

Modelling environmental impacts of treated municipal wastewater reuse for tree crops irrigation in the Mediterranean coastal region

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2 **MODELLING ENVIRONMENTAL IMPACTS OF TREATED MUNICIPAL**  
3 **WASTEWATER REUSE FOR TREE CROPS IRRIGATION IN THE MEDITERRANEAN**  
4 **COASTAL REGION**

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## **Abstract**

Wastewater reuse provides valuable solutions to solve the societal challenges of decreasing availability and limiting access to secure water resources. The present study quantifies the environmental performance of nectarine orchards irrigation using TMW and surface water using a unique dataset based on field experimental data. One open field nectarine cultivation orchard located in Apulia Region, South of Italy, has been selected as case study. Climate change, toxicity (for human and freshwater), eutrophication (marine and freshwater) and acidification impacts were analysed using the impact assessment method suggested by the International Reference Life Cycle Data System (ILCD). The water footprint associated to the life cycles of each system has been estimated using the Available Water REmaining (AWARE) method. Monte Carlo simulation was used to assess data uncertainty. The irrigation of nectarine orchards using TMW performs better than the irrigation using surface water for eutrophication impact categories. Compared with surface water resources, the potential impacts of TMW reuse in agriculture on climate change and toxicity are affected by the wastewater treatment phase (WWT). Only eutrophication and acidification burdens are generated by in-field substitution of surface water with TMW. Considering human and ecosystem water demand, the irrigation with TMW increases water consumption of 19.12 m<sup>3</sup> per kg of nectarine produced. Whereas, it shows a positive contribution to water stress (-0.19 m<sup>3</sup>) if only human water demand is considered. This study provides important results that allow for a better understanding of the potential environmental consequences of TMW reuse in agriculture. It suggests that embracing the type of WWTs, the replacement of fertilizers, the effects on water scarcity and ecosystem quality might be useful to redefine water reuse regulations and increase public acceptance for the reuse of TMW in agriculture. Moreover, this study reveals the need for developing consensus and standardized guidance for life cycle analysis of water reuse applications.

48

49 **Keywords:**

50 • Wastewater reuse;

51 • Surface water;

52 • Nectarine orchards production;

53 • Life cycle assessment;

54 • Water footprint;

55 • Water scarcity.

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## 58 **1 Introduction**

59 Continuous population growth and increasing urbanization are challenging secure water availability  
60 and wastewater disposal (Angelakis and Snyder, 2015). However, freshwater resources are mostly  
61 consumed in water scarce regions (Scherer and Pfister, 2016) and the exploitation of unsustainable  
62 water resources is predicted to increase in the future (Wada and Bierkens, 2014). Water reuse  
63 provides valuable solutions to solve the societal challenges linked to water scarcity and environmental  
64 pollution (Asano, 1994). The contribution of agriculture to total freshwater consumption can reach  
65 80% in Southern European Regions (EEA, 2009). This is partly on account of the unique climate and  
66 geographical conditions of the Mediterranean coast and because of freshwater shortage. The reuse of  
67 treated municipal wastewater (TMW) as an alternative water source in agriculture has been  
68 recommended in many studies (Bedbabis et al., 2015; Grattan et al., 2015; Vivaldi et al., 2013; Weber  
69 et al., 2014). However, the high variability of soil-climate conditions, the large number of wastewater  
70 treatment technologies and water management options, make the estimation of the environmental  
71 impacts of water reuse systems complicated, mostly in regions where agriculture needs a large amount  
72 of water.

73 Recognizing that Apulia region has the smallest amount of available water resources (136  
74 m<sup>3</sup>/capita/year) and the lowest rainfall average value (i.e. about 660 mm/y) in Italy (Lopez et al.  
75 2010), water reuse becomes a promising alternative for conserving and extending available water  
76 sources. Despite a large amount of TMW that is locally available (more than 100 million m<sup>3</sup>/year),  
77 only 5 million m<sup>3</sup> are recovered yearly (Arborea et al. 2017), while the remainings are dumped into  
78 the sea. The lack of wastewater reuse is due to inadequate infrastructures for storage and distribution,  
79 restrictive legislation on reuse of wastewater, and scarce public acceptance fostered by inadequate  
80 information on the environmental impact of this alternative water source. Spain is the leading country  
81 on life cycle studies of water reuse in agriculture. Rodriguez-Garcia et al. (2011) analysed the life  
82 cycle eutrophication and global warming potentials (GWP) of six different wastewater treatment  
83 plants and associated water reuse applications (agriculture, industrial and aquifer recharge). They

84 found the performance of TMW reuse in agriculture as worse compared with the other alternatives in  
85 terms of eutrophication, while TMW resulted as beneficial compared with acquirer recharge for  
86 Global Warming Potential (GWP). While analysing different tertiary treatments and wastewater reuse  
87 scenarios, Meneses et al. (2010) compared agriculture with landscape irrigation wastewater reuse,  
88 finding nutrient replacement as the main driver for the more beneficial outcomes of the former.  
89 Amores et al. (2013) compared the environmental profile of the current water cycle in Tarragona  
90 (Spain), with alternative scenarios where tertiary treatment and water reuse in agriculture were added  
91 to the current system. Reuse showed to be beneficial only for water use impact which was reduced  
92 when TMW is reused. However, these studies relied on previous studies to estimate of the  
93 environmental consequences of water reuse for non-potable applications. To the authors' knowledge,  
94 only Muñoz et al. (2009)) used experimental data to analyse the environmental impacts of TMW  
95 reuse. They found that irrigation with TMW performed better than using desalinated water or  
96 groundwater, in terms of eutrophication, aquatic ecotoxicity and energy use. To establish a scientific  
97 basis to evaluate environmental performances of TMW reuse for fruit orchards irrigation, the  
98 environmental impacts of nectarine grove irrigation with TMW and surface water in southern Italy are  
99 compared based on a unique experimental dataset.

## 100 **2 Materials and method**

### 101 *2.1 General description of the field experiment.*

102 An experimental trial was carried out in Trinitapoli in a commercial nectarine grove (Apulia Region,  
103 Southern Italy, 41°22'.92" N; 16°03'16.27" E; Altitude 1m). Two different water sources were  
104 compared for the irrigation of the nectarine orchard: treated municipal wastewater (TMW), made by a  
105 membrane filtration public plant located near the experimental site, and surface water (SW) that  
106 represents the conventional water used in this area. Acknowledging the high variability of water  
107 quality observed during a previous water monitoring campaign (data not published), the water  
108 fertilization contribution was not examined and the experiment was not set up to maximize fertilisers  
109 efficiency. Therefore, the same amount of fertilizers (102, 27, 0 kg ha<sup>-1</sup> of N<sub>2</sub>-P<sub>2</sub>O<sub>5</sub>-K<sub>2</sub>O, respectively)  
110 were applied to both treatments. Moreover, as reported in Pedrero et al. (2018), the irrigation volume

111 was calculated by the water balance method with restitution of 100% crop evapotranspiration<sup>1</sup> (ET<sub>c</sub>)  
112 lost in each irrigation interval, and the same amount of water was applied to both treatments. The  
113 yearly average (2012–2014) yield resulted higher in the SW (18,444 kg × ha<sup>-1</sup>) compared with the  
114 TMW (17, 932 kg ha<sup>-1</sup>) treatment. Additional information about the experimental trial, irrigation and  
115 fertilization treatments are reported in Vivaldi et al. (2015) and provided in the supplementary  
116 materials (S1). Descriptive statistics of the physical and chemical characteristics of soil and water  
117 types are provided in the supplementary materials (Table S1 and Table S2).

## 118 2.2 Orchard modelling and boundaries definition.

119 Two cradle-to-farm gate scenarios were modelled for the irrigation of nectarine orchards using TMW  
120 (Scenario 1) and surface water (Scenario 2). In Scenario 1, the agricultural production process  
121 interacted with other product systems: (i) the secondary wastewater treatment (hereinafter referred to  
122 as WWT), generating an open loop recycling, and (ii) the water system by avoiding freshwater  
123 withdrawals. Along with the decrease in freshwater withdrawals, the reuse of TMW rise the same  
124 issue of multi-functionality as in the case of systemes producing more than one product (co-  
125 production) (Schrijvers et al., 2016). However, if a recycled material is used as input for a subsequent  
126 life cycle the allocation of the impacts should be tackled with the same procedures as for co-products  
127 (Guinée, 2002). As described in Schrijvers et al. (2016), the “cut-off” approach could be use to  
128 include treated wastewater in the analysis. However, Pradel et al. (2016) showed that this procedure  
129 holds only if treated wastewater is considered a final waste. In the study, the wastewater is considered  
130 a product containing nutrients and minerals, which directly affect agricultural production. Therefore,  
131 the system boundaries in Sceanrio 1 include the whole treated wastewater life cycle (Pradel et al.,  
132 2016). As suggested by the ISO standards 14044:2006 (ISO, 2006), the multi-functionality issue is  
133 addressed in this study by using system expansion, which guarantees the comparability between

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<sup>1</sup> The crop evapotranspiration (ET<sub>c</sub>) was calculated using the equation recommended by the FAO:  $ET_c = K_r \times K_c \times ET_0$ , where  $K_r$  is the reduction coefficient ( $K_r = 0.75$ ),  $K_c$  (0.50  $K_{cini}$ , 1.15  $K_{cmid}$ , 0.85  $K_{cend}$ ) is the crop coefficient, and  $ET_0$  is reference evapotranspiration.

134 multi-dimensional (Scenario 1) and mono-dimensional (Scenario 2) systems (Schrijvers et al., 2016;  
135 Weidema, 2001). In system expansion, the functional unit (hereinafter referred to as FU) must be  
136 enlarged to include the function or service provided by the recycled product and the co-product  
137 (Hauschild et al., 2013). Therefore, Scenario 1 has been expanded by including the impacts generated  
138 by the avoided freshwater withdrawals and the discharging of TMW into the sea (*green boxes in*  
139 *Figure 1*). All agricultural processes (e.g. sowing, pruning, harvesting) were managed using the same  
140 equipment and machineries in the experimental field. Therefore, no machineries and equipment, fuels,  
141 or energy consumptions were accounted in the studied systems. However, fertilizers and pesticides  
142 production was incorporated in the system boundaries due to their impact in agricultural LCA  
143 (Brentrup et al., 2004; Margni et al., 2002). The application of fertilizers was carried out using the  
144 same equipment in both scenarios. Previous studies reported negligible environmental impact of the  
145 construction, dismantling, of a WWT plant compared to the operational phase (Lorenzo-Toja et al.,  
146 2016a; Muñoz and Fernández-Alba, 2008; Raluy et al., 2006). Moreover, water storage and  
147 distribution infrastructures are large-scale infrastructures managing large amounts of water and  
148 serving not only the agricultural sector, and they were thus expected to have negligible impacts on the  
149 environmental profile of the studied systems. Therefore, wastewater treatment plant, surface water  
150 storage infrastructures and the distribution network (e.g. dam, pumps and water network) were  
151 excluded from the analysis.

152 *Figure 1: Material flows diagrams for nectarine field irrigation with TMW and surface water*

153 The limitations arising from using a mass-based functional unit (FU) have been reported by several  
154 studies (Milà i Canals et al., 2006; Mouron et al., 2012; van der Werf et al., 2007) In this regard,  
155 Cerutti et al. (2014) recommend the use of both mass-based and land-based FU to avoid the  
156 overvaluation of the more resource efficient farming system. However, the experimental trial was set-  
157 up to maximize yield. Therefore, a mass-based FU (one kg of nectarines produced) reflecting a  
158 product-oriented expression of the agricultural systems (Hayashi, 2013) was used in this study.



159 2.3 *Modelling assumptions and parameters specification*

160 Field operations data, including the water inventory, were directly measured during the three years of  
161 the experimental trial (2012 - 2014). Background data for fertilizers production was provided by the  
162 ecoinvent v3 database (Wernet et al., 2016). Although Lorenzo-Toja et al. (2016b) reported  
163 considerable differences in terms of eutrophication net environmental impact and GWP according to  
164 the size of the WWT plant, materials flows, energy consumption and operational data for the WWT  
165 plant were modelled adopting an average WWT plant in the Mediterranean countries serving 144 000  
166 population equivalent as defined by Amores et al. (2013) and Pasqualino et al. (2009). Data and  
167 assumptions on the WWT processes were presented in the supplementary materials (S2).

168 *Table 1. Inputs-outputs inventory for Scenario 1 and 2 (per kg of nectarine produced).*

169 All the emissions generated by fertilizers and water application have been accounted for. Ammonia,  
170 nitrate and nitrous oxides emissions were estimated according to Emmenegger et al. (2009). To  
171 accommodate the model specification, the average nitrogen uptake by nectarine trees reported by  
172 Johnson and Urui (1989) has been assumed. The SALCA – Phosphorus model (Prasuhn, 2006),  
173 including the modification proposed by Emmenegger et al. (2009) was used to estimate phosphorus  
174 emissions evaluating only run-off and leaching risks. According to the model, phosphorus emissions  
175 from fertilizers application depend on the slope of the field. The latter was 0% in the studied orchards,  
176 thus, only phosphorus leaching due to soil erosion has been accounted in the model. Field emissions  
177 from pesticide were estimated using PestLCI 2.0 model (Dijkman et al., 2012) and the approach  
178 suggested by Margni et al. (2002). Both irrigation water types included considerable concentration of  
179 macro and micronutrients (Table S1). However, the experimental trials in this study were not arranged  
180 to capture detailed data on plant uptakes and nutrients mobilisations at the soil-plant interface.  
181 Therefore, calcium, sulfur, magnesium and the other micronutrients in both water types have been  
182 accounted as indirect emissions directly reaching soil and water bodies.

183 2.4 *Impact assessment*

184 To address the lack of standardized impact categories in LCA, the International Reference Life-Cycle  
185 Data System (ILCD) recommendations (European Commission - Joint Research, 2011) were used in

186 this study to define the best-suited methods for characterizing the impact categories at the midpoint  
187 level. Moreover, Bessou et al. (2013) and Cerutti et al. (2014) found global warming, eutrophication  
188 and toxicity impacts as the most affected by agriculture production. Accordingly, the following  
189 impact categories were quantified: Climate change (CC); Human toxicity (HT); Acidification (AC);  
190 Freshwater and marine eutrophication (FE, ME) and Freshwater ecotoxicity (FEc). The impacts for  
191 these categories were assessed using the IPCC 2007 GWP at 100 years (Ipcc, 2014), USEtox  
192 (Rosenbaum et al., 2008), and Accumulated Exceedance (Seppälä et al., 2006) methods, respectively.  
193 The rationale behind the selection of impact categories lay in the fact that the chosen methods are  
194 scientifically robust for assessing water degradation categories (Hauschild et al., 2013). In LCA, the  
195 direct impacts of water consumption and increasing water competition among different uses were  
196 recently addressed (Boulay et al., 2015b). Starting from the general framework proposed by Bayart et  
197 al. (2010), several methods addressing these issues were developed. Most of the LCA methods  
198 focused on water use impact (midpoint) while some addressed also the potential effects on human  
199 health and ecosystem quality (endpoint). At the midpoint level, the existing methods quantify water  
200 scarcity based on a use-to-availability ratio, which was referred to as a water stress index (Boulay et  
201 al., 2015b). The use-to-availability ratio approach assumes different theoretical and practical forms  
202 across the different methods. Withdrawal-to-availability (WTA) and consumption-to-availability  
203 (CTA) approaches only account for human water use, failing to examine ecosystem water  
204 requirements (Boulay et al., 2015b; Kounina et al., 2013). A water consumption pathway leading to  
205 an indicator, which encompassed potential deprivation of both ecosystem and human, was developed  
206 by including the demand-to-availability approach (DTA) in the water use characterization pathway  
207 (Boulay et al., 2015a). Among the three approaches proposed by the WaterLCA working group of the  
208 UNEP-SETAC Life Cycle Initiative (Boulay et al., 2017; Boulay et al., 2015a), the consensus was to  
209 use the inverse of water availability minus the water demand approach (1/AMD) and the Available  
210 Water REMaining (AWARE) method. The AWARE method was used in this study to represent the  
211 impact of scenarios 1 and 2 on water use. This method builds on the assumption that the potential to  
212 deprive another user of water is directly proportional to the water consumed and inversely

213 proportional to the available water remaining per unit of surface and time in a region (Boulay et al.,  
214 2018). The sensitivity of the modelled systems to the AWARE method assumptions was analysed by  
215 comparing the water use impact with the Water Stress Indicator (hereinafter WSI) proposed by  
216 Boulay et al. (2011). The WSI is the result of a CTA-based model for assessing the potential impacts  
217 from water use, which accounts only for human water consumption in the definitions of the water  
218 stress characterization factors (CFs) (Boulay et al., 2011). Since LCI databases only provide unit  
219 process for water production with global or country annual average resolution (Quinteiro et al., 2017),  
220 country level CFs were used in this study to guarantee the adequate connection with the inventory  
221 flows. For the life cycle impact assessment SimaPro 8.4 software (PRé Consultants, 2017) was used.  
222 The uncertainty arising from the variability in the experimental data was assed using Monte Carlo  
223 simulation (10,000 trials) with all input-output parmeters varying according the assumed distributions  
224 (Table 1).

## 225 **3 Results**

### 226 *3.1 Life cycle impact assessment (LCIA)*

227 Results are expressed at midpoint level, which translates impacts into the six selected environmental  
228 themes to identify key differences between nectarine orchards irrigation with TMW or surface water.  
229 Table 2 reports the LCIA outcomes, which are expressed in units per kg of nectarines produced at the  
230 farm gate. Scenario 1 performs worse than Scenario 2 for CC, HT, AC, FE<sub>c</sub>, while it achieves better  
231 outcomes for FE and ME. The irrigation with TMW (Scenario 1) generates lower eutrophication  
232 impacts (net environmental benefit for ME) compared with surface water (Scenario 2) due to the  
233 avoided discharging of TMW into the sea. The differences between the two scenarios are less  
234 emphasised for CC, HT and FE<sub>c</sub>, with Scenario 1 performing worse than Scenario 2. The differences  
235 range from around 9% to 25% for the above mentioned impact categories and they are mainly  
236 attributable to WWTs.

237 *Table 2: Characterization of the environmental impacts for the modelled scenarios. (FU = 1 kg of*  
238 *nectarines)[Standard Deviation]*

239 The cultivation phase is the main phase responsible for the impacts on AC and FE (Table 3) in both  
240 scenarios. Accounting for the more than the 70% of the whole impact for CC and toxicity (both for  
241 humans and freshwater), the WWT phase is the main driver of the negative outcomes of Scenario 1  
242 for these environmental categories. Electricity consumptions and the incineration of sewage sludge  
243 are the main contributor to the environmental impacts of the WWT phase.

244 *Table 3: Process contributions to the midpoint environmental impacts of Scenario 1 and Scenario 2.*

245 Figure 2 shows the environmental impacts of Scenario 1 and Scenario 2 normalised according to  
246 Benini et al.(2014). The category with the highest normalized impact is FEc (82% and 80%  
247 respectively) in both scenarios. CC accounts for almost 20% and 15% in both scenarios, respectively.  
248 AC and eutrophication impacts categories show only marginal effects on the environmental  
249 performance of Scenario 1 (lower than 1%), while AC and FE contribute for 4% and 2% respectively  
250 to the normalized impact of Scenario 2. Electricity consumption and the incineration of sewage sludge  
251 in the WWT phase are the main contributors to the environmental performance of Scenario 1 for CC,  
252 HT and FEc, while the outcomes of Scenario 2 for these impact categories are mainly driven by  
253 fertilizers and pesticides production. In both scenarios, the impacts on AC and eutrophication (both FE  
254 and ME) are mainly generated by the field operations.

255 *Figure 2: Normalized processes contribution to environmental impacts of Scenario 1 and Scenario 2.*

256 *(FU = 1 kg of nectarines)*

### 257 3.2 Water footprint

258 Substituting surface water (Scenario 2) with TMW (Scenario 1) generates an almost 10 m<sup>3</sup> increase in  
259 water consumption per kg of nectarine produced (Table 4). The WWT is the main contributor to the  
260 water use impact of Scenario 1 (around 27 m<sup>3</sup>). The contribution of fertilizers and pesticides  
261 production to the water use impact of Scenario 1 is almost three order of magnitude lower compared  
262 with WWT. Whereas, field irrigation with TMW generates net benefit arising from the avoided  
263 freshwater withdrawals (-9.07 m<sup>3</sup>). The overall water use impact of Scenario 2 resulted in almost 10  
264 m<sup>3</sup>, of which only 1.33 m<sup>3</sup> are associated with fertilisers and pesticides production. Field operations  
265 contributes for more than 90% to the burden associated to nectarine orchard irrigation with surface

266 water (8.8 m<sup>3</sup>). According to the WSI method, Scenario 1 shows a positive contribution to water stress  
267 (-0.19 m<sup>3</sup>) which is attributable to the avoided surface water consumption for irrigation (Table 4). The  
268 WWT results in a water stress impact almost five order of magnitude lower compared with the water  
269 use impact. Scenario 2 shows to be detrimental (0.19 m<sup>3</sup>), with surface water withdrawals for field  
270 irrigation accounting for more than the 90% of the overall water stress impact. For both scenarios, the  
271 water stress impact of fertilizers and pesticides production is three order of magnitude lower than field  
272 operation.

273 *Table 4: Water footprint and Water stress for both scenarios 1 and 2. (FU = 1 kg of nectarines)*  
274 *[Standard Deviation]*

275 Furthermore, the comparison of CC and water use performances of both modelled systems with  
276 different scenarios, where the energy production mix used in the WWT process has been changed,  
277 shows that the climate change and water use performances of Scenario 1 could differ substantially  
278 according to the energy production mix used in the WWT phase (Figure 4).

## 279 **4 Discussion**

280 The substitution of surface water with TMW for nectarine orchards irrigation generates environmental  
281 trade-offs. The irrigation of nectarine orchards using TMW (Scenario 1) performs better than the  
282 irrigation with surface water (Scenario 2) for eutrophication impacts categories. It performs worse for  
283 CC, AC, toxicity (both for humans and freshwater bodies), and water use. T-test statistics report  
284 significant differences between Scenario 1 and 2 for all impact categories (p-values < .001). The  
285 probability of Scenario 1 performing better than Scenario 2 is null for CC, AC and water use, it equals  
286 100% for ME, and it varies within a range from 14% to 72% for HT, FEc, and FE respectively (Figure  
287 3).

288 *Figure 3: Probabilities density functions of the differences between Scenario 1 and Scenario 2 for HT,*  
289 *FEc, FE and water use impact categories (FU = 1 kg of nectarines).*

290 Although applying different assumptions on system boundaries, scales and water reuse scenarios,  
291 several studies found the reuse of TMW more beneficial for the environment than desalinated water,  
292 conventional potable water and groundwater (Meneses et al., 2010; Muñoz et al., 2009; Pasqualino et

293 al., 2011), but none of them contemplated surface water. TMW reuse for irrigation provides several  
294 direct benefits to water resources and crop productivity (Angelakis et al., 2003; Asano and Levine,  
295 1996; Pedrero et al., 2010). Among these, nutrients recycling through the direct use of TMW for crop  
296 irrigation is beneficial for resources depletion and indirectly decreases the energy and water intensity  
297 of agricultural production (Mo and Zhang, 2013). However, the contribution of the fertilizers  
298 production phase results at least one order of magnitude lower compared with WWTs for all impact  
299 categories. Therefore, even a complete replacement of fertilizers with the nutrients provided with  
300 TMW is expected to provide only marginal benefits on the potential environmental impact of  
301 Scenario 1. A large share of the impacts generated in the cultivation phase originate in the use of  
302 fertilizers and pesticides. The excess chemicals, macro and micronutrients accumulate into soils, leach  
303 into groundwater and pollute aquatic ecosystems (Tuomisto et al., 2012). Moreover, in Scenario 1, the  
304 avoided discharge of TMW into the sea generates net environmental benefits for ME, which are  
305 attributed to the cultivation phase. Besides AC and eutrophication impacts, which are driven by  
306 nutrients accumulation and leaching into soil and water bodies respectively, the WWT stage in  
307 Scenario 1 resulted as the main contributor to the performance of Scenario 1 for water use, CC and  
308 toxicity impacts (Table 3). Energy production accounts for more than 65% and 90% of the impacts of  
309 the WWT process on CC and water use respectively (Table S4). Although, the high energy  
310 consumption of WWTs operations and their indirect contribution to GHG emissions and human  
311 toxicity have been documented by several studies (Amores et al., 2013; Lorenzo-Toja et al., 2016a;  
312 Rodriguez-Garcia et al., 2011), the implication on water availability is unclear. In this regard, the  
313 Italian energy production mix strongly relies on hydropower generation mix (around 20%, (Eurostat,  
314 2017). The use of surface water for electricity generation prevents water resources to be allocated to  
315 other uses (drinking water, irrigation, etc.). According to the AWARE model specifications, this  
316 results in less water available after the demand of humans and ecosystems has been met, which can  
317 explain the estimated impact of WWT on water use. Therefore, changing the energy production mix  
318 towards less water dependent generation pathways could potentially modify the overall outcomes of  
319 the study.

320 *Figure 4 Water use and climate change impacts of Scenario 1 using different energy production*  
321 *schemes in the WWT process.*[The impact scores estimated for the different alternative energy  
322 production mixes in the WWT process are normalized using the impacts Scenario 1 as baseline] (*FU*  
323 *= 1 kg of nectarines*)

324 Although the use of water-use-intensive energy generation sources, such as hydropower and nuclear,  
325 generate beneficial effects on climate change impacts, they cause worse potential impacts on water  
326 use compared with the baseline Scenario 1 (Figure 4). The inverse trade-off is achieved when “water-  
327 free” generation technologies as biomass and fossil fuels are included in the energy production mix,  
328 reducing the burden on water resources while charging the system with higher potential impacts on  
329 climate change. Beneficial effects for both environmental impact indicators could result only if more  
330 than the 20% of both “carbon and water-free” energy generation technologies (e.g. wind and solar)  
331 provide the electricity used in the WWT process in Scenario 1.

332 The model chosen to assess the minimum water requirements to maintain freshwater ecosystems  
333 brings a high degree of uncertainty in the AWARE CFs (Boulay et al. 2017; Damiani et al. 2018). The  
334 freshwater ecosystem demand is defined according the method proposed by Pastor et al. (2014),  
335 which sets the 30-60 % of mean monthly flow as degradation limits for maintaining the freshwater  
336 ecosystem in a fair ecological condition. These water flow dynamics require a level of temporal  
337 resolution which is beyond the impact assessment capabilities most impact assessment methods.

338 The sensitivity of the AWARE model to this assumption has been tested by comparing water use  
339 impacts with the WSI of both scenarios respectively. The different impacts on water resources  
340 resulting from the two models are a result of the different underlying approaches used to define the  
341 CFs: CTA and DTA, respectively. The former considers local water scarcity as being affected only by  
342 the water that is actually allocated to human activities, while the latter contemplates both ecosystem  
343 water demand and human consumption as factors affecting local water availability. According to the  
344 WSI method, the water used for hydropower generation in the WWT stage, which is then returned to  
345 freshwater bodies, does not affect local water availability. In contrast, the AWARE model

346 specification regards the water allocated to hydropower generation as unavailable for freshwater  
347 ecosystems and therefore reducing local water availability.

348 This study sought to estimate the environmental impacts of field application of TMW as substitute of  
349 surface water for trees irrigation. Therefore it used data from Amores et al. (2013) to model the  
350 background WWT process, the quality and amount of water influent, the technology used from a  
351 WWT plant located in the Mediterranean region. Although Meneses et al. (2010) reported similar  
352 environmental profiles for different wastewater treatment technologies and wastewater treatment  
353 plants located in the Mediterranean region, this assumption increases the uncertainty of the outcomes,  
354 and might led to unrealistic scenarios especially in relation to water (Risch et al., 2014) and energy  
355 (Baresel et al., 2015) use in the WWT process. Unlike regionalized LCA water stress CFs, the higher  
356 spatial resolution ones (global or country level) might generate results bias when targeting obvious  
357 local environmental issues such as water scarcity (Bai et al., 2017). However, Quinteiro et al. (2017)  
358 and Yang et al. (2017) underlined the need to establish a consistent link between local CFs and the  
359 related spatial inventory to better recognise the relevance of the impact results on local water use. The  
360 latter remains a major challenge in LCA (Yang, 2016; Yang et al., 2017). Since water inflows are  
361 unevenly distributed through Italy (Scarascia et al., 2006), the use of national LCI databases for  
362 background processes and country level CFs might have led to underestimate or overestimate the  
363 water use impacts generated by surface water use for irrigation purposes in the Southern regions.

## 364 **5 Conclusion**

365 This study sought to contribute to the water recycling literature by measuring the environmental  
366 profile of fruit orchard irrigation with TMW and by comparing it with the use of surface water. The  
367 use of TMW achieved better outcomes for eutrophication related environmental burdens, while it  
368 performed worse for climate change, toxicity (humans and freshwater), acidification and water use.

369 Although the input-output flows are directly related to the specific location and characteristics of the  
370 experimental field, the study has identified areas of potential beneficial contribution and the main  
371 potential sources of environmental harm of TMW reuse in agriculture. Compared with conventional  
372 water resources, the potential environmental impacts of TMW reuse in agriculture are affected by the



373 WWT phase. This is true for the potential impacts on climate and both humans and freshwater  
374 toxicity. In-field substitution of surface water with TMW generates both environmental burdens, for  
375 the potential impacts on acidification and freshwater eutrophication, and benefits for the potential  
376 impacts on marine eutrophication.

377 Moreover, the study shows that the assumptions embedded in the water footprint characterization  
378 models have a crucial role in defining the water footprint of TMW reuse in agriculture. This issue,  
379 together with the development of a consistent local inventory, deserves particular attention and should  
380 be the matter of future research for reducing the uncertainty of water use impacts on local water  
381 scarcity. This study provided important results that allow for a better understanding of the potential  
382 environmental consequences of TMW reuse in agriculture. The existing regulations on the reuse of  
383 TMW for irrigation distinguish the use or non-use options based on the microbiological quality of the  
384 effluents and the type of irrigated crops. The outcomes of this study suggest that a broader  
385 environmental perspective, embracing the WWTs technologies, the replacement of fertilizers, the  
386 effects on water availability and ecosystem quality might be useful to redefine these regulations and  
387 increase public acceptance for the reuse of TMW in agriculture.

388 Moreover, it reveals the need for developing consensus and standardized guidance for life cycle  
389 analysis of water reuse applications.

390

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540 **Tables**

541 **Table 1: Inputs-outputs inventory for Scenario 1 (waste water field irrigation) and 2 (surface**  
 542 **water field irrigation).** (*FU = 1 kg of nectarines*)

Input/output	Average Nominal Range [min; max]				Units	Distribution
	Scenario 1		Scenario 2			
<i>Field operation</i>	-					
Water	-	-	0.190	[0.68; 0.21]	m <sup>3</sup>	Triangular
Ammonium sulphate	0.005	-	0.005	-	kg	-
Phosphate fertilizer	0.001	-	0.001	-	kg	-
Treated wastewater	0.202	[0.17; 0.22]	0.196	[0.16; 0.21]	m <sup>3</sup>	Triangular
Pyridate <sup>b</sup>	0.043	[0.039; 0.046 ]	0.042	[0.038; 0.044]	g	Triangular
Tebuconazole <sup>b</sup>	0.009	[0.006; 0.11]	0.009	[0.006; 0.11]	g	Triangular
Thiram <sup>b</sup>	0.595	[0.44; 0.69]	0.578	[0.43; 0.67]	g	Triangular
Benzoate <sup>b</sup>	0.455	[0.39; 0.59]	0.443	[0.38; 0.57]	g	Triangular
White mineral oil <sup>b</sup>	1.56	[1.17; 2.34]	1.52	[1.14; 2.28]	g	Triangular
<i>Emissions to air</i>						
Ammonia (fertilizers)	0.530	-	0.530	-	g	-
Ammonia	1.400	[SD <sup>2</sup> : 1.19]	1.010	[SD <sup>2</sup> : 1.04]	g	Lognormal
Nitrogen oxides	1.740	[SD <sup>2</sup> : 1.74]	1.080	[SD <sup>2</sup> : 1.00]	g	Lognormal
Pyridate <sup>b</sup>	0.34	-	0.33	-	mg	-
Tebuconazole <sup>b</sup>	0.23	-	0.23	-	mg	-
Thiram <sup>b</sup>	2.16	-	2.11	-	mg	-
Benzoate <sup>b</sup>	0.18	-	0.18	-	mg	-
White mineral oil <sup>b</sup>	0.16	-	0.15	-	g	-
<i>Emissions to water</i>						

Nitrate (freshwater)	1.180	[SD <sup>2</sup> : 1.00]	2.140	[SD <sup>2</sup> : 1.00]	g	Lognormal
Phosphorous	2.310	[SD <sup>2</sup> : 2.31]	2.520	[SD <sup>2</sup> : 2.52]	g	Lognormal
Pyridate <sup>b</sup>	0.003	-	0.003	-	mg	-
Tebuconazole <sup>b</sup>	0.29	-	0.28	-	mg	-
Thiram <sup>b</sup>	9.85E-5	-	9.57E-5	-	mg	-
Benzoate <sup>b</sup>	16.37	-	15.92	-	mg	-
White mineral oil <sup>b</sup>	0.13	-	0.13	-	g	-
Nitrate <sup>a</sup>	-1.280	[SD <sup>2</sup> : 1.54]	-	-	g	Lognormal
Phosphate <sup>a</sup>	-2.780	[SD <sup>2</sup> : 5.15]	-	-	g	Lognormal
Ammonium <sup>a</sup>	-8.180	[SD <sup>2</sup> : 1.89]	-	-	g	Lognormal
Fluoride <sup>a</sup>	-1.290	[SD <sup>2</sup> : 1.52]	-	-	g	Lognormal
Chloride <sup>a</sup>	-29.350	[SD <sup>2</sup> : 4.20]	-	-	g	Lognormal
Sulphate <sup>a</sup>	-19.490	[SD <sup>2</sup> : 1.75]	-	-	g	Lognormal
Sodium <sup>a</sup>	-8.180	[SD <sup>2</sup> : 1.89]	-	-	g	Lognormal
Potassium <sup>a</sup>	-5.360	[SD <sup>2</sup> : 1.71]	-	-	g	Lognormal
Calcium <sup>a</sup>	-17.720	[SD <sup>2</sup> : 3.36]	-	-	g	Lognormal
Magnesium <sup>a</sup>	-4.080	[SD <sup>2</sup> : 2.09]	-	-	g	Lognormal
<i>Emissions to soil</i>						-
White mineral oil <sup>b</sup>	1.19	-	1.16	-	g	-

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<sup>a</sup> Avoided emission into the sea

<sup>b</sup> Active principle

**Table 2: Characterization of the environmental impacts for the modelled scenarios. ( $FU = 1$  kg of nectarines) [Standard Deviation]**

	<b>Unit</b>	<b>Treated municipal wastewater (S1)</b>		<b>Surface water (S2)</b>	
Climate change	kg CO <sub>2</sub> eq	2.29E-01	[SD: 0.03]	2.18E-02	[SD: 2.56E-3]
Human toxicity	CTUh	3.02E-08	[SD: 2.08E-8]	7.84E-09	[SD: 4.11E-9]
Acidification	molc H <sup>+</sup> eq	7.17E-03	[SD: 3.94E-4]	5.02E-03	[SD: 7.86E-5]
Freshwater eutrophication	kg P eq	1.47E-03	[SD: 1.27E-3]	2.51E-03	[SD: 1.19E-3]
Marine eutrophication	kg N eq	-6.41E-03	[SD: 2.20E-3]	6.55E-04	[SD: 3.17E-6]
Freshwater ecotoxicity	CTUe	1.04E+00	[SD: 1.32]	1.22E-01	[SD: 2.45E-2]

**Table 3: Process contributions to the midpoint environmental impacts of Scenario 1 and Scenario 2.**

		Processes contribution						
	Unit	Treated municipal wastewater (S1)				Surface water (S2)		
		WWT <sup>a</sup>	Fertilisers	Pesticides	Field operation	Fertilisers	Pesticides	Field operation
Climate change	kg CO <sub>2</sub> eq	2.09E-01	1.11E-02	1.06E-02	-	1.08E-02	1.03E-02	-
Human toxicity	CTUh	2.23E-08	2.26E-09	5.75E-09	8.01E-12	2.15E-09	5.60E-09	7.78E-12
Acidification	molc H+ eq	8.61E-04	8.92E-05	2.20E-04	6.00E-03	2.97E-10	2.14E-04	4.72E-03
Freshwater eutrophication	kg P eq	6.13E-05	3.94E-06	3.94E-06	1.39E-03	3.75E-06	3.83E-06	2.52E-03
Marine eutrophication	kg N eq	1.67E-04	8.27E-06	2.15E-05	-6.58E-03	8.59E-05	2.09E-05	6.25E-04
Freshwater ecotoxicity	CTUe	9.19E-01	3.56E-02	5.45E-02	3.27E-02	3.38E-02	5.30E-02	3.17E-02

<sup>a</sup> Adapted form (Amores et al., 2013) according to the specific case study (Supplementary Materials S2).



**Table 4: Water footprint and Water stress for both Scenarios 1 and 2. (FU = 1 kg of nectarines) [Standard Deviation]**

Units		Reclaimed wastewater (S1)			Surface water (S2)				
		Average	SD	Process contribution	Average	SD	Process contribution		
Water use	m <sup>3</sup>	19.12	[3.31]	<i>Field operation</i>	-9.07E+00	10.06	[0.50]	<i>Field operation</i>	8.80E+00
				<i>Fertilizers</i>	7.22E-03			<i>Fertilizers</i>	6.76E-03
				<i>Pesticides</i>	1.36E+00			<i>Pesticides</i>	1.32E+00
				<i>WWT</i>	2.71E+01				
Water stress	m <sup>3</sup>	-0.19	[0.02]	<i>Field operation</i>	-1.99E-01	0.19	[0.009]	<i>Field operation</i>	1.94E-01
				<i>Fertilizers</i>	1.14E-04			<i>Fertilizers</i>	1.06E-04
				<i>Pesticides</i>	4.18E-05			<i>Pesticides</i>	4.07E-05
				<i>WWT</i>	2.93E-04				

Figure captions

**Figure 1: Material flows diagrams for nectarine field irrigation with treated wastewater and surface water.** (*FU = 1 kg of nectarines*)

**Figure 2: Normalized processes contribution to environmental impacts of Scenario 1 (waste water field irrigation) and Scenario 2 (surface water field irrigation).** (*FU = 1 kg of nectarines*)

**Figure 3: Probabilities density functions of the differences between Scenario 1 and Scenario 2 for HT, FEc., FE and water use impact categories.** (*FU = 1 kg of nectarines*)

**Figure 4: Water use and climate change impacts of Scenario 1 (waste water field irrigation) using different energy production schemes in the WWT process.** [The impact scores estimated for the different alternatives energy production mix in the WWT process are normalized using the impacts Scenario 1 as baseline] (*FU = 1 kg of nectarines*)