1 **Comparative life cycle environmental and economic assessment of anaerobic membrane**

2 **bioreactor and disinfection for reclaimed water reuse in agricultural irrigation: a case**

3 **study in Italy**

4

5 Alessia Foglia [1], Corinne Andreola [1], Giulia Cipolletta [1], Serena Radini [1], Çağrı Akyol [1,*], Anna

6 Laura Eusebi [1,*], Peyo Stanchev [2], Evina Katsou [2,*], Francesco Fatone [1]

7

8 [1] Department of Science and Engineering of Materials, Environment and Urban Planning-SIMAU, Marche

9 Polytechnic University, 60131 Ancona, Italy

10 [2] Department of Civil Engineering and Environmental Engineering, Institute of Environment, Health and Societies,

11 Brunel University London, Middlesex, UB8 3PH Uxbridge, United Kingdom

12

13 *Corresponding authors (Ç. Akyol; A.L. Eusebi; E. Katsou)

14 E-mail: c.akyol@staff.univpm.it; a.l.eusebi@staff.univpm.it; evina.katsou@brunel.ac.uk

15 Tel: +39 071 2204911 Fax: +39 071 2204729 (Ç. Akyol)

16
17 **ABSTRACT**

18 Reuse of treated wastewater for irrigation purposes is a measure to reduce water stress and

19 overexploitation of freshwater resources. This study aims to investigate the environmental and

20 economic impacts of a current conventional wastewater treatment plant (WWTP) in Peschiera

21 Borromeo (Milan, Italy), and compare possible scenarios to enable reclaimed water reuse for

22 agriculture purposes. Accordingly, we propose alternative disinfection methods (i.e. enhanced

23 UV, peracetic acid) and replace conventional activated sludge (CAS) with upflow anaerobic

24 sludge blanket (UASB) for biological treatment and use anaerobic membrane bioreactor

25 (AnMBR) as the tertiary treatment. Life cycle assessment (LCA) and life cycle costing (LCC)

26 were implemented on the existing full-scale wastewater treatment line and the hypothetical

27 scenarios. In most cases, the impact categories are primarily influenced by fertilizer application

and direct emissions to water (i.e. nutrients and heavy metals). The baseline scenario appears to have the largest environmental impact, except for freshwater eutrophication, human ecotoxicity and terrestrial ecotoxicity. As expected, water depletion is the most apparent impact category between the baseline and proposed scenarios. The UASB + AnMBR scenario gives relatively higher environmental benefits than other proposed scenarios in climate change (-28%), fossil fuel depletion (-31%), mineral resource depletion (-52%), and terrestrial ecotoxicity compared to the baseline. On the other hand, the highest impact on freshwater eutrophication is also obtained by this scenario since the effluent from the anaerobic processes is rich in nutrients. Moreover, investment and operational costs varied remarkably between the scenarios, and the highest overall costs are obtained for the UASB + AnMBR line mostly due to the replacement of membrane modules (24% of the total cost). The results highlighted the importance of the life cycle approach to support decision making when considering possible upgrading scenarios in WWTPs for water reuse.

Keywords: Anaerobic membrane bioreactor (AnMBR); irrigation; life cycle assessment (LCA); life cycle cost (LCC) analysis; tertiary wastewater treatment; reclaimed water reuse

1. Introduction

Mediterranean region has been facing increasing pressure from water scarcity and droughts where freshwater availability is likely to decrease substantially by 2% to 15% for 2°C increase of global temperature due to climate change alone (MedECC Network, 2019). Between 50% and 90% of the total water demand in the Mediterranean basin is dedicated to irrigation, and this demand is projected to rise by 18% until the end of the century (UNEP/MAP Plan Bleu, 2019). Meanwhile, seawater intrusion is another critical problem along the Mediterranean coasts as a consequence of over-exploitation of groundwater (Giannoccaro et al., 2019). All of these issues together with population and economic growth continuously stress freshwater

supplies, which consequently increase the demand for non-conventional water resources (Lee et al., 2018).

Reclaimed wastewater reuse is seen as a solution to help to address above-mentioned challenges, but its potential remains largely untapped from a technical and legislative point of view (Rizzo et al., 2018). Treated wastewater can be used either for non-potable purposes, such as aquifers recharge, irrigation/fertigation, and industrial use, or as a source for drinking water supply after additional treatments. This can help to protect the environment and to enhance water security by managing water resources of the hydrological cycle in a more circular way (Diaz-Elsayed et al., 2019; Giannoccaro et al., 2019). The reuse for agricultural irrigation is by far the most established end-use for reclaimed water (Rizzo et al., 2020). However, the use of reclaimed water relies on many types of advances, not only related to technological approaches but also health, socioeconomic and legal aspects (Salgot and Folch, 2018). In most cases, water reuse strategies are often intended to address the problem of water scarcity without aggravating other environmental problems, thus reflecting the need for their environmental assessment (Meneses et al., 2010). Moreover, water reuse practices can be expensive since a high degree of treatment is required and a separate piping system is needed for the reuse systems to distribute the water.

Currently, approximately 1 billion cubic meters of treated urban wastewater is reused in the EU annually, which accounts for about 2.4% of the treated urban wastewater effluents and less than 0.5% of annual EU freshwater withdrawals. Water-scarce EU countries such as Italy, Spain, and Greece only reuse between 5% to 12% of their effluents (EC, 2020a). This is mainly due to the existing constraints for reclaimed water reuse at the national level. For example, in Italy, the agricultural use of reclaimed water is strongly restricted by law D.Lgs 185/2003 (Ventura et al., 2019). Indeed, the treated wastewater must comply with a range of water directives at the EU and national levels to protect the environment, but the reuse of reclaimed water has to

comply with additional directives/regulations depending on the purpose (Vojtěchovská Šrámková et al., 2018). Recently, the European Commission has developed the Regulation 2020/741 on "*Minimum Requirements for Water Reuse*" (EC, 2020b), where specific indications are provided for the assessment of reclaimed water reuse.

Tertiary treatment (including filtration and/or disinfection) is commonly required to meet the quality standards of reused treated wastewater (Carré et al., 2017). Conventionally, chemical or physical disinfection is applied during wastewater treatment, complying with the stringent microbial safety required for water reuse (Angelakis and Snyder, 2015). Alternatively, well designed and operated membrane bioreactors (MBRs) can also provide efficient removals of solids and pathogens (Foglia et al., 2020). Hai et al. (2014) provided an in-depth overview of the mechanisms and influencing factors of pathogens removal by MBRs and highlighted the practical issues, such as reduced chemical disinfectant dosages and associated economic and environmental benefits. Anaerobic MBR (AnMBR) is a very attractive technology in terms of energy efficiency with energy recovered from sewage and without aeration requirements. In fact, AnMBR has been reported to be net energy positive, leading in cost savings up to €0.023 per m^3 of treated water (Pretel et al., 2016). At the same time, the combined used of anaerobically treated effluent for fertigation can further reduce CO_2 emissions (Jiménez-Benítez et al., 2020).

In most cases, decisions about wastewater treatment are primarily influenced by direct capital and operating costs as long as the design is meeting the required standards, while life-cycle cost (LCC) and life-cycle environmental impacts are rarely considered (Awad et al., 2019). The consideration of a life cycle perspective can help to achieve sustainable wastewater treatment. The Life Cycle Thinking approach is widely applied to assess the environmental sustainability of treatment processes and reveal trade-offs across various environmental impact categories. Besides, life cycle assessment (LCA) provides quantitative information that can support

103 decision making in water reuse practices when considering possible operational scenarios

104 during a strategic planning of reclaimed water reuse (Corominas et al., 2020). For instance,

105 Meneses et al. (2010) investigated tertiary treatment alternatives (i.e. chlorination plus UV

106 treatment; ozonation; and ozonation plus hydrogen peroxide) to enable urban wastewater reuse

107 for non-potable uses (both agricultural and urban uses). Although the assessed disinfection

108 methods had similar environmental impacts, most of the indicators were about 50% higher than

109 the UV disinfection except for the acidification (100% higher) and photochemical oxidation

110 (less than 5%), while chlorination plus UV treatment disinfection was found to have the lowest

111 impact. Up to date, there have been few studies that investigated the LCA of tertiary disinfection

112 methods for reclaimed water reuse (Carré et al., 2017; Muñoz et al., 2009; Pan et al., 2019;

113 Pasqualino et al., 2011) (see the **e-Supplementary file**). Although LCA and market prospects

114 for AnMBR technology are discussed in the review work of Krzeminski et al. (2017b), there

115 are still limited studies on the LCA of AnMBRs for urban wastewater treatment and water reuse

116 mainly due to the lack of full-scale data (Krzeminski et al., 2017a).

117 In this study, advanced tertiary treatment processes were assessed within the frameworks of life

118 cycle approach to analyze water reuse options in a municipal WWTP of Peschiera Borromeo

119 in Northern Italy. LCA and LCC were carried out to compare the impacts of treated wastewater

120 discharge and using conventional sources to supply the water and nutrient demand of the

121 surrounding agricultural area (Baseline scenario) with proposed alternative reuse strategies.

122 Fertigation coupled with different disinfection methods, such as peracetic acid (PAA) and UV-

123 disinfection, was evaluated as the alternative scenarios. Furthermore, a third scenario was

124 suggested to replace the conventional activated sludge process with an anaerobic biological

125 process (i.e. upflow anaerobic sludge blanket (UASB)) and to use AnMBR as tertiary treatment

126 and finally to reuse the effluent in fertigation practice. The main aim was to identify: i) potential

127 environmental and economic benefits and ii) undesired impacts of integrated wastewater

128 treatment and water reuse system. We believe that the outcomes of this work can help to guide

129 reclamation managers for possible upgrading opportunities in WWTPs considering the

130 sustainability aspects.

131 **2. Materials and methods**

132 **2.1.Description of the study area**

133 2.1.1. Peschiera Borromeo WWTP

134 The target WWTP is located in the municipality of Peschiera Borromeo (Lombardy, Italy) and

135 serves a large urban territory (Milan and neighboring municipalities) with a total catchment

136 area of 2,230 ha. Currently, the final effluent is discharged into the Lambro River. The plant

137 has a real treatment capacity of 322,376 population equivalent (PE) with a total average inflow

138 rate of 126,322 m^3/d in 2019 treated in two different wastewater lines as shown in **Fig. 1**. Line

139 1 (**Fig. 1a**) collects and treats the wastewater from the municipalities of Brugherio (MB),

140 Carugate, Cassina de' Pecchi, Cernusco sul Naviglio, Cologno Monzese, Peschiera Borromeo,

141 Pioltello, Segrate, and Vimodrone. Line 2 treats the wastewater from the eastern district of

142 Milan. After pre- and primary treatments, Line 1 consists of a conventional activated-sludge

143 process followed by biological filtration to remove inorganic nitrogen and a final chemical

144 disinfection using peracetic acid (PAA). Line 2 (**Fig. 1b**) includes a two-stage upflow biological

145 filtration (Biofor ®) and two parallel lines of UV disinfection operating at a UV dose of 50

146 mWs/cm^2. Although Line 2 is designed for the purpose of reclaimed water reuse, the effluent

147 is discharged into the Lambro River in both cases. The sludge line consists of the following

148 processes: gravity and dynamic pre-thickening, two-stage anaerobic digestion, gravity post-

149 thickening, and dewatering via centrifuges. The dewatered sludge is transformed in defecation

150 lime and then applied as soil improver. The produced biogas is valorized in two combined heat

151 and power (CHP) units recovering electricity for internal purposes and thermal energy to heat

152 the digesters. The biogas is stored in two gasometers where the unused fraction is burned by

153 two torches.

154   2.1.2.   Surrounding irrigation area

155 Peri-urban areas in the south of Milan (near Parco Agricolo Sud Milano) suffer from water

156 scarcity. Its water demand (12.03 hm^3/y) is mainly required for irrigation purposes. This request

157 can be widely covered by the outflow of Line 2. The surrounding agricultural land has an area

158 of approximately 1500 ha and its main crop is tomato. The nutrient needs (N and P) of tomato

159 in drip irrigation systems are 160 kg N/ha/y and 20 kg P/ha/y (Jiménez-Benítez et al., 2020).

160   **2.2.Treatment scenarios**

161 In order to enable the reuse of the final effluent for agricultural purposes, the following

162 proposed scenarios focused only on the Line 2 of Peschiera Borromeo WWTP. The

163 environmental impacts of the current no reuse configuration was compared to alternative

164 reclamation solutions permitting water reuse. **Table 1** illustrates the effluent characteristics of

165 the plant and the wastewater reuse limits set out by the current Italian legislation as well as

166 those established by the new European Regulation 2020/741 on minimum requirements for

167 water reuse (EC, 2020b).

168 **Table 1.** Effluent concentrations and wastewater reuse limits.

| Parameters	Unit	Effluent Line 1	Effluent Line 2	DM183/2005 **	2020/741 Class A	2020/741 Class B	2020/741 Class C
E. coli	CFU/100ml	284	847	<10	<10	<100	<1000
COD	mg/l	19.3	17.9	<100	-	-	-
BOD$_5$	mg/l	6.7	6	<20	<10	<25	<25
TN	mg/l	10.3	8.4	<15	*	*	*
NH$_4$	mg/l	3.9	1.1	<2	*	*	*
TP	mg/l	0.5	0.7	<2	*	*	*
TSS	mg/l	7.2	6.5	<10	<10	<35	<35
Al	mg/l	0.19	0.12	<1	*	*	*
Fe	mg/l	0.19	0.31	<2	*	*	*

* Italian Ministerial Decree on Water Reuse

** defined by a site-specific risk assessment to be carried out

169

170   The initial (baseline) scenario refers to the current treatment chain of Line 2 where the final

171   effluent is discharged on surface water and the irrigation and nutrient demand are supplied by

172   freshwater and spreading of mineral fertilizers, respectively.

173   To comply with the water reuse regulation, the proposed reuse scenarios (Figure 1) involve

174   upgrading or process modifications of Line 2 as follows:

175   • UV disinfection at higher UV dose (Scenario 1),

176   • Chemical disinfection using peracetic acid PAA (Scenario 2)

177   • Biological treatment with UASB followed by AnMBR (Scenario 3).
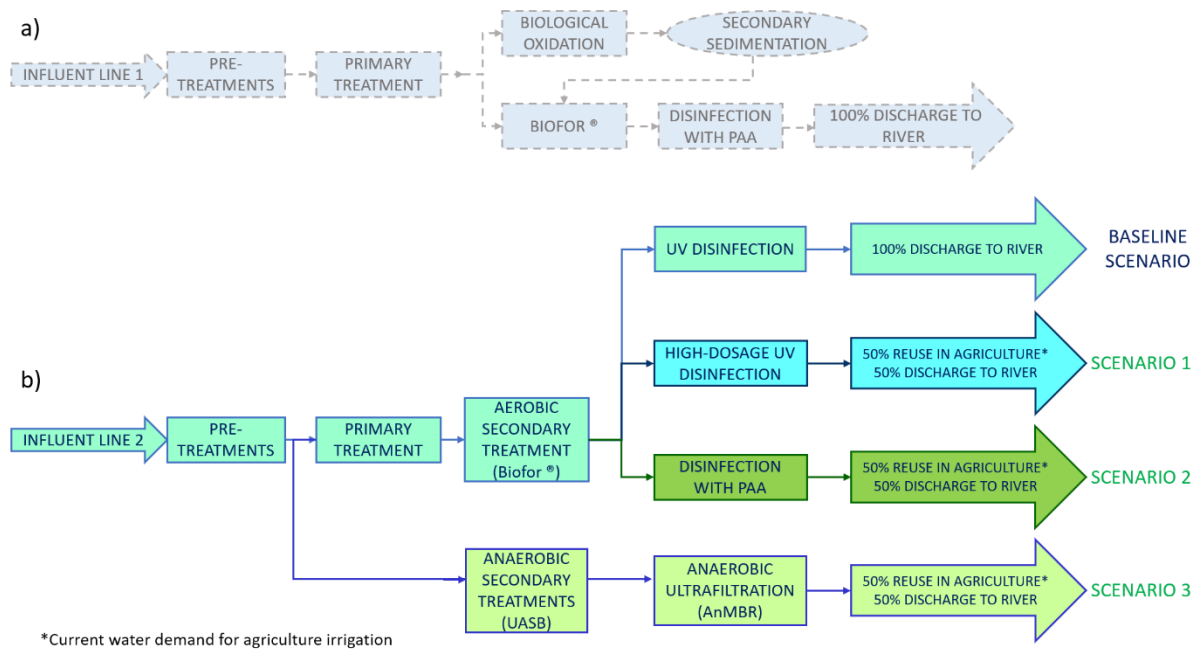
178

179   **Fig. 1.** Flow scheme of the Peschiera Borromeo WWTP: a) Line 1 and b) baseline and proposed

180   scenarios applied to Line 2.

181   In Scenario 1, the existing UV disinfection operates at a dose of 80 mWs/cm^2 to ensure a 3.5

182   log reduction (DEMOWARE, 2016) required to achieve a quality effluent of Class A. In

183   Scenario 2, the UV disinfection is substituted by chemical disinfection unit of 2200 m^3, with a

184   contact time of 49 min and a dosage of 5 mgPAA/L to guarantee the same log reduction

185   (Antonelli et al., 2013) of Scenario 1. Finally, in Scenario 3, an UASB reactor is installed

186 replacing the aerobic secondary treatment. The UASB reactor works at ambient temperature

187 and has a volume of 24,106 m^3, with a hydraulic retention time (HRT) of 9 hours. Then, the

188 UASB is coupled with an anaerobic hollow-fiber ultrafiltration membrane (AnMBR) as the

189 tertiary treatment. The membrane (267,842 m^2) has a nominal pore size of 0.03 μm and operates

190 at the specific flux of 10 L/m^2/h. The ultrafiltration technology in Scenario 3 provides pathogen-

191 free effluent. Therefore, all the alternative scenarios are modeled to reach reclaimed water of

192 class A quality (*E. coli* < 10 CFU/100 ml).

193 The described configurations are assumed to treat the entire inflow rate of Line 2 (64,282 m^3/d).

194 On the other hand, the effective request of water for irrigating the surrounding area is accounted

195 for the half of the WWTP flow. Therefore, 32,959 m^3/d are reused in agriculture and 31,323

196 m^3/d are discharged in the Lambro river. At the same time, the nutrient demand of crops is first

197 covered by the N- and P-content of the reclaimed water and then by a supplementary amount

198 of mineral fertilizer if needed.

199   **2.3.Life cycle assessment methodology**

200 The above-described scenarios were compared to determine the sustainability of the different

201 water reclamation and reuse practices in terms of environmental and economic impacts. The

202 study was carried out following four phases: goal and scope definition, inventory analysis,

203 impact assessment and interpretation. This approach was followed within the framework and

204 principles universally valid to plan and conduct an LCA as established by ISO14044(ISO,

205 2006).

206 The analysis considered the environmental impact directly related to the treatment system

207 (foreground system), as well as the background impact from the supplementary supply chains

208 delivering energy, chemicals, or auxiliaries (background system) using the Ecoinvent v.3.6

209 databases published and maintained by the Ecoinvent Centre in Switzerland, since it is the most

210 renowned database for life cycle inventory (LCI) datasets. It contains approximately 4500-5000

211 harmonized, reviewed and validated datasets for use in LCA that are all fully documented. The

212 Life Cycle Impact Assessment (LCIA) phase was largely automated thanks to the use of LCA

213 software Umberto LCA+ v10.0 in this research. It uses graphic modelling of the product life

214 cycle and allows analyzing, assessing and visualizing the environmental impacts in different

215 impact categories.

216   2.3.1.   System boundaries and functional unit

217 The physical system boundaries (**Fig. 2**) were defined according to the goal and scope of the

218 study, i.e. the comparison of different tertiary treatment schemes. It included not only the water

219 line processes (L2) but also the water and nutrient demand of the surrounding irrigation area

220 (1500 ha). To model foreground and background processes, the following data were considered:

221 the volume and the quality of all water streams, direct GHG emissions from processes, energy

222 consumption, production and transportation of chemicals, wastes disposal, surface water

223 withdrawal and production and spreading of fertilizer. To compare the environmental

224 performance of the different scenarios, 1 m^3 of treated wastewater was selected as the functional

225 unit.
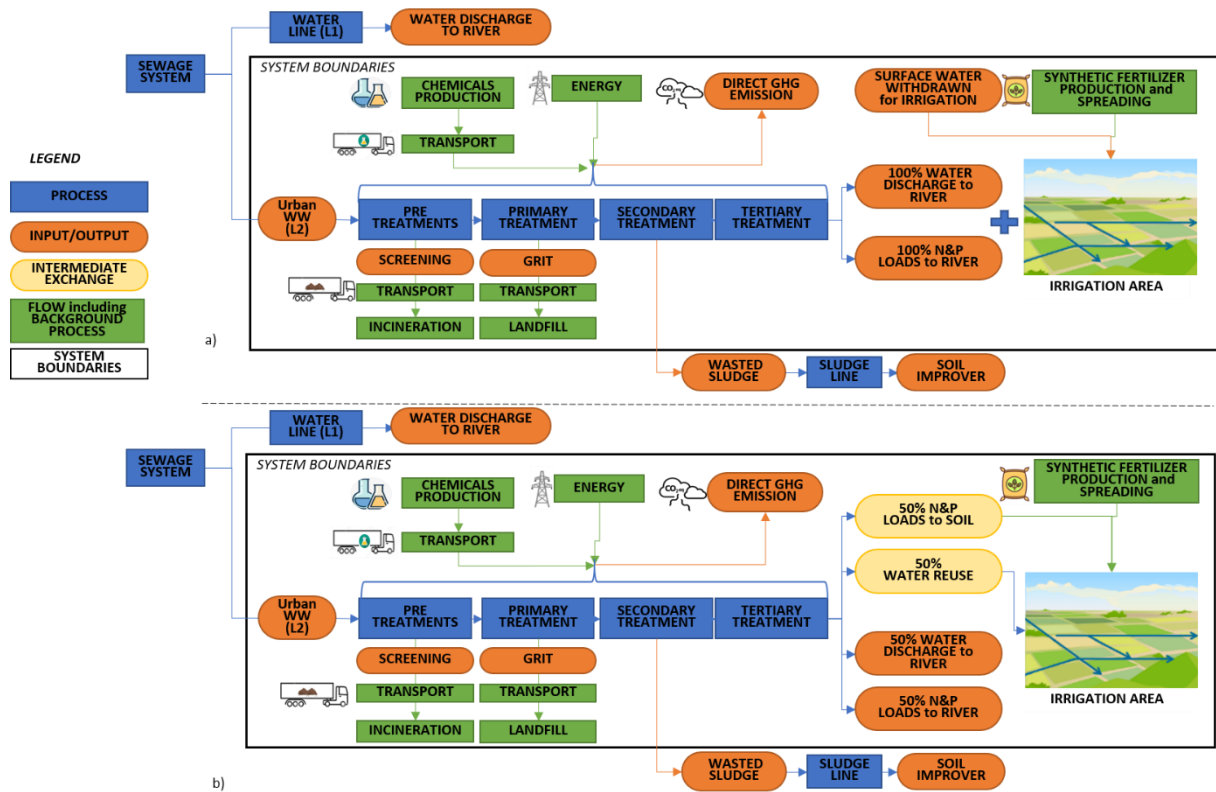
Fig. 2. System boundaries for the life cycle assessment: a) baseline configuration; b) alternative scenarios.

2.3.2. Life cycle inventory

A summary of the Life cycle inventory (LCI) of all the scenarios is given in **Table 2**. The data refer to the main units investigated in this study. The principal parameters of the foreground processes (primary data) were provided by the water utility of the Peschiera Borromeo WWTP. Water quality, consumption of energy and chemicals, amount of waste produced, and related distance to disposal sites refer to the information gathered in 2019. For alternative scenarios, relevant literature values were mainly considered. In Scenario 1, to apply a UV dose of 80 mJ/cm^2 (DEMOWARE, 2016), the disinfection unit utilizes 5,472 kWh/d of electricity. Irrigating with the treated wastewater, 275 kgN/d and 34 kgP/d are provided to crops. Therefore, a supplementary consumption of mineral fertilizer (383 kgN/d and 48 kgP/d) were considered to ensure required plant growth (Jiménez-Benítez et al., 2020). In Scenario 2, the chemical disinfection consumes 2009 kg/d of 16% PAA and 43 kWh/d of electricity. The need

11

for supplementary mineral fertilizer (N and P) was assumed to be equal to Scenario 1. In Scenario 3, based on the data taken from the study of Pretel et al. (2013), the electricity consumption of the UASB was accepted to be 900 kWh/d, while the electricity and thermal energy productions were taken as 1350 kWh/d and 4236 MJ/d, respectively. Furthermore, the electricity consumption of the AnMBR was calculated as 12,381 kWh/d according to Pretel et al. (2013). Considering the membrane cleaning, the amount of NaOCl at 15% for the ordinary cleaning and citric acid at 100% for the recovery cleaning were estimated as 618 kg/d and 93 kg/d, respectively. The N-content in the AnMBR effluent exceeds the N-demand for crops growth, thus only 18 kg/d of supplementary P-fertilizer was considered to be applied to cover the crop requirements.

Table 2. Life cycle inventory of the operation stage for the four scenarios.

Parameters	Units	Baseline scenario	Scenario 1	Scenario 2	Scenario 3
		No reuse	Reuse of class A reclaimed water		
		UV	High dosage UV	PAA	AnMBR
Q treated (L2)	m^3/d	64,282	64,282	64,282	64,282
Q discharged to river	m^3/d	64,282	31,323	31,323	31,323
Q required by crop	m^3/d	32,959	32,959	32,959	32,959
Q surface water withdrawn	m^3/d	32,959	0	0	0
Q water reused for irrigation	m^3/d	0	32,959	32,959	32,959
TN effluent concentration	g/m^3	8	8	8	24
TN required by crop	kg/d	658	658	658	658
TN added by water	kg/d	0	275	275	791
TN added by mineral fertilizers	kg/d	658	383	383	-
Excess TN to soil	kg/d	-	-	-	133
TN discharged to surface water	kg/d	536.35	261.35	261.35	751.73
TP effluent concentration	g/m^3	1.04	1.04	1.04	1.94
TP required by crop	kg/d	82	82	82	82
TP added by water	kg/d	-	34	34	64
TP added by mineral fertilizers	kg/d	82	48	48	18
Excess TP to soil	kg/d	0	0	0	0
TP discharged to surface water	kg/d	67.12	32.71	32.71	60.88
Consumed electricity (secondary treatments)	kWh/d	9792	9792	9792	900

Consumed electricity (tertiary treatments)	kWh/d	2517	5472	43	12,381
Consumed electricity (whole plant)	kWh/d	20,318	23,273	17,844	21,290
Produced electricity	kWh/d	0	0	0	1350
Self-produced heat	MJ/d	0	0	0	4236
PAA at 16% w/w	kg/d	0	0	2009	0
Citric acid at 100% w/w (membrane cleaning)	kg/d	0	0	0	93
NaOCl at 15% w/w (membrane maintenance)	kg/d	0	0	0	618

252

253   Regarding the background processes, the following assumptions were considered: the PAA

254   production was modeled by the production processes of acetic acid (CH_3COOH) and hydrogen

255   peroxide (H_2O_2) assuming that the production of 1 kg of PAA requires 0.45 kg of CH_3COOH,

256   0.79 kg of H_2O_2 and 0.28 kg of water (Buonocore et al., 2018). The lifetime of a UV lamp is

257   equal to 10,000 hours as indicated by the WWTP manager. The residues from screening

258   (disposed of in municipal incineration) were assumed to be composed of 50% of "waste

259   packaging paper" and 50% of "plastic mixture" (Buonocore et al., 2018; Doka, 2003). The final

260   disposal in landfill of the residues from gritting was simulated with "disposal, inert waste, to

261   inert material landfill" (Buonocore et al., 2018; Lorenzo-Toja et al., 2016). The electricity was

262   modeled based on the "Market for electricity, low voltage [IT]".

263   As conducted in other studies (Yoshida et al., 2018), "calcium ammonium nitrate production

264   [RER]" and "triple superphosphate production [RER]" were considered for the N and P

265   fertilizer production, respectively. The mineral fertilizer application was, instead, modeled by

266   the Ecoinvent process "fertilising, by broadcaster [CH]".

267   The impact of transport derives from "Freight, lorry 3.5-7.5 metric ton, EURO 4" for chemicals

268   and "Freight, lorry 16-32 metric, EURO 4" for wastes and sludge disposal. Furthermore, direct

269   GHGs emissions like non-fossil carbon dioxide, fossil methane, and dinitrogen monoxide were

270   also considered in the model.

271   2.3.3. Impact assessment

272   The life cycle impact assessment was carried out by applying the "ReCiPe 2008 Midpoint (H)

273   V1.13 no LT" (results without long-term emissions) method for the following impact

274   categories: climate change (CC), fossil fuel depletion (FD), freshwater eutrophication (FE),

275   mineral resource depletion (MRD), water depletion (WD), freshwater ecotoxicity (T-FET),

276   human toxicity (T-HT) and terrestrial ecotoxicity (T-TET).

277   **2.4.Life cycle cost assessment methodology**

278   Direct capital costs include the cost of infrastructures, mechanical equipment and installation,

279   and electrical and automation systems (Harclerode et al., 2020). For the conventional treatment

280   facilities, the capital expenditures (CAPEX) was developed based on the scaling of costs from

281   comparable projects implemented by the authors, while costs for less common processes like

282   AnMBR were estimated using equipment market pricing and estimated quantities for materials,

283   such as concrete, tank covers, and pre-engineered buildings. The effects of price development

284   (e.g. rising energy prices) and inflation (i.e. the loss of value for money) were not considered

285   in the calculation. The investment cost for a conventional aerobic secondary treatment was

286   taken as 0.04 €/m^3 considering a lifetime of 25 years (Harclerode et al., 2020). Similarly, the

287   CAPEX for the disinfection units were assumed to be 0.0008 and 0.0002 €/m^3 for the UV

288   (scenario 1) and the PAA disinfection (scenario 2), respectively (Collivignarelli et al., 2000;

289   Luukkonen et al., 2015). For scenario 3, a specific total CAPEX of 0.096 €/m^3 was assumed

290   for both secondary and tertiary treatments (Harclerode et al., 2020). For operating expenses

291   (OPEX), most of the information was provided by the water utility, otherwise the Ecoinvent

292   database was considered.

293   The economic lifetime was set to 25 years to be conservative since the investment cost includes

294   both constructions and buildings with a typical lifespan higher than 30 years and machinery to

295   be replaced every 20 years or less. This choice is stated in the ''EVALUATION of the Council

296   Directive 91/271/EEC of 21 May 1991'' concerning urban waste-water treatment that suggests

297   a lifetime of 25 years for WWTPs. **Table 3** provides a summary of the main CAPEX and

298   specific OPEX values.

299   **Table 3**. CAPEX and OPEX cost considered in the proposed scenarios.

CAPEX costs (Peschiera WWTP)	U.M	Values	Reference
Preliminary and primary treatment	k€	8836	(Harclerode et al., 2020)
Conventional activated sludge secondary treatment	k€	22396	(Harclerode et al., 2020)
Disinfection UV	k€	470	(Collivignarelli et al., 2000)
Disinfection PAA	k€	147	(Luukkonen et al., 2015)
Anaerobic treatment (UASB+AnMBR)	k€	36450	(Harclerode et al., 2020)
Biogas conditioning and CHP	k€	11772	(Harclerode et al., 2020)
Specific OPEX costs (Peschiera WWTP)	**U.M**	**Values**	**Reference**
Electricity	€/kWh	0.14	Company information
PAA 16%	€/kg	0.74	Company information
NaOCl 100%	€/kg	0.34	Ecoinvent EURO2005
Citric acid 100%	€/kg	0.78	Ecoinvent EURO2005
MBR replacement frequency	years	10	(Harclerode et al., 2020)
MBR replacement cost for WW treated	€/m^3/d	190	(Harclerode et al., 2020)
UV replacement frequency	hours	10000	Trojan UV technical factsheet
UV lamp cost	€/lamp	343	Trojan UV technical factsheet
N fertilizer	€/kg N	0.47	Ecoinvent EURO2005
P fertilizer	€/kg P$_2$O$_5$	0.24	Ecoinvent EURO2005
Number of labors	N°	6	Company information
Labor salary	€/h	25	Company information
Irrigation water withdrawn from the channel	€/m^3	0.016	ISPRA, 2012
Reclaimed water market price	€/m^3	0.016	ISPRA, 2012

300

301   The total cost in the results is reported as the annual costs, corresponding to the annual OPEX

302   with the CAPEX per annum:

303   Annual costs = OPEX (€/y) + CAPEX (€/y)

304   CAPEX (€/y) = ($\sum$ ⟦investment costs (€))/(economic lifetime (y))

305   **3. Results**

306   **3.1.Baseline scenario assessment**

307 **Fig. 3a** illustrates the allocation between foreground and background environmental impacts.

308 The foreground impact is dominating among all impact categories mainly as a result of the

309 agricultural activities (fertilizer spreading) and the direct emissions to air, water, and soil that

310 are related to the treatment process and to the final effluent discharge. More than 96% of the

311 impact on freshwater eutrophication (FE), water depletion (WD), and human toxicity (T-HT)

312 are caused by direct emissions. **Fig 3b** shows the breakdown of the environmental footprint

313 among the different stages of the water treatment supply chain, namely: pre- and primary

314 treatments, biological process, disinfection and final use. The latter includes water withdrawal

315 and fertilizer application in agriculture. As expected, the most significant environmental impact

316 is related to the final use, followed by primary treatment where phosphorous is chemically

317 removed by dosing poly-aluminium chloride (PAC). Specifically, the final use causes about

318 75% of the impact on climate change and fossil fuel depletion, and more than 98% on

319 freshwater eutrophication and water depletion. The relative impact of primary treatments

320 (>7%), as well as biological processes (>6%), are more evident on climate change, fossil fuel

321 depletion, mineral resource depletion and terrestrial ecotoxicity categories. As an energy-

322 intensive process, the disinfection affects mainly the fossil fuel depletion and climate change

323 categories, however, it is still significantly lower than the other stages (<2%). **Fig. 3c** shows

324 the contribution analysis of each impact category based on the origin of the impact and related

325 resources (i.e. energy, chemicals, direct emissions, etc.). Fertilizer spreading has a significant

326 contribution to climate change, fossil fuel depletion, mineral resource depletion and terrestrial

327 ecotoxicity since it is strongly related to fossil fuel combustion. The direct emissions to water

328 refer to the nutrients and heavy metals content of the discharged effluent to the surface water

329 body. They affect mainly the freshwater eutrophication, the freshwater ecotoxicity, and human

330 ecotoxicity with relative contributions of about 77%, 65%, and 86%, respectively.

331 Approximately 20% of the environmental burden in the freshwater eutrophication category is

due to the P-content in the irrigation water (direct emission to soil) and 80% is due to the P-content in the discharged water (direct emission to water). The water depletion is influenced almost entirely by the direct withdrawal of water from the environment while climate change, fossil fuel depletion, and terrestrial ecotoxicity are mainly affected by electricity consumption and transportation. The chemicals mostly have an impact on the categories of terrestrial ecotoxicity (20%), fossil fuel depletion (8%), freshwater ecotoxicity (7%), and climate change (7%).
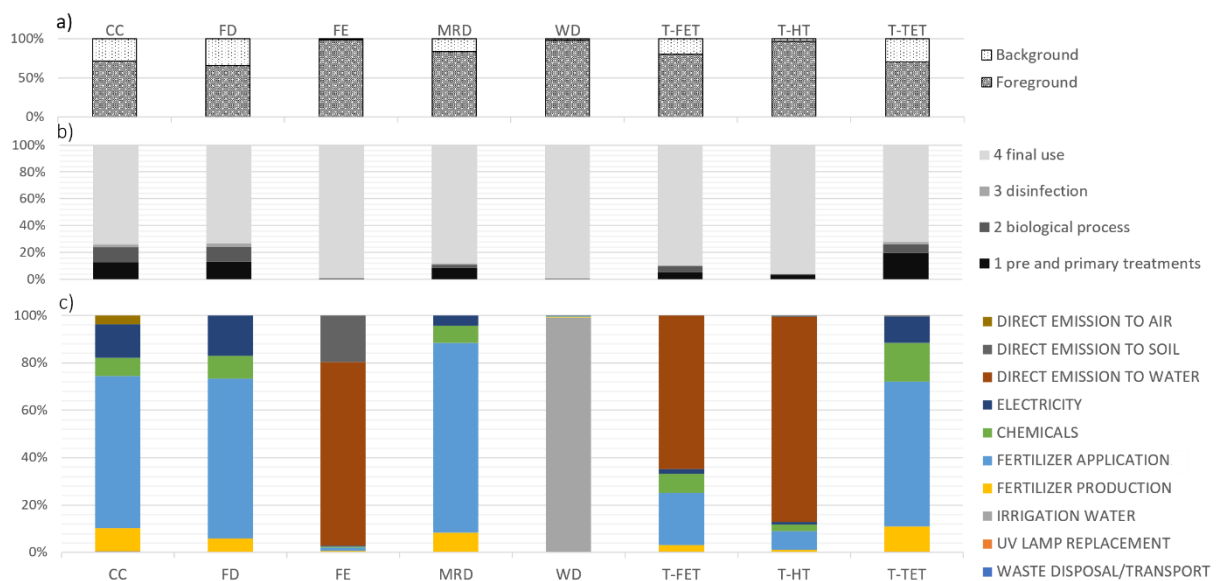
Fig. 3. Environmental profile of the existing treatment configuration (baseline scenario) in the Peschiera Borromeo WWTP **a)** foreground and background environmental impacts **b)** impact of each treatment stage **c)** contributions on each impact category

3.2. Scenario analysis

An overall comparison of the relative impacts of each scenario is presented in **Fig. 4**. In most impact categories, the proposed water reuse scenarios show significant environmental benefits. The baseline scenario represents the largest environmental impact in all categories, except for freshwater eutrophication, human toxicity, and terrestrial ecotoxicity. As expected, the largest benefit is observed in water depletion category since the abstraction of freshwater is replaced with reclaimed water reuse. Scenario 1 and 2 show a significant reduction in freshwater

17

eutrophication due to the lower amount of P directly discharged to the river. On the other hand,

Scenario 3 rises the impact on freshwater eutrophication since the UASB + AnMBR effluent is

highly rich in nutrients that leads to higher rate of P-release even if the same low quantity of

water is discharged into the river. However, due to the savings of producing and spreading

mineral fertilizer, this scenario has a relatively lower impact on fossil fuel depletion (68%). A

slight reduction of 3% and 6% in fossil fuel depletion impact is observed in Scenarios 1 and 2

compared to baseline scenario, respectively, since they are strongly related to fossil fuel

combustion required in energy production and in the transport of the disinfection agents.

Looking at the toxicity-related categories, the toxicity in the water environment is higher in the

Baseline since traces of heavy metals present in the effluent are fully discharged into the river.

However, increased toxicity levels for terrestrial and human categories are observed in the

alternative scenarios where the toxic compounds are partially sent to the soil.
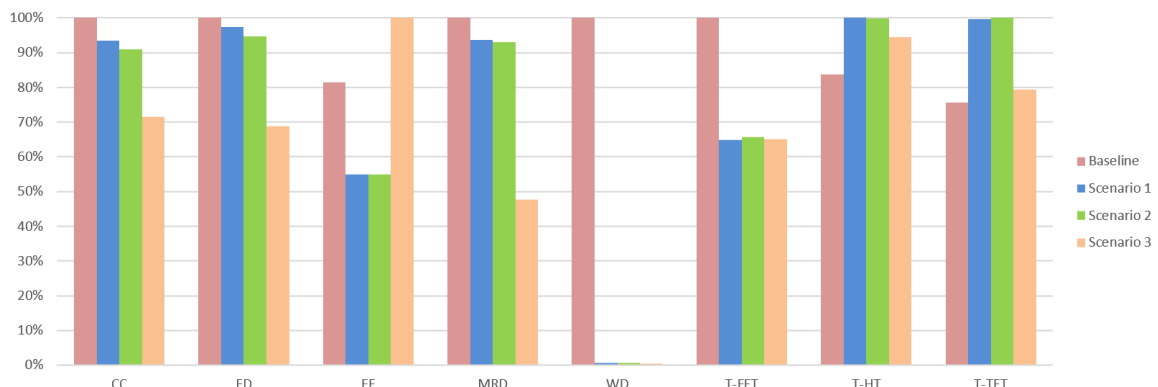
Fig. 4. Comparison of the relative environmental impacts of each scenario.

3.2.1. Scenario 1 – Enhanced UV disinfection

Scenario 1 is the upgraded version of the baseline scenario with an enhanced UV application

to reach an effluent quality of class A (*E. coli* <10) to be reused in agriculture. The current plant

configuration performs poor nutrient removal resulting in a final effluent of N=8mg/l and P=1

mg/l. **Fig. 5** shows the environmental performance of Scenario 1 relative to the baseline

scenario. Although there is higher electricity consumption in Scenario 1, the climate change

impact shows a 7% reduction. This is because the avoided emissions of the displaced fertilizer

production and application that are much higher than the ton of CO_2 equivalent related to the

intensified energy demand. Since the irrigation water comes from the reuse of reclaimed water,

the largest benefit is observed in the water depletion category. Freshwater eutrophication shows

a significant reduction (32%) due to the avoided direct emissions to water produced by the

effluent discharge. For the same reason, a large change (-35%) is seen in the freshwater

ecotoxicity category compared to the baseline scenario. However, a significant negative impact

is observed in human toxicity (+19%), and terrestrial ecotoxicity (+32%) due to the presence

of traces of heavy metals in the reclaimed water. The shift of direct emissions from water to the

soil results in a trade-off between freshwater ecotoxicity and human toxicity and terrestrial

ecotoxicity. The reduction of the mineral resource depletion (6%) is also affected by the
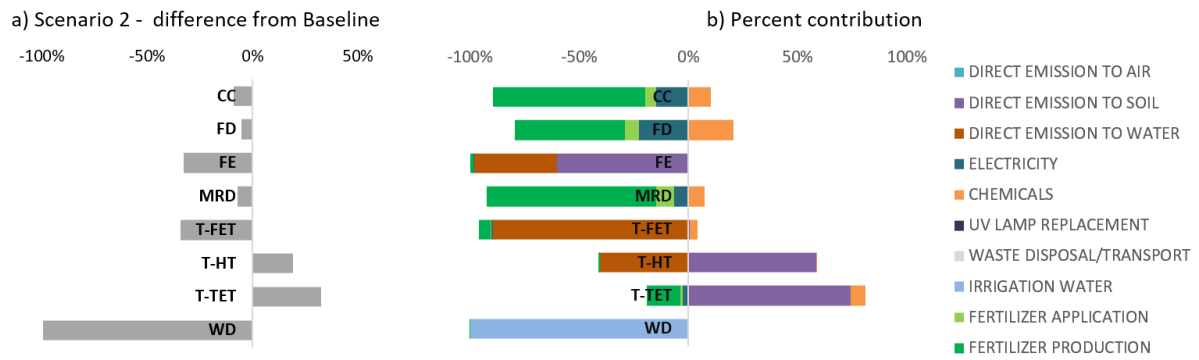
displaced N and P fertilizer.
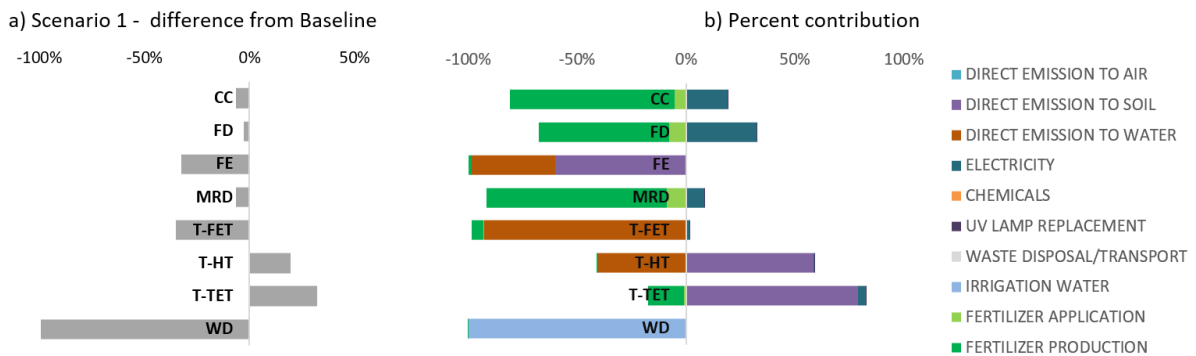
Fig. 5. Environmental performance of enhanced UV disinfection relative to baseline scenario
a) as overall relative differences in each category; b) percentage contribution analysis based on
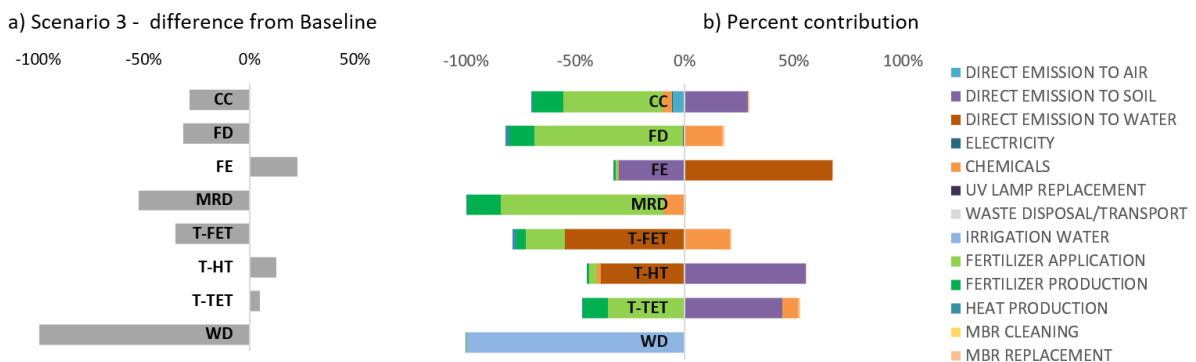the individual processes and sources of impact.

3.2.2. Scenario 2 - Chemical disinfection using peracetic acid

Scenario 2 is the alternative version of the baseline scenario where the UV disinfection is

replaced by PAA disinfection. **Fig. 6** shows the environmental performance of Scenario 2

relative to the baseline scenario.

a) Scenario 2 - difference from Baseline   b) Percent contribution

Legend:
- DIRECT EMISSION TO AIR
- DIRECT EMISSION TO SOIL
- DIRECT EMISSION TO WATER
- ELECTRICITY
- CHEMICALS
- UV LAMP REPLACEMENT
- WASTE DISPOSAL/TRANSPORT
- IRRIGATION WATER
- FERTILIZER APPLICATION
- FERTILIZER PRODUCTION

Fig. 6. Environmental performance of scenario with chemical disinfection using peracetic acid relative to baseline scenario: a) as overall relative differences in each category; b) percentage contribution analysis based on the individual processes and sources of impact.

The overall environmental performance of the chemical disinfection scenario is similar to that of Scenario 1. The use of chemicals for disinfection leads to an additional impact in climate change (+9%), fossil fuel depletion (+18%) and mineral resource depletion (+6%) compared to baseline case. However, the avoided emissions from energy savings and displaced fertilizer outweigh them significantly and result in an overall reduction in most of the impact categories. The chemical disinfection of Scenario 2 shows a slightly higher reduction (2%) compared to the energy-intensive UV disinfection of Scenario 1, both in climate change and fossil fuel depletion. Similar to Scenario 1, there is a significant reduction in freshwater ecotoxicity while the end-use of water on land plays a large role in human toxicity and terrestrial ecotoxicity. Finally, the impact reduction on freshwater eutrophication is again determined by the avoided direct emissions to water.

3.2.3. Scenario 3 - Biological treatment with UASB followed by AnMBR as tertiary treatment

Scenario 3 includes the UASB as biological treatment followed by the AnMBR as the tertiary treatment and thus eliminates the need for a disinfection unit. The final effluent is richer in N and P contents compared to the other scenarios as 24 mg/l and 2 mg/l, respectively. **Fig. 7** shows

the environmental performance of Scenario 3 relative to the baseline scenario. Besides water

depletion, Scenario 3 shows much higher relative benefits than Scenario 1 and 2 in climate

change (-28%), fossil fuel depletion (-31%), mineral resource depletion (-52%) and freshwater

ecotoxicity (-35%) compared to the baseline scenario (**Fig.7 a)**. As can be seen from the

contribution analysis in (**Fig.7 b)** the latter is attributed to the avoided fertilizer spreading,

where relative reductions of 45% in climate change, 68% in fossil fuel depletion, 74% in

mineral resource depletion and 18% in freshwater ecotoxicity are obtained. The direct

emissions to soil (heavy metals and nutrients) are the main contributor to human toxicity

(+55%) and terrestrial ecotoxicity (+45%). It is assumed that the dissolved methane in the

permeate is not recovered through advanced treatment and thus raising the global warming

potential by 28%. However, this is balanced by the avoided direct emissions from aerobic

processes and the reduced amount of chemicals required for P-removal via chemical

precipitation. Moreover, the greater nutrient content of the effluent provides the highest

fertilizer substitution rate. It produces the most positive effect on impact reduction. However,

Scenario 3 shows a significantly larger impact on eutrophication (68%) due to the increased

amount of direct emissions to water (+23% compared to the baseline) mainly related to the

fraction of nutrient-rich water which is not reused but directly discharged into a water body.

The additional impact from membrane replacement, maintenance and cleaning results in a

negligible additional impact (<1%) compared to other factors. Finally, differently to Scenarios

1 and 2, the trade-off between the toxicity categories is relatively smaller.

433

Fig. 7. Environmental performance of UASB followed by AnMBR relative to baseline scenario: a) as overall relative differences in each category; b) percentage contribution analysis based on the individual processes and sources of impact.

3.3. Economic performance assessment

The economic performance of the considered scenarios is shown in **Fig. 8**. In terms of biological treatment, Scenario 3 does not have a significant rising in CAPEX compared to the other scenarios. CAPEX comprises 40-44% of the total cost during the lifetime of the plant. In all scenarios, the greatest OPEX belongs to energy consumption except for Scenario 3 where the membrane replacement plays a major role followed by the energy demand. The differences between the UV disinfection (scenario 1), PAA disinfection (scenario 2) and the baseline are negligible in terms of total costs. On the other hand, the relative contributions are different. Specifically, in Scenario 1, the UV replacement has a relative impact of 8%, while in Scenario 2, the chemical consumption has a relative impact of 19%. The larger cost for PAA supply is balanced by lower energy consumption and the avoided periodic replacement of expensive equipment like a UV lamp. Although the Scenario 3 has the best environmental performance considering almost all the indicators, it has the highest overall costs due to the membrane investment and replacement. The substitution every ten years of the membrane modules contributes to 24% of the total cost. From a wastewater treatment point of view, the environmental benefit of Scenario 3 should encompass the highest investment and operational cost of the membrane reactor.
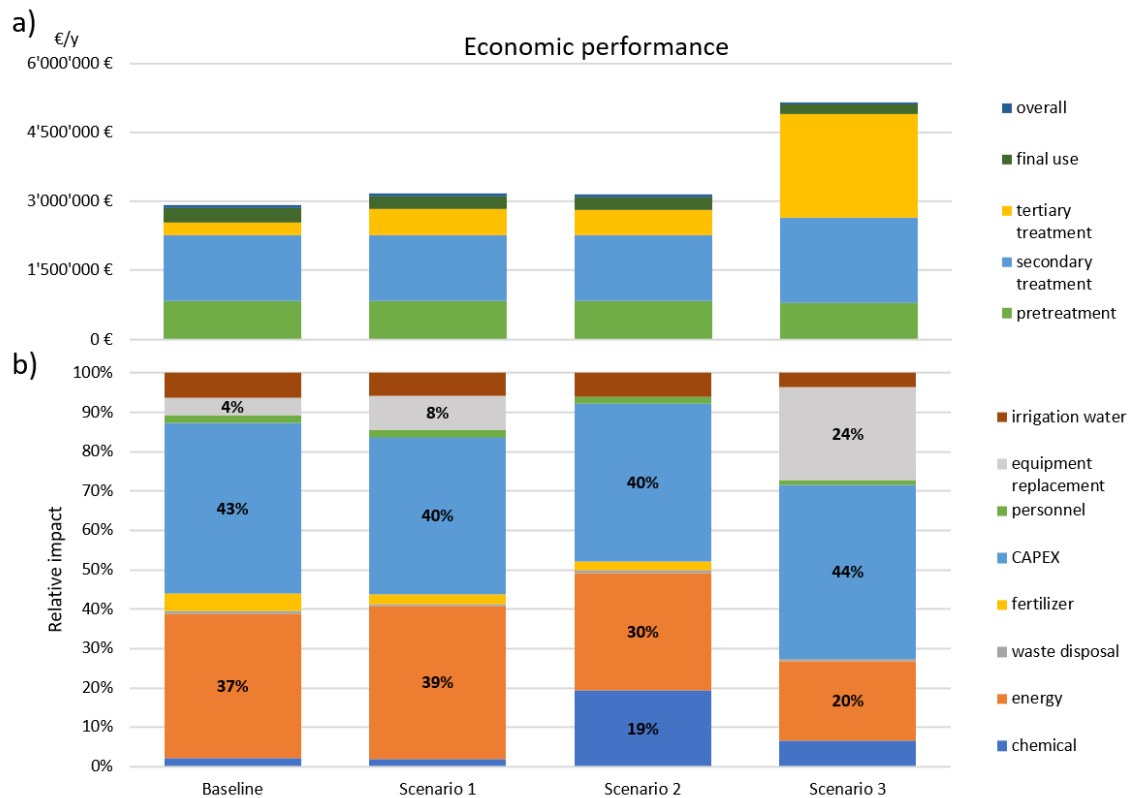
Fig 8. Economic evaluation in each scenario: a) phase contribution b) relative impact.

4. Discussion

This work demonstrated that the combination of anaerobic secondary treatments (i.e. UASB) with an ultrafiltration chamber (AnMBR) can strongly reduce the environmental impact of final discharges compared to the CAS line followed by disinfection processes (Baseline, Scenario 1 and Scenario 2) when the reclaimed water is intended to be reused in agriculture. Furthermore, it showed the necessity to recognize WWTPs more like water resource recovery facilities (WWRFs) where not only water but also value-added materials, nutrients and energy are recovered (Akyol et al., 2020), while economic cost and carbon footprint are minimized. At the same time, this could provide an economic benefit for farmers since they can reduce mineral fertilizer acquisition, resulting in an economic and environmental win-win situation. In alternative scenarios, high environmental impacts are associated with eco- and human toxicity categories as a result of using reclaimed water in agriculture. The impacts on eco- and human toxicity are primarily related to heavy metals contamination of soil. Tangsubkul et al. (2005)

23

470 noted that the increased impacts on the terrestrial environments might be inevitable when

471 selecting a technology that optimizes the recycling of wastewater nutrients, due to the

472 potentially higher metals loading associated with the higher nutrient recovery and reuse (Fang

473 et al., 2016). Turan et al. (2018) evaluated the effects of chitosan (CH) and biochar (BC) on

474 growth and nutritional quality of brinjal plant together with in situ immobilization of heavy

475 metals in a soil polluted with heavy metals due to irrigation with wastewater. In fact, this is a

476 critical point that the reclamation managers and farmers should pay attention to the possible

477 ways to neglect heavy metal contamination via reclaimed wastewater reuse. Strong exposure

478 of plants to heavy metals in the soil modifies the majority of metabolic and cellular processes

479 in plant cells, which in return pose serious ecological risks and human health hazards (Turan,

480 2019).

481 In a recent study, environmental and human health impacts of water reclamation for crop

482 irrigation was comparatively evaluated by the combination of scenario modeling, life-cycle

483 impact analyses and Monte Carlo simulations (Pan et al., 2019). Similar to our findings, the

484 authors indicated that adverse environmental and human health impacts were dependent on

485 energy and chemical inputs (such as iron chloride for enhanced phosphorus removal). In fact,

486 the direct benefits of water reclamation could be offset by other adverse environmental and

487 human health impacts, (e.g. mineral depletion, global warming, ozone depletion, ecotoxicity)

488 which are associated with increased usage of energy and chemicals for rigorous removal of

489 contaminants, that can further affect decision-making. LCA may provide some surprising

490 results, too, such as the case of Carré et al., (2017). Five different tertiary treatments were

491 compared where the combination of a sand filter with UV disinfection or the use of UF alone

492 was found to be equivalent in terms of environmental impact for most of the midpoint indicators

493 chosen although the processes completely vary from each other.

494   Specifically, in our study, the system boundaries involved the water and nutrient demand of

495   crops, besides different technical solutions for water reclamation. Hence, our inventory includes

496   the off-set of mineral fertilizer production and freshwater withdrawn as conducted by previous

497   works (Cornejo et al., 2016; Pan et al., 2019). Further, the LCI also considers the spreading of

498   fertilizer via tractors, as well as nutrients excess due to reclaimed water if the case. The

499   spreading plays a major role in such impact categories related to fossil fuel combustion, while

500   nutrients in the eutrophication. When fertigation is implemented, N and P are directly supplied

501   through irrigation system avoiding the use of tractors and broadcasters. Therefore, it strongly

502   influences and reduces the environmental impact. This also stresses the higher benefits obtained

503   by the anaerobic processes (Scenario 3) in almost all the categories, except for the freshwater

504   eutrophication. To overcome the eutrophication issue that occurs when P-rich effluents are

505   discharged into water bodies, using both aerobic and anaerobic treatments is recommended.

506   This will make the modulation of the quality of the treated wastewater possible: the UASB-

507   AnMBR effluent will provide the crops with nutrients and water while the effluent from aerobic

508   CAS system will be used for nutrient dilution or irrigation. Temporal variability of the nutrients

509   and water demands from crops will determine the flow rate partition between the two treatment

510   lines.

511   **5. Conclusion**

512   All three proposed configurations aim to obtain an effluent quality of class A (*E. coli* < 10

513   CFU/100 ml) according to the new EU Regulation on minimum requirements for water reuse

514   in agriculture. The LCA clearly demonstrated that the reuse of reclaimed water provides more

515   environmental benefits than the discharge of treated water. No significant differences were

516   obtained between the disinfection by peracetic acid (PAA) or UV. The environmental

517   performance of the PAA disinfection scenario is mainly affected by chemical transportation,

518   while the UV disinfection is influenced by the energy production mix and amount of energy

519 used. The impact related to energy consumption is expected to be less significant in the future

520 with the increase amount of renewable energy. In almost every impact category, higher benefits

521 were obtained by applying the anaerobic configuration (UASB + AnMBR), except for the

522 freshwater eutrophication. Furthermore, the highest overall costs belong to the AnMBR line,

523 but its environmental benefits can encompass the high investment and operational cost. For

524 future research, actual removal of heavy metals, as well as contaminants of emerging concern,

525 can be considered in the proposed scenarios, especially stressing the differences between CAS

526 and AnMBR systems.

527 **AUTHOR CONTRIBUTIONS**

538 **FUNDING**

542 **ACKNOWLEDGMENTS**

545 **REFERENCES**

546 Akyol, Ç., Foglia, A., Ozbayram, E.G., Frison, N., Katsou, E., Eusebi, A.L., Fatone, F., 2020.

547 Validated innovative approaches for energy-efficient resource recovery and re-use from

548 municipal wastewater: From anaerobic treatment systems to a biorefinery concept. Crit.

549 Rev. Environ. Sci. Technol. 50, 869–902.

550 https://doi.org/10.1080/10643389.2019.1634456

551 Angelakis, A.N., Snyder, S.A., 2015. Wastewater treatment and reuse: Past, present, and

552 future. Water (Switzerland) 7, 4887–4895. https://doi.org/10.3390/w7094887

553 Antonelli, M., Turolla, A., Mezzanotte, V., Nurizzo, C., 2013. Peracetic acid for secondary

554 effluent disinfection: A comprehensive performance assessment. Water Sci. Technol. 68,

555 2638–2644. https://doi.org/10.2166/wst.2013.542

556 Awad, H., Gar Alalm, M., El-Etriby, H.K., 2019. Environmental and cost life cycle

557 assessment of different alternatives for improvement of wastewater treatment plants in

558 developing countries. Sci. Total Environ. 660, 57–68.

559 https://doi.org/10.1016/j.scitotenv.2018.12.386

560 Buonocore, E., Mellino, S., De Angelis, G., Liu, G., Ulgiati, S., 2018. Life cycle assessment

561 indicators of urban wastewater and sewage sludge treatment. Ecol. Indic. 94, 13–23.

562 https://doi.org/10.1016/j.ecolind.2016.04.047

563 Carré, E., Beigbeder, J., Jauzein, V., Junqua, G., Lopez-Ferber, M., 2017. Life cycle

564 assessment case study: Tertiary treatment process options for wastewater reuse. Integr.

565 Environ. Assess. Manag. 13, 1113–1121. https://doi.org/10.1002/ieam.1956

566 Collivignarelli, C., Bertanza, G., Pedrazzani, R., 2000. A comparison among different

567 wastewater disinfection systems: Experimental results. Environ. Technol. (United

568   Kingdom) 21, 1–16. https://doi.org/10.1080/09593332108618137

569   Cornejo, P.K., Zhang, Q., Mihelcic, J.R., 2016. How Does Scale of Implementation Impact

570   the Environmental Sustainability of Wastewater Treatment Integrated with Resource

571   Recovery? Environ. Sci. Technol. 50, 6680–6689.

572   https://doi.org/10.1021/acs.est.5b05055

573   Corominas, L., Byrne, D.M., Guest, J.S., Hospido, A., Roux, P., Shaw, A., Short, M.D., 2020.

574   The application of life cycle assessment (LCA) to wastewater treatment: A best practice

575   guide and critical review. Water Res. 184. https://doi.org/10.1016/j.watres.2020.116058

576   DEMOWARE, 2016. Partial disinfection technologies for water reuse: case studies and

577   design guidelines.

578   Diaz-Elsayed, N., Rezaei, N., Guo, T., Mohebbi, S., Zhang, Q., 2019. Wastewater-based

579   resource recovery technologies across scale: A review. Resour. Conserv. Recycl. 145,

580   94–112. https://doi.org/10.1016/j.resconrec.2018.12.035

581   Doka, G., 2003. Life cycle inventory of wastewater treatment. Swiss Cent. Life Cycle Invent.

582   Tech. Rep. Ecoinvent Rep.

583   EC, 2020a. Water Reuse Report. https://ec.europa.eu/environment/water/reuse.htm (accessed

584   on 14.09.2020).

585   EC, 2020b. Regulation (EU) 2020/741 of the European Parliament and of the Council of 25

586   May 2020 on Minimum Requirements for Water Reuse.

587   Fang, L.L., Valverde-Pérez, B., Damgaard, A., Plósz, B.G., Rygaard, M., 2016. Life cycle

588   assessment as development and decision support tool for wastewater resource recovery

589   technology. Water Res. 88, 538–549. https://doi.org/10.1016/j.watres.2015.10.016

590   Foglia, A., Akyol, Ç., Frison, N., Katsou, E., Laura, A., Fatone, F., 2020. Long-term

591   operation of a pilot-scale anaerobic membrane bioreactor (AnMBR) treating high salinity

592   low loaded municipal wastewater in real environment. Sep. Purif. Technol. 236, 116279.

593 https://doi.org/10.1016/j.seppur.2019.116279

594 Giannoccaro, G., Arborea, S., de Gennaro, B.C., Iacobellis, V., Piccinni, A.F., 2019.

595 Assessing reclaimed urban wastewater for reuse in agriculture: Technical and economic

596 concerns for Mediterranean regions. Water 11, 1511. https://doi.org/10.3390/w11071511

597 Hai, F.I., Riley, T., Shawkat, S., Magram, S.F., Yamamoto, K., 2014. Removal of pathogens

598 by membrane bioreactors: A review of the mechanisms, influencing factors and

599 reduction in chemical disinfectant dosing. Water (Switzerland) 6, 3603–3630.

600 https://doi.org/10.3390/w6123603

601 Harclerode, M., Doody, A., Brower, A., Vila, P., Ho, J., Evans, P.J., 2020. Life cycle

602 assessment and economic analysis of anaerobic membrane bioreactor whole-plant

603 configurations for resource recovery from domestic wastewater. J. Environ. Manage.

604 269, 110720. https://doi.org/10.1016/j.jenvman.2020.110720

605 ISO, 2006. Environmental Management—Life Cycle Assessment—Requirements and

606 Guidelines (ISO 14044). Geneva.

607 Jiménez-Benítez, A., Ferrer, F.J., Greses, S., Ruiz-Martínez, A., Fatone, F., Eusebi, A.L.,

608 Mondéjar, N., Ferrer, J., Seco, A., 2020. AnMBR, reclaimed water and fertigation: Two

609 case studies in Italy and Spain to assess economic and technological feasibility and CO2

610 emissions within the EU Innovation Deal initiative. J. Clean. Prod. 270.

611 https://doi.org/10.1016/j.jclepro.2020.122398

612 Krzeminski, P., Katsou, E., Dosoretz, C.G., Esteban-García, A.L., Leverette, L., Malamis, S.,

613 Nenov, V., Robles, Á., Seco, A., Syron, E., 2017a. Membranes in wastewater treatment,

614 in: Lema, J.M., Suarez, S. (Eds.), Innovative Wastewater Treatment & Resource

615 Recovery Technologies: Impacts on Energy, Economy and Environment. IWA

616 Publishing, p. 0. https://doi.org/10.2166/9781780407876_0129

617 Krzeminski, P., Leverette, L., Malamis, S., Katsou, E., 2017b. Membrane bioreactors – A

review on recent developments in energy reduction, fouling control, novel

configurations, LCA and market prospects. J. Memb. Sci. 527, 207–227.

https://doi.org/10.1016/j.memsci.2016.12.010

Lee, E.J., Criddle, C.S., Geza, M., Cath, T.Y., Freyberg, D.L., 2018. Decision support toolkit

for integrated analysis and design of reclaimed water infrastructure. Water Res. 134,

234–252. https://doi.org/10.1016/j.watres.2018.01.037

Lorenzo-Toja, Y., Alfonsín, C., Amores, M.J., Aldea, X., Marin, D., Moreira, M.T., Feijoo,

G., 2016. Beyond the conventional life cycle inventory in wastewater treatment plants.

Sci. Total Environ. 553, 71–82. https://doi.org/10.1016/j.scitotenv.2016.02.073

Luukkonen, T., Heyninck, T., Rämö, J., Lassi, U., 2015. Comparison of organic peracids in

wastewater treatment: Disinfection, oxidation and corrosion. Water Res. 85, 275–285.

https://doi.org/10.1016/j.watres.2015.08.037

MedECC Network, 2019. Risks associated to climate and environmental changes in the

Mediterranean region, A preliminary assessment by the Mediterranean Experts on

Climate and Environmental Change (MedECC) Network Science-policy interface.

Meneses, M., Pasqualino, J.C., Castells, F., 2010. Environmental assessment of urban

wastewater reuse: Treatment alternatives and applications. Chemosphere 81, 266–272.

https://doi.org/10.1016/j.chemosphere.2010.05.053

Muñoz, I., Rodríguez, A., Rosal, R., Fernández-Alba, A.R., 2009. Life Cycle Assessment of

urban wastewater reuse with ozonation as tertiary treatment. A focus on toxicity-related

impacts. Sci. Total Environ. 407, 1245–1256.

https://doi.org/10.1016/j.scitotenv.2008.09.029

Pan, Y.R., Wang, X., Ren, Z.J., Hu, C., Liu, J., Butler, D., 2019. Characterization of

implementation limits and identification of optimization strategies for sustainable water

resource recovery through life cycle impact analysis. Environ. Int. 133, 105266.

643   https://doi.org/10.1016/j.envint.2019.105266

644   Pasqualino, J.C., Meneses, M., Castells, F., 2011. Life Cycle Assessment of Urban

645   Wastewater Reclamation and Reuse Alternatives. J. Ind. Ecol. 15, 49–63.

646   https://doi.org/10.1111/j.1530-9290.2010.00293.x

647   Pretel, R., Moñino, P., Robles, A., Ruano, M. V., Seco, A., Ferrer, J., 2016. Economic and

648   environmental sustainability of an AnMBR treating urban wastewater and organic

649   fraction of municipal solid waste. J. Environ. Manage. 179, 83–92.

650   https://doi.org/10.1016/j.jenvman.2016.04.057

651   Pretel, R., Robles, A., Ruano, M. V., Seco, A., Ferrer, J., 2013. Environmental impact of

652   submerged anaerobic MBR (SAnMBR) technology used to treat urban wastewater at

653   different temperatures. Bioresour. Technol. 149, 532–540.

654   https://doi.org/10.1016/j.biortech.2013.09.060

655   Rizzo, L., Gernjak, W., Krzeminski, P., Malato, S., McArdell, C.S., Perez, J.A.S., Schaar, H.,

656   Fatta-Kassinos, D., 2020. Best available technologies and treatment trains to address

657   current challenges in urban wastewater reuse for irrigation of crops in EU countries. Sci.

658   Total Environ. 710. https://doi.org/10.1016/j.scitotenv.2019.136312

659   Rizzo, L., Krätke, R., Linders, J., Scott, M., Vighi, M., de Voogt, P., 2018. Proposed EU

660   minimum quality requirements for water reuse in agricultural irrigation and aquifer

661   recharge: SCHEER scientific advice. Curr. Opin. Environ. Sci. Heal. 2, 7–11.

662   https://doi.org/10.1016/j.coesh.2017.12.004

663   Salgot, M., Folch, M., 2018. Wastewater treatment and water reuse. Curr. Opin. Environ. Sci.

664   Heal. 2, 64–74. https://doi.org/10.1016/j.coesh.2018.03.005

665   Tangsubkul, N., Beavis, P., Moore, S.J., Lundie, S., Waite, T.D., 2005. Life Cycle

666   Assessment of Water Recycling Technology. Water Resour. Manag. 19, 521–537.

667   https://doi.org/10.1007/s11269-005-5602-0

668   Turan, V., 2019. Confident performance of chitosan and pistachio shell biochar on reducing

669   Ni bioavailability in soil and plant plus improved the soil enzymatic activities,

670   antioxidant defense system and nutritional quality of lettuce. Ecotoxicol. Environ. Saf.

671   183, 109594. https://doi.org/10.1016/j.ecoenv.2019.109594

672   Turan, V., Khan, S.A., Mahmood-ur-Rahman, Iqbal, M., Ramzani, P.M.A., Fatima, M., 2018.

673   Promoting the productivity and quality of brinjal aligned with heavy metals

674   immobilization in a wastewater irrigated heavy metal polluted soil with biochar and

675   chitosan. Ecotoxicol. Environ. Saf. 161, 409–419.

676   https://doi.org/10.1016/j.ecoenv.2018.05.082

677   UNEP/MAP Plan Bleu, 2019. State of the environment and development in the

678   Mediterranean. Technical Report. Athens.

679   Ventura, D., Consoli, S., Barbagallo, S., Marzo, A., Vanella, D., Licciardello, F., Cirelli, G.L.,

680   2019. How to overcome barriers for wastewater agricultural reuse in Sicily (Italy)?

681   Water (Switzerland) 11, 1–12. https://doi.org/10.3390/w11020335

682   Vojtěchovská Šrámková, M., Diaz-Sosa, V., Wanner, J., 2018. Experimental verification of

683   tertiary treatment process in achieving effluent quality required by wastewater reuse

684   standards. J. Water Process Eng. 22, 41–45. https://doi.org/10.1016/j.jwpe.2018.01.003

685   Yoshida, H., ten Hoeve, M., Christensen, T.H., Bruun, S., Jensen, L.S., Scheutz, C., 2018.

686   Life cycle assessment of sewage sludge management options including long-term

687   impacts after land application. J. Clean. Prod. 174, 538–547.

688