advanced water treatment were compensated for by benefits from MP abatement (EDIP 2003 method). The result was similar to most existing studies for benefits in wastewater treatment (Igos et al., 2012, 2020; Köhler et al., 2012; Zepon Tarpani and Azapagic, 2018; Arzate et al., 2019). However, the result contrasted with two of the studies reviewed (Türk et al., 2013; Li et al., 2019), where the estimated benefit was justified from an ecological perspective. This is explained by the large number of MPs included in the calculation and compounds with a higher toxicity potential such as to 17β-estradiol (Li et al., 2019). Nevertheless, it is concluded that from a global LCA perspective, the investigated additional barriers against MPs in drinking water production do not appear to be justified. Further discussions of the results and benefit estimation approach in LCA in general are provided in Text S8.

In summary, this benefit estimation showed that the results are based on several factors such as toxicity, data quality, number of MPs investigated and methods used. Therefore, the interpretation of the results should be treated with caution. Various additional potential hazardous effects, e.g., the formation of by-products in oxidation processes, genotoxicity (AMES test), mix toxicity of MPs in the investigated drinking water, or endocrine-disrupting effects cannot yet be included in the utilized impact assessment methods and might lead to considerable uncertainties, as further discussed below.

3.3. Uncertainties and limitations of this study

LCAs are models, simplifying reality. Consequently, not all process information can be included due to their complexity (Klöpffer and Grahl, 2014). During the modelling process, boundaries, assumptions, or subjective evaluations are defined, which implies uncertainties that potentially can impact the results significantly. Further, results must be interpreted within the scope of the study. In this section the main uncertainties are described and discussed.

During the goal and scope definition, a few assumptions have been made. No additional abatement of MPs in the GAC filter (scenario 1) was assumed, although some abatement appears to be realistic. For the MAR, a conservative scenario was considered with the 20th percentile abatement efficiency data from the SAT studies. A full-scale treatment might hypothetically have a higher MP abatement, e.g., due to longer retention times in the soil and aquifer (Filter et al., 2021), and would therefore allow smaller designs of the additional treatment for MP abatement, and consequently, reduce the environmental impacts. Experimental data at pilot-scale (UV/H_2O_2) and bench-scale (LPRO) were used to estimate the relative abatements of the investigated MPs in the treatment scenarios. For a scale-up, long-term pilot-scale experiments of all treatment stages are advisable in order to have more reliable data.

Compiling the LCA inventory, the selected database and processes can impact the environmental impacts calculated. Databases are useful and necessary; however, the database selected should be applicable for the system boundaries in the LCA study. This includes region, raw materials, energy, and chemicals etc. (Klöpffer and Grahl, 2014). The database applied here (ecoinvent 3.3) was considered as the most relevant for this LCA. Within a database, the selection of processes is an additional factor, which affects the assessment results. Further the allocation approach has to be documented. In this study the cut-off approach was used (ending of ecoinvent processes: "Alloc Rec") whereas the impacts of materials and resulting waste are allocated to its primary user. For the transparency, all processes used and adaptions of those used processes are listed in Tables S10–S15 (SI). Usually, data quality should be most accurate for the parts, which are main contributors. For the electrical energy and chemicals used as process auxiliaries, which were shown to be the main contributors to the environmental impacts, the data accuracy was considered to be comprehensive in the utilized database. However, for more accurate data on the site-specific inventory of the sensitive parts, better follow-up with the manufacturers of, e.g., GAC, H_2O_2 or LOX would be recommended. Another possibility to analyze the statistical variation of each treatment scenario in Monte Carlo simulations, which was also performed in some of the reviewed literature (Mohapatra et al., 2002; Bonton et al., 2012).

As impact assessment methods can be selected arbitrarily, results can vary greatly. Each method has a different approach with impact calculations based on different values, assumptions and reference data for normalization or weighting factors. These differences must be considered when using an impact assessment method. It should be mentioned again that it is not a standard yet to use several impact assessment methods for a LCA. However, it was shown that the use and comparison of more than one impact method can validate the results and interpretation. In addition, although it is not in line with the ISO guideline to make comparative statements for the public, this was done deliberately because in complex systems it is sometimes necessary for effective decision-making.

Despite this, the resulting environmental impact avoided should be treated with caution. In toxicity impact categories especially, greater differences can be seen which were also observed within other LCA studies (Renou et al., 2008; Saouter et al., 2017). Furthermore, there is still a greater lack of data on long-term, sum-toxicity impacts of MPs (Fent, 2013) and MPs that occur below the analytical detection limit with very high hazardous effect potential such as pyrethroids and endocrine disruptors (Brausch and Rand, 2011; Moschet et al., 2014). Approaches for the inclusion of transformation by-products that could occur in the oxidation processes or mix-toxicity would be further useful and are not yet available, as pointed out by other authors (Renou et al., 2008; Igos et al. 2012, 2020).

4. Conclusions

In this study the environmental impacts of a UV/H_2O_2 advanced oxidation process (AOP) and a low-pressure reverse osmosis (LPRO) based approach were compared as additional treatments in drinking water production for micropollutant (MP) abatement. Further, a methodological LCA approach to accomplish a defined treatment goal was provided. To answer the questions (i-iii) posed at the beginning, the following conclusions can be drawn:

- The LPRO based treatment is the environmentally preferable option for the specific site due to the local electricity mix (100% renewable energy, mainly hydro-based). This result was independent of the impact assessment method applied. This allows a generalized application for other regions even for the sensitivity analysis results.
- Key contributors to the environmental impacts were mainly the operational inputs such as the electrical energy source as well as the impact of the productions of liquid O_2 (LOX, for O_3 generation), H_2O_2 and GAC. The impactsof the operational inputs in the AOP-based treatment accounted for $\approx$ 90% of the environmental impacts. In contrast, a notable contribution of the construction was observed in the LPRO treatment (membrane resources, building, steel), depending on the impact assessment method (24–54%).
- With the market electricity mixes in Switzerland or Europe, the AOP-based treatment caused notable lower environmental impacts than the LPRO-based treatment. The LPRO-based treatment is the preferred option if the electical energy is produced from >80% renewable energy sources (hydro or wind power).
- Variation of treatment parameters of the key contributors impacted the environmental impact within a range of about $\pm$ 10–30%. Considering that the same goal is achieved with different parameters, an optimization from a LCA point of view would be possible.
- The estimated impacts avoided in comparison to the environmental burden of the treatments were negligible and would not justify the treatment from a purely global LCA perspective.
- Further studies are recommended to verify the statements: Pilot-scale experimental data on MP abatement with both treatment approaches in the baseline scenario and the treatment parameters used in the sensitivity analysis (GAC bed volumes, H_2O_2 and O_3 dosages) would help to prove the efficiency of the treatments. In addition, the optimization of the retentate treatment needs further experimental investigation and analysis to include the effective MPs abatement and bromate formation. Other approaches for the O_3 generation (e.g., on-site oxygen production from air) or greener production of H_2O_2 could be investigated to analyze wether the environmental performance would vary. Investigations with alternative sources for activated carbon might be also interesting in consideration of the significant contribution to the environmental impact. To ensure data quality, the emission and input data in the ecoinvent processes should be reconciled with manufacturers wherever possible. As for the benefit estimation in LCA studies, it would be advisable to include only MPs that are known to be toxic to the environment or humans in lower concentrations (e.g. comparable to diclofenac and

below) and where environmental quality standards (EQS) or acceptable daily intake (ADI) values are available for the best data accuracy. In addition, a human health risk-based assessment for the benefit estimation could be also a possible alternative to underline the justification for an additional MP treatment in the drinking water sector. Moreover, a method would be desirable to include possible transformation products and mixing effects in LCA.

CRediT authorship contribution statement

Christine Roth: Investigation, Visualization, Writing – original draft. **Robin Wünsch:** Supervision, Conceptualization, Investigation, Writing – review & editing. **Fredy Dinkel:** Supervision, Validation, Review. **Christoph Hugi:** Supervision, Review. **Richard Wülser:** Conceptualization, Resources, Investigation, Review. **Ralf Antes:** Supervision, Review. **Michael Thomann:** Project Lead, Supervision, Conceptualization, Writing – review & editing.

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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Appendix A. Supplementary data

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