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Research article

Insights into the environmental performance of nature-based wastewater technologies towards water and carbon neutrality

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ABSTRACT

Water reuse has become a promising strategy to address water scarcity and reduce pressure on natural water resources, particularly for irrigation practices. However, the implementation of water reuse for irrigation requires careful planning to address potential challenges, such as water quality concerns, public acceptance and the environmental impacts of treatment processes. To support informed decision-making, Life Cycle Assessment (LCA) serves as a powerful tool for evaluating the environmental implications of different water reuse strategies.

This study presents a comparative analysis of several biological treatment systems across different regions of the Iberian Peninsula, including membrane bioreactors, filtration systems, constructed wetlands, and advanced oxidation processes. In addition to assessing the environmental performance of these technologies, this research work also links the LCA outcomes with the concepts of water and carbon neutrality in wastewater treatment plants.

The results indicate that, among the four processes, only one achieves a positive water balance: the bio-filter system, which is also the best performing technology in most LCA categories. In contrast, the membrane bioreactor combined with integrated fixed film activated sludge shows worse performance, largely due to its high energy consumption. Regarding carbon neutrality, operating constructed wetlands requires approximately 0.79 ha of cropland to offset greenhouse gas emissions. This is significantly lower than the estimated 16.7–566 ha needed for non-nature-based solutions, which are not designed for carbon capture.

1. Introduction

Water scarcity, intensified by climate change, rapid urbanization, and aquifer overexploitation, poses a significant threat to agricultural sustainability (Karimi et al., 2024; Vörö;smarty et al., 2000). Agriculture is undoubtedly the most water-intensive sector considering that the global agricultural area is approximately 1.6 billion hectares, 17 % of this land under irrigation systems (Ingrao et al., 2023). However, water consumption varies significantly across regions (UNESCO, 2024, 2022). In arid and semi-arid areas, such as the Mediterranean, agriculture accounts for more than 80 % of water use, whereas in humid regions such as the United Kingdom, agricultural consumption represents only 5 %, which highlights the importance of adapting solutions to local conditions (Rhodes, 2017).

Sustainable water management in agriculture is critical to ensuring long-term food security (FAO & UN-Water, 2021). Reclaimed water for agricultural irrigation is a viable solution and accordingly it has been used on a variety of crops, including citrus, olives, tomatoes, lettuce and cereals (Christou et al., 2024; Ofori et al., 2021). Beyond meeting quality standards, reclaimed water can provide essential nutrients, reducing dependence on chemical fertilizers (Maestre-Valero et al., 2019).

Wastewater recovery and reuse in agriculture can be achieved through physicochemical and biological treatment systems, often followed by tertiary and quaternary processes (Rezaei et al., 2019; Al-Hazmi et al., 2023). Physicochemical processes include membrane filtration technologies, such as ultrafiltration (UF) and reverse osmosis (RO), as well as coagulation-flocculation. While effective, they require substantial energy and chemical inputs and generate brine and solid waste. Biological treatments, such as the conventional activated sludge (CAS) process, efficiently remove organic matter and nutrients like nitrogen and phosphorus. However, they involve high electricity demand for aeration, generate sludge requiring further treatment, and may fail to remove recalcitrant micropollutants (Batstone et al., 2015). Constructed wetlands (CWs), as an example of Nature-Based Solutions (NBS), can remove nutrients, organic matter, and heavy metals while offering low operating costs and ecological benefits. The main drawbacks are large

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land areas and are sensitive to climatic conditions (Ferreira et al., 2023). Advanced oxidation processes, such as ozonation, can remove persistent organic pollutants, making them suitable for sensitive crops, but their high energy demand increases operating costs (Khan et al., 2023). Photocatalysis based on the use of ultraviolet (UV) light and catalysts like titanium dioxide (TiO_2) shows potential for removing organic contaminants and microorganisms. However, its large-scale application is limited by infrastructure and operating costs (Carré et al., 2017; Ingrao et al., 2020).

The feasibility of water reuse must be evaluated, as it could be an alternative to drinking water treatment, incorporating broader indicators than just those related to irrigation water quality, water risk, access to freshwater and water availability (Leão et al., 2022). Life Cycle Assessment (LCA) has become a key tool for assessing the environmental sustainability of wastewater treatment technologies and among these, those focused on wastewater reclamation. The most popularly evaluated with LCA were the membrane bioreactors, reverse osmosis and capacitive deionization (Huang and Lee, 2026). Evidence from previous LCAs studying these technologies consistently highlights energy consumption as the dominant driver of environmental impacts. This is critical because both wastewater and water treatment processes are highly energy-dependent. Agricultural water reclamation must therefore reduce the energy demand in order to improve the environmental impact, which is mainly analyzed through the global warming potential indicator (Canaj et al., 2021b; Crovella et al., 2024). As important is the quantification of the energy use as the directly generated greenhouse gas emissions, since the proposal of strategies to control them ensure the achievement of a zero-carbon emissions process (Guo et al., 2024). While monitoring both direct and indirect emissions is essential towards achieving a sustainable water irrigation system, many lower carbon processes reduce fossil fuel use but rely more on water-intensive steps (i.e., cooling, washing or growing biomass).

Therefore, and in order to preserve the ecosystems considering both air quality and water availability, it is necessary to measure not only the global warming potential but also the water demand. The goal is to reach water neutrality and be environmentally friendly, which is especially important in water scarcity regions. For this reason, LCA can be used as a tool for monitoring both climate change and water resource exploitation while simultaneously defining restorations initiatives (Crovella et al., 2024).

However, reviews show that LCA studies from the last decade focused on case-specific scenarios accounting for the impacts of design and operation before the reclaimed wastewater treatment was used in agriculture (Bayart et al., 2010; Wang et al., 2021; Mehmeti and Canaj, 2022). Beyond the process perspective of the studies, the LCAs are also midpoint-oriented, with European system boundaries, with the use of volumetric based functional units and with the use of ReCiPe and Ecoinvent as LCIA models and background databases. Nevertheless, the examination of the benefits and trade-offs is underrepresented within the literature, especially when there is a need for a continuous upgrading of wastewater treatment plants with novel technologies and a lack of data of these technologies at full-scale (Rashid et al., 2023). Also, numerous studies concentrate on the research of carbon dioxide emissions from energy demand and form sludge treatment management (Nguyen et al., 2020). Carbon sequestration is a benefit that is not directly related to nutrient recycling, but it is interesting to understand the fate of carbon in order to foresee the global warming potential and the application of carbon-based bioproducts from wastewater treatment plants. Indeed, carbon sequestration was studied for the agricultural land application of sludge and the emissions coefficients and release fluxes from plant uptake from BNS (Lam et al., 2020; Engida et al., 2020 Zhang et al., 2025). Representative research for the application of the LCA for the concept of zero-carbon emissions is the work of Fang et al. (2025), who have applied it to biological material production, steel making, battery manufacturing, water purification and hydrogen processing. The water municipal sector was also analyzed at national level but the carbon neutrality on the wastewater treatment facilities, that are not NBS, is still underrepresented (Lehtoranta et al., 2025).

On the other hand, there are academic and technical articles/works that relate LCA to concepts closely tied to water neutrality (especially through water footprinting, water impact assessment, analysis of the conceptual framework and identification of initiatives), but the explicit pairing of “LCA” with “water neutrality” is still relatively niche in the formal literature (Kumar et al., 2024; Mwafy et al., 2025).

In this context, this study incorporates the existing life cycle approach considering typical goals with similar impact categories, functional unit, methods and background databases of an LCA while paying attention to the estimation of offsetting emissions towards the achievement of both a carbon and water neutrality. As usual, critical environmental aspects are first identified and compared to determine the most suitable option among case studies chosen from the Iberian Peninsula, an area prone to water scarcity. In contrast with this standardized perspective, the study examines water and carbon emissions mitigation using primary also primary data from full-scale facilities. Reclamation can offset indirect water demand during technology operation, while NBS may also act as carbon sinks. To capture this dynamic, carbon trade-offs are investigated and climate compensation strategies through active forest management. This integrated approach combines multiple environmental dimensions to avoid burden shifting and to capture impacts across different processes and categories (Chen et al., 2023).

2. Methodology and methods

In this study, LCA was applied to evaluate the environmental impacts of water reclamation processes and wastewater treatment plants (WWTPs) whose effluents are intended for agricultural use, thereby reducing reliance on natural water sources such as groundwater and rivers. Four case studies from the Interreg Sudoe European project I-Rewater *(“Sustainable management of water resources in irrigated agriculture in the SUDOE space”)*, were studied.

2.1. Description of the different wastewater treatment configurations

The Alfândega da Fé (ALF) case study (Fig. 1a) is a WWTP located in the Bragança district of the Trás-os-Montes region in Portugal, with a treatment capacity of 388 m^3/d. Reclaimed water is used for drip irrigation in a nearby almond grove covering 2400 m^2, with irrigation taking place daily from mid-May to early September. The pre-treatment stage includes both a coarse and fine screening, as well as a grit removal unit, which is connected to a sand classifier. The primary treatment stage consists of two primary settling units. The sludge from the first settling unit is transported by gravity into a thickener, where it undergoes dewatering. In the secondary treatment stage, two biofilters and two secondary settling units are implemented. The effluent from the settlers is split into two streams: one is directed to the tertiary treatment, while the other is recycled into the biofilter. The effluent from the water line is stored in a 10 m^3 tank before being pumped into a disc filtration unit. To prevent microbial growth in the filter, sodium hypochlorite is added. The treated effluent then undergoes UV disinfection before storage as reclaimed water. The chemicals used in the plant are polyelectrolyte, which acts as a flocculant in the sludge line, and 13 % sodium hypochlorite, which is used for routine membrane cleaning.

The Mirandela (MIR) case study (Fig. 1b) is an industrial WWTP located in the Braganza district of the Trás-os-Montes region, Portugal with a treatment capacity of 3444 m^3/d. Reclaimed water is used for the irrigation of a nearby olive grove, with a monthly volume of 1000 m^3 from mid-May to September. The resulting sludge is utilized as compost. The pre-treatment process includes coarse and fine screening units. During the gritting stage, sand is removed and transferred to a classifier and dewatering unit. After the biological reactor, a portion of the treated effluent undergoes filtration and UV disinfection. To prevent microbial

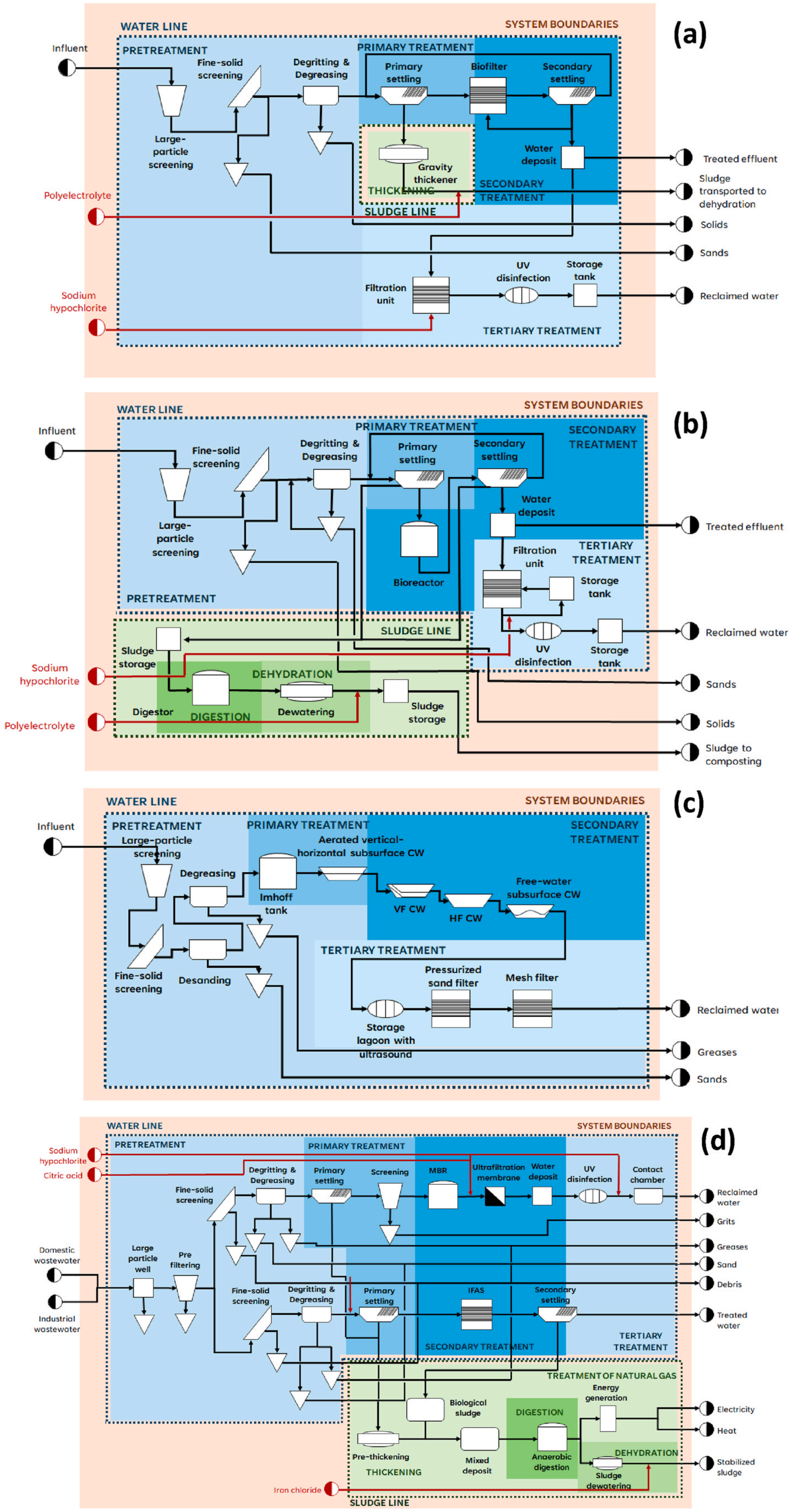


Fig. 1. Process flow diagram of Alfândega da Fé (**a**), Mirandela (**b**), Carrión de los Céspedes (**c**) and Gavá-Viladecans (**d**) wastewater treatment plant.

growth in the filter, sodium hypochlorite is added during filtration. Sludge from the primary and secondary sedimentation stages is stored in a tank and processed by anaerobic digestion and dewatering. The dehydrated sludge is then stored in a 40 m^3 tank before being transported for composting.

The CENTA or Experimental Centre for New Water Technologies case study (Fig. 1c) is a research facility located in Carrión de los Céspedes (Spain). The wastewater is pumped to CENTA via a 300 mm collector. The wastewater influent flow varies according to the experimental configuration selected. In this case, the daily flow has been estimated at approximately 40 m^3/d. The wastewater undergoes a comprehensive NBS treatment process, mainly based on CW, such as Vertical subsurface CWs, Horizontal subsurface CW, Surface CW and Floating helophyte wetland. The CWs are fed by the outlet of the pre-existing Imhoff Tank, which has been pre-treated with screening and an aerated grit and grease removal chamber, with one CW receiving pre-treated wastewater directly. Post-treatment begins in the storage lagoon, where ultrasound treatment is applied for microalgae and *E. coli* removal. This is followed by a filtration system, consisting of a pressure sand filter and a mesh filter. Finally, the reclaimed water is conveyed to the olive grove plot via a pressure group, where it is used for surface drip irrigation.

The GAVA or Gavá-Viladecans WWTP case study (Fig. 1d) is located in Barcelona, Spain, and treats pretreated domestic and industrial wastewater, serving Castelldefels, Gavà, Viladecans, areas of Sant Boi de Llobregat, and Sitges, which comprise the Barcelona metropolitan area. It is part of a sewerage system that includes nine pumping stations, a 45 km collection network, and a 2.4 km outfall. The treatment process consists of pre-treatment, primary treatment, and secondary treatment. The pre-treatment stage includes coarse screening, followed by sand and grease removal. In the primary treatment, suspended solids settle in primary settling tanks and are then directed to the sludge line for further processing. The secondary treatment operates through two parallel systems: a membrane bioreactor (MBR) and an integrated fixed film activated sludge (IFAS) system. Both processes remove nitrogen, phosphorus, and organic matter while ensuring high treatment efficiency. After biological treatment, the water undergoes ultrafiltration and UV disinfection, followed by the addition of hypochlorite. The reclaimed water is then suitable for agricultural irrigation, urban green areas, and street cleaning. The treated water is discharged into the sea through a 1.5 km underwater tunnel at a depth of 15 m. Additionally, a portion of the treated water is redirected to the Llobregat Delta, contributing to the preservation of wetland ecosystems, including the Arrayán lagoon.

2.2. *Goal and system boundaries definition*

Defining the boundaries of the system is a prerequisite for conducting an LCA (Bayart et al., 2010). The most common definition of system boundaries in wastewater treatment considers a process perspective, including only the operation phase. The reason for excluding infrastructure in this type of facility was related to its minor or even insignificant contribution compared to the operational phase (Foley et al., 2010). For the constructed wetlands, this aspect has been analyzed separately, as it is known to be a significant factor for a 10-year lifespan of the facility (Machado et al., 2007; Resende et al., 2019). The attributional LCA has been chosen as emission accounting perspective. On the other hand, the functional unit was defined as 1 m^3 of water purified and/or wastewater treated.

Additionally, a *Cradle-to-gate* approach was applied, which includes not only the direct emissions generated during facility operation but also those arising from the manufacturing processes of the energy and chemicals consumed (ISO 14040, 2006). The impacts associated with the use of co-products or the treatment of residues were not considered. Regarding the geographical system boundaries, the scope of the study was defined as European, with a focus on Spain and Portugal, where the WWTPs are located.

2.3. *Collection of inventory data*

The LCI represents a critical phase in which all the inputs and outputs related to the process under evaluation are systematically quantified. For each scenario, Table 1 compiles primary data for 2024. In this study, the foreground data were primarily collected through questionnaires completed by treatment plants. Nevertheless, some estimates were derived from previous publications (as shown in Table 1) and databases (i.e., Eurostat and PRTR or *Protocol on Pollutant Release and Transfer Registers*). For air emissions, only direct emissions have been accounted for CH_4 and N_2O, while CO_2 was considered as a biogenic gas. The underlying reason lies in the source (organic matter in human excreta and food waste) of carbon emissions, as indicated by the IPCC 2006 methodology based on the type of WWTP (IPCC, 2006). Bibliographic data from indirect emissions were taken from the Ecoinvent® V3.10 database (Moreno Ruiz et al., 2020). In accordance with the system boundaries (Section 2.2) and to ensure comparability of the processes based on the

Table 1
Life cycle inventory of the 4 case studies expressed per FU (1 m^3 of wastewater treated).

Life cycle items/Case Study Name	ALF	MIR	CENTA	GAVA
Inputs from Technosphere				
Construction				
Gravel (kg)	–	–	$204.55^{(1,2,4)}$	–
Concrete (L)	–	–	$0.91^{(1,2,4)}$	–
PVC (g)	–	–	$60.04^{(1,2,4)}$	–
PP (g)	–	–	$12.17^{(1,2,4)}$	–
Aluminum (g)	–	–	$0.32^{(1,2,4)}$	–
PE (g)	–	–	$0.28^{(1,2,4)}$	–
Consumables				
Polyelectrolyte (g)	0.48	63.22	–	–
Sodium hypochlorite (g)	1.93	1.93	–	7.37
Citric acid (g)	–	–	–	1.93
Iron chloride (g)	–	–	–	14.65
Gravel (g)	–	–	$32.39^{(1)}$	–
Iron (g)	–	–	$1.09^{(1)}$	–
Sand (mg)	–	–	$821.97^{(1)}$	–
Nickel (mg)	–	–	$56.06^{1)}$	–
Sodium chloride (mg)	–	–	$45.27^{(1)}$	–
Energy				
Total energy demand (kWh)	0.12	0.29	$0.26^{(3,4)}$	0.51
Transportation				
Construction materials transport (kg·km)	–	–	$33{,}554.63^{(5)}$	–
Consumables transport (kg·km)	1.20	113.15	$5.66^{(5)}$	$4.02^{(5)}$
Emissions to the environment				
Treated effluent				
Treated water (m^3)	0.97	0.97	$0.35^{(4)}$	0.57
COD (g)	23.57	25.33	$43.00^{(4)}$	20.85
BOD_5 (g)	11.86	13.17	$4.00^{(4)}$	2.47
TN (g)	8.21	8.28	$2.20^{(4)}$	27.00
TP (g)	1.18	1.44	$3.10^{(4)}$	0.52
TS (g)	9.13	2.13	$3.00^{(4)}$	4.48
Emissions to the air				
N_2O (mg)	$150.57^{(6)}$	$158.78^{(6)}$	$477.95^{(6)}$	$0.16^{(7)}$
CH_4 (mg)	$222.80^{(6)}$	$673.24^{(6)}$	$10{,}125.00^{(6)}$	$1.63^{(7)}$
Outputs to the Technosphere				
Co-products				
Reclaimed water (L)	35.42	35.44	350	425.61

Note: Data estimations from (1): (Machado et al., 2007), (2) (Peñacoba-Antona et al., 2021), (3): (ICRA, 2025), (4): (Ávila et al., 2015), (5) (Eurostat, 2024), (6) (IPCC, 2006), (7) (PRTR, 2025).

facilities geographical location, the background processes were chosen to be representative of a European context. Some exceptions, based on availability within Ecoinvent®, were sodium hydroxide, sodium bicarbonate, iron chloride, sand, polyethylene, Polyvinylchloride, concrete and aluminum. On the other hand, and since electricity was proven to be a strong contributor (more information in Section 3 related the description of results), not only the European electricity grid was tested but also the research incorporates and explanation of the influence of the location (Portugal or Spain) of the facilities in the environmental profile from changes in the electricity grid.

2.4. Life cycle impact assessment

The inventory data from each case study shown Table 1 was implemented in the SimaPro® software V10.0.0.29 to facilitate its translation into environmental impacts (PRé Sustainability, 2022). In this study, the ReCiPe 2016 Midpoint (H) V1.07 method was selected to provide results for the 8 most common categories for the literature related with wastewater treatment: climate change (CC - kg $CO_{2eq.}$), stratospheric ozone depletion (SOD – kg $CFC11_{eq.}$), terrestrial acidification (TA - kg $SO_{2eq.}$), freshwater eutrophication (FE - kg $P_{eq.}$), marine eutrophication (ME - kg $N_{eq.}$), land use (LU - m^2a $crop_{eq.}$), fossil resource scarcity (FRS - kg $oil_{eq.}$) and water consumption (WC - m^3) (Pfister et al., 2009; Mehmeti and Canaj, 2022).

After assessing the environmental profiles using the ReCiPe 2016 method, another was used: Environmental Footprint (EF) 3.0 (ERC-JRC, 2018). The rationale for shifting to this method was to evaluate how the results differ when using location-specific characterization factors (Europe instead of global). This was tested with two categories in wastewater treatment: CC and FE. Both indicators capture the relevance of energy demand and effluent emissions in the impact profiles. Water demand was also included as a midpoint category in this comparative analysis. However, in this case, the two selected methods apply different indicators: ReCiPe measures water consumption, while EF assesses the potential for water deprivation in a given area.

2.5. A connection between the LCA, carbon capture and carbon neutrality in WWTPs

To fully evaluate the environmental profile of the WWTPs, it is crucial to account for carbon capture, with the CENTA case standing out as the most relevant among the four due to its greater natural capacity as a carbon sink. The calculation of this natural carbon footprint is simulated with Eq. (6), as indicated by Global Footprint Network (2025).

$$CF = \left[\frac{(P_C + P'_C)(1 - S_{Ocean})}{Y_C}\right] \cdot EQF \tag{6}$$

In this definition (Eq. (6)) CF represents the carbon footprint of CWs (gha or global hectares); P_C and P'_C are the net indirectly emitted equivalent carbon emission and net direct CH_4 and N_2O emissions in equivalent carbon ($tCO_2eq./y$), respectively. Literature values are used in Eq. (6) as S_{Ocean}, which corresponds to the fraction of anthropogenic CO_2 emissions sequestered by oceans in a given year (dimensionless), Y_C is the annual rate of carbon uptake per hectare of world average forest land ($tCO_2eq./ha/y$) and EQF as equivalent factor of CF for crop-land (dimensionless). The default value set for each of the previously mentioned variables of Eq. (6) were 0.30, 3.59 and 2.5 according to 2019 National Footprints Accounts (Świąder et al., 2020). The EQFs are related to the land-use type and therefore, they achieve different values. For example, 0.46 for grazing land, 0.37 for inland fishing grounds and 1.28 for forests.

The carbon sequestration capacity (g $CO_2/m^2/y$) of the constructed wetlands is defined as a function of net primary productivity (NPP). This parameter is determined as the fraction of organic carbon fixed by wetland plants through photosynthesis without accounting for their own consumption. The calculation is represented in Eq. (7) as described by (Zang, 2021) where 1.62 acts as a conversion for carbon emissions. NPP was estimated considering two vegetation populations: *Phragmites australis* (with 2470.33 g $C/m^2/y$ of NPP) and *Typha* (2271.1 g $C/m^2/y$ of NPP), as identified as the most predominant species (Wang et al., 2023).

$$C = 1.62 \cdot NPP \tag{7}$$

Although the CENTA case study is the most complex due to the estimation of carbon sequestration capacity, the other technological alternatives for wastewater recovery (known as ALF, MIR, and GAVA) can also provide results in terms carbon capture aiming to achieve carbon neutrality. However, these case studies are not able to take the carbon directly from facility on-site since there are not technologies able to achieve this purpose. Apart from constructed wetlands, some of them could be microalgae cultivation systems to fix CO_2, use of physical absorbents to capture CO_2 from flue gases, or use of bio-electrochemical systems (which can also produce H_2) (Zhou et al., 2025). Therefore, the compensation of the carbon emissions from the operation or construction must be done externally to the wastewater treatment plant boundaries. Possible strategies could be CO_2 pipeline and storage in saline formations, reforestation and afforestation activities, purchase of carbon credits from other companies or green investments, soil and blue carbon sequestration. Among this, the present research provides the estimation of the average global area from a possible plantation needed to measure the offset direct (such as the previously described carbon sequestration from constructed wetland) and indirect emissions (from electricity or chemicals production) during the operation of the facility. In this regard, this research provides a comparative assessment also based on the cropland (and other plantations) area needed to mitigate the effects of climate change within the system boundaries. Thus, carbon compensation as active cropland management project was estimated for all the facilities considering first their natural ability to capture carbon (only for CENTA) and secondly its external greenhouse gas emissions mitigation.

2.6. A connection between the LCA and water neutrality in WWTPs

In addition to its application in assessing carbon-related impacts, LCA provides a robust analytical framework for supporting strategic sustainability objectives such as water neutrality across all life cycle stages. LCA enables the compilation of detailed water inventories that support water balance calculations, as well as the evaluation of water-related impacts, including water scarcity (e.g., through characterization models such as AWARE) and water quality effects, reflected in impact categories such as eutrophication and ecotoxicity. These analyses are grounded in the systematic identification and quantification of water-stress hotspots associated with water consumption, thereby contributing to the protection of ecosystem integrity. In the present study, water-related impacts were assessed within the LCA framework by examining key components of wastewater treatment processes that are critical to the WC impact category. In addition, an absolute WC indicator was calculated to account for both direct and indirect water demands throughout the system.

Following impact assessment, the standard methodological approach involves identifying feasible water reduction strategies and defining appropriate replenishment measures, such as watershed restoration initiatives or water reuse and recycling schemes. The literature reports a wide range of reclaimed water reuse applications, including agricultural irrigation, pavement construction, urban landscaping (e.g., parks, golf courses, and green areas), and other non-potable uses in industrial and service sectors, such as cooling systems, manufacturing operations, vehicle washing, and laundry services (Harishbabu et al., 2024; Mwafy et al., 2025). Within the scope of the I-rewater project, this research specifically evaluates the extent to which selected case studies can approach water neutrality when reclaimed water is applied for agricultural irrigation, with the dual objective of supplementing freshwater withdrawals and delivering nutrients to crops in support of sustainable farming practices. Nevertheless, the deployment of reclaimed water for irrigation may be constrained by multiple barriers, including water salinity, the presence of emerging contaminants and heavy metals, administrative and regulatory inefficiencies, limited institutional coordination, high infrastructure investment costs that can elevate reclaimed water prices, and societal acceptance challenges (Villacorta-Ranera et al., 2024).

Water neutrality can be substantiated when water withdrawals and replenishment actions are demonstrated to offset one another based on LCA-derived evidence. This approach avoids potentially misleading claims that rely solely on direct or site-specific water use and instead ensures transparency, credibility, and reproducibility through standardized metrics, while remaining consistent with established frameworks such as ISO 14046 and the Alliance for Water Stewardship.

3. Results

3.1. Contribution analysis per case study

3.1.1. Alfândega da Fé WWTP

The LCIA of the ALF scenario is demonstrated to be primarily influenced by energy consumption, being the critical element in 4 (TA, LU, FRS and WC) of the 8 selected categories of the ReCiPe method (shown in Fig. 2a). This is caused by an intensive electricity demand, 0.12 kWh/m^3, compared to the influence of the other elements such as chemicals and direct emissions.

Although it is also significant in the CC category (41 %), air emissions from the biological treatment process contributed the most (54 %). This aligns with the findings of Maktabifard et al. (2023), who showed that direct process emissions in WWTPs can account for over 60 % of climate change impacts. Additionally, effluent emissions have a strong influence on eutrophication, with FE at 98 % and ME at 99.9 %, primarily due to phosphorus and nitrogen loads, respectively. Regarding chemicals, the main hotspot is the consumption of sodium hypochlorite in WC (around 14 %).

3.1.2. Mirandela WWTP

For the MIR scenario (Fig. 2b), energy consumption largely affected 5 (between 36 % and 91 %) of the 8 categories but only in two (CC and LU). This is in line with results from literature, on which around 50 % of the carbon emissions of WWTPs came from electricity demand (Chen et al., 2023). In contrast with the results from ALF, Mirandela also has a profile characterized by the demand for chemicals, particularly the polyelectrolyte that is used during the dehydration of sludge. This chemical is relevant in 4 of the categories (as shown in Fig. 2b) with contributions in the range of 34 %–55 %. Emissions of N_2O and CH_4 from biological treatment contribute to CC (~26 %) and SOD (96 %). Effluent emissions are a key driver of eutrophication in both freshwater and marine ecosystems, mainly due to high phosphorus and nitrogen levels. Although both nutrients are vital for crop fertilization, only 3.5 % is used for water reclamation, with the rest discharged directly into the environment.

3.1.3. Carrión de los Céspedes WWTP

As noted in Section 2.2, wastewater treatment configurations based on nature-based solutions, such as CENTA, are expected to have a greater impact during the construction phase than during operation (Lopsik, 2013). It was reported in the literature that around 20–45 % of the impacts originated from construction materials (Andreo-Martínez et al., 2021; Cao et al., 2025; Yin et al., 2025).

CENTA is completely characterized by the materials of construction in 5 (CC, TA, LU, FRS and WC) of the 8 categories analyzed (Fig. 2c). Among these five categories, FRS has the highest contribution (99 %) and CC the lowest (93 %). The main factors in construction are gravel production and material transport. Specifically, 93 % of LU and 94 % of

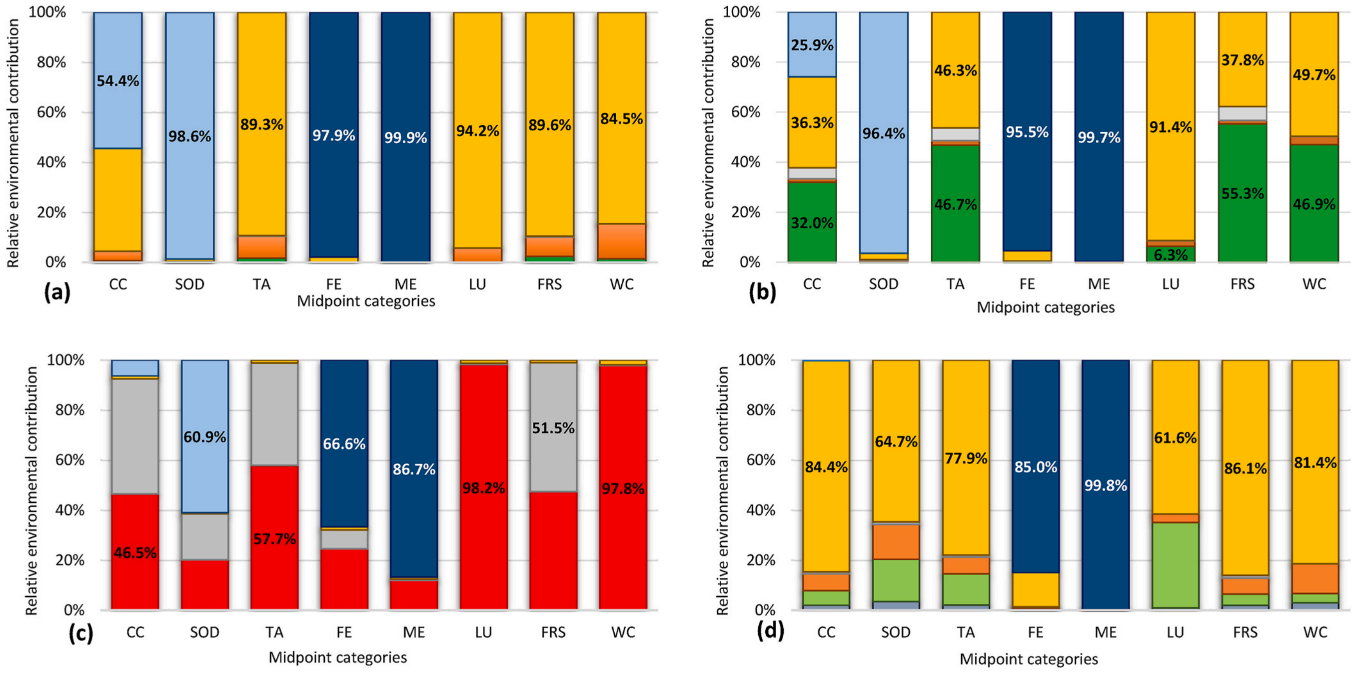


Fig. 2. Relative environmental impact (MidPoint) profile of the case studies of Alfândenga da Fé WWTP (**a**), Mirandela (**b**), Carrión de los Céspedes (**c**) and Gavá-Viladecans (**d**). 🟩Polyelectrolyte; 🟨 Electricity; 🟧Sodium hypochlorite; 🟦 Wastewater effluent emissions; ⬜Transportation; 🟦 Air direct emissions; ■ Iron chloride; 🟩Citric acid; 🟥Construction.

WC impacts come from gravel extraction, while about half of the impacts in CC, TA, and FRS are due to transportation. Similar outcomes were reported by Dufossé et al. (2025) who reported that gravel, sand, stainless were the three most contributing materials in the profile of vertical flow wetlands. Also, the impacts related with construction and end-of-life stages was dominated in TA (49 %) and FRS (around 60 %) (Dufossé et al., 2025).

The contribution of the operation phase ranges from 1.1 % to 88 % across categories, with FRS and ME representing the lowest and highest impacts, respectively. The most affected impact categories during operation are SOD, FE, and ME. SOD is primarily influenced by direct air emissions (about 61 %), with dinitrogen monoxide from nitrogen compound degradation in the wetlands being the most significant factor. FE and ME are mainly affected by aquatic emissions of organic matter, nitrogen, and phosphorus. Similar outcomes were achieved by Yin et al. (2025), who revealed that direct emissions were also relevant and accounted for around 46 % in CWs.

3.1.4. *Gavá-Viladecans WWTP*

Lastly, GAVA has been environmentally assessed, showing that energy consumption affects 6 out of 8 midpoint categories studied (Fig. 2d). The range of impacts for electricity is quite broad, with the highest contribution attributed to FRS at 86.1 % and the lowest to ME (<1 %). The major (64 %) impact from electricity comes from the operation of the MBR and IFAS (biological treatment) followed by the primary treatment and pre-treatment. The aquatic emissions were the second contributing factor for the ME (99 %) and FE (85 %) categories. Special attention should be given to the chemicals used throughout the operation, particularly citric acid, as it is the second most significant contributor to environmental impact in all categories except for the two of eutrophication.

3.2. *Comparative analysis of the facilities*

Fig. 3 shows the comparison for the operation of the different scenarios. According to the results, the best-performing scenario is ALF, which ranks highest in 5 impact categories (CC, TA, LU, FRS, and WC). The worst-performing scenario is GAVA, which scores lowest in four categories. Accordingly, the ranking of the facilities based on their environmental impact is: ALF, CENTA, MIR, and GAVA.

The ranking of facilities for energy consumption, which can be exemplified with the climate change category, is: 0.098, 0.27, 0.55 and 0.58 kg CO_2eq./m^3 for the operation of ALF, MIR, GAVA and CENTA (shown in Table 2), respectively. These results are lower than those reported in the literature such as Bisinella de Faria et al. (2015) (1.05 kg CO_2eq./m^3), Gupta et al. (2024) (0.51–1.14 kg CO_2eq./m^3) and Sun et al. (2024) (0.76–1.09 kg CO_2eq./m^3). Moreover, the deviation of the European electricity mix profile, chosen in line with the system boundaries, is lower in all cases than 16.8 % when used the country specific electricity mix.

It should be noted that, except for GAVA, energy consumption at the facilities is also lower than the average reported for Spanish WWTPs (PRTR, 2025). The average and median values are provided for 57 Spanish WWTPs (shown in Fig. 4). The median and average value of energy consumption is 0.37 and 0.32 kWh/m^3, respectively. It should be taken also into account that, as shown in Fig. 4, the energy demand of the majority (40 %) of the facilities is between 0.2 and 0.4 kWh/m^3. In accordance with this, ALF performs below the average, MIR and CENTA

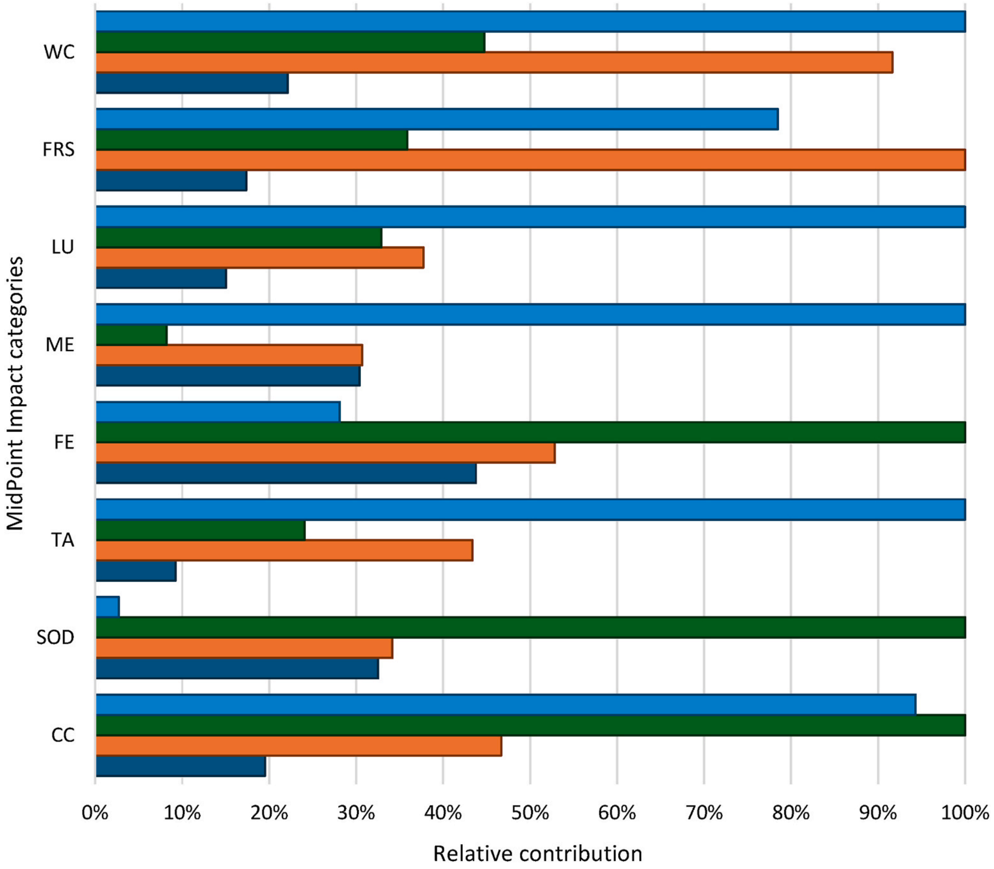
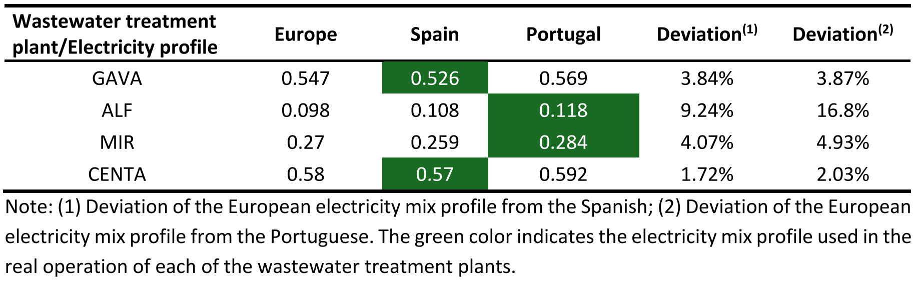


Fig. 3. Comparative environmental impact (MidPoint) profile of the case studies. 🟦 Gava-Viladecáns; 🟩 Carrión de los Céspedes; 🟧 Mirandela; ■ Alfândenga da Fé.

Table 2
Carbon footprint (expressed in kg CO_2eq./m^3) estimated with the ReCiPe 2016 method in all 4 cases studies.



Wastewater treatment plant/Electricity profile	Europe	Spain	Portugal	Deviation(1)	Deviation(2)
GAVA	0.547	0.526	0.569	3.84%	3.87%
ALF	0.098	0.108	0.118	9.24%	16.8%
MIR	0.27	0.259	0.284	4.07%	4.93%
CENTA	0.58	0.57	0.592	1.72%	2.03%

Note: (1) Deviation of the European electricity mix profile from the Spanish; (2) Deviation of the European electricity mix profile from the Portuguese. The green color indicates the electricity mix profile used in the real operation of each of the wastewater treatment plants.

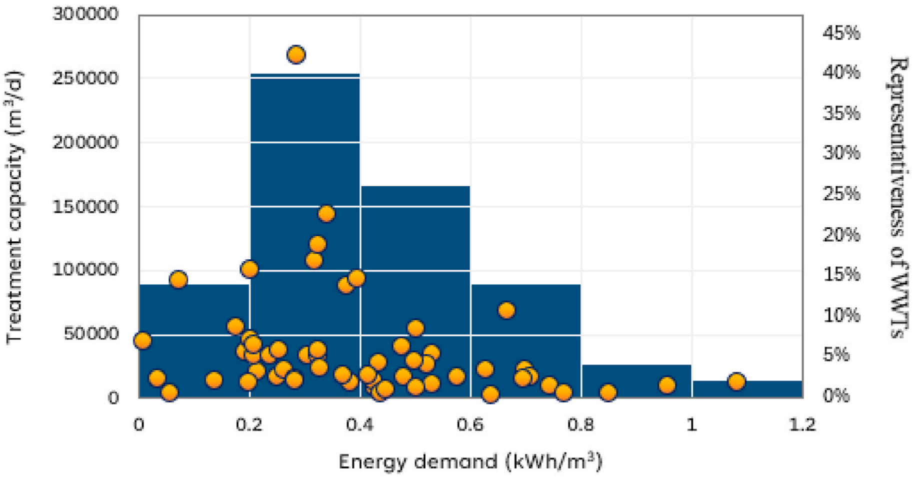


Fig. 4. Energy demand per capacity in wastewater treatment plants in agreement with the data shown in the Spanish PRTR database (left y-axis) representativeness of the number of facilities with the same energy consumption range (right y-axis).

are within the expected values and GAVA has the worst operating conditions. If considering only the electricity demand, the climate change of GAVA would be 0.17 kg CO_2eq./m^3 compared with the 0.13 kg CO_2eq./m^3 to be reported by an average Spanish facility.

3.3. The use of location specific factors

Table 3 presents the results for some of the impact categories studied (those with similar units) but also compares them to those obtained using the EF 3.0 method. The largest differences between the two methods are 9.24 % for CC and 38 % for FE. This indicates that using regional versus global factors has little effect on climate change results but can significantly influence eutrophication. Consequently, the release of phosphorus or organic matter in sensitive areas may require the application of higher local characterization factors, leading to a larger absolute impact on eutrophication. It is therefore important to apply local characterization factors that can be differentiated for both the foreground and each background system.

The comparison of water demand between the two methods is less straightforward, as the indicators reflect different concepts, consumption versus deprivation. The results of the EF3.0 could be up to 25 times higher than those of ReCiPe. In the water scarcity category, impacts reflect the potential to deprive other users of water based on water use and the remaining water per unit area (Boulay et al., 2018). In this context, GAVA has the highest water demand, while MIR has the greatest potential to limit water availability for others.

When considering water consumption, all four scenarios report an indirect demand caused by chemical and energy use ranging from 0.9 to 3.8 m^3 per cubic meter of wastewater treated. The most significant element with a contribution to this water withdrawal is the electricity demand during the operation. ALF is the facility with the highest representativeness for electricity with around 84.5 % despite being the one with the lowest use of this resource. On the other hand, and as indicated in Section 2.6, the effluent of the facilities is planned to be used for agricultural irrigation. However, the inventory shown in Table 1 demonstrates that the reclamation has been performed partially and not all the effluent has been used. Under the current situation, none of the facilities can be claimed as water neutral since the water

Table 3
Results of the most significant impact categories through the E.F. 3.0 method and the ReCiPe 2016 method in all 4 cases studies.

Impact category		Unit	Method	ALF	MIR	CENTA	GAVA
Water consumption	WC	L consumed	ReCiPe	0.9	3.5	1.7	3.8
Water scarcity	WS	L deprived	E.F. 3.0	15.1	88.9	28.8	71.4
Climate change	CC	g CO_2 eq.	ReCiPe	98.53	270.9	580.4	547.4
Climate change	CC	g CO_2 eq.	E.F. 3.0	95.97	226.1	621.1	585.2
Eutrophication, freshwater	FE	mg P eq.	ReCiPe	1820.6	2197.8	4158.4	1169.5
Eutrophication, freshwater	FE	mg P eq.	E.F. 3.0	1224.0	1558.6	3202.9	727.3

reclamation does not compensate for the indirect use from the consumption of resources, such as chemicals and electricity. However, ALF could potentially achieve self-sufficiency in the future if the effluent were recycled for use in the required consumables besides the current farming purposes, since its water use is lower than the water generated or discharged. Currently, 0.9 m^3 of water is consumed for 0.035 m^3 of effluent being reclaimed for the ALF scenario. This is notable because, among the four scenarios, ALF is the only one that contributes more to the water cycle than it consumes during treatment, resulting in a net positive water flow. Thus, the other scenarios should work not only on increasing the reclamation rate of the effluent but also resource minimization measures should be incorporated (as this could be translated into a reduction of indirect water withdrawal).

3.4. *Performance in terms of carbon capture*

Table 4 depicts the calculation of the carbon balance applied onto the vegetation in CENTA WWTP as described in Section 2.5. If only the carbon footprint of direct emissions is taken into account, the carbon capture of the CW vegetation indicates that facility emits only 0.029 kg CO_2eq./m^3. Translated into average cropland area, the facility is not able to compensate direct emissions and needs and extra of 0.79 ha (with an EQF of 2.5) of cropland plantation.

When direct and indirect emissions from operations are taken into account, the results of the climate change category from the previous Section 3.2 indicates that the carbon sequestration by the vegetation in the CWs cannot fully offset emissions from the operation phase, as it releases around 0.12 kg CO_2eq./m^3 to the atmosphere. The carbon uptake is also insufficient to compensate for the construction phase, resulting in 5.9 % of total carbon being sequestered. This is a more conservative estimate than that reported by Cao et al. (2025), who suggested that photosynthesis in CWs could offset up to 15 % of GHG emissions. In terms of cropland area, a total of 0.79 ha is needed to offset CW activities. Incorporating the results of infrastructure construction translates into the need to plant an additional 46.8 ha of forest.

CENTA can act as a carbon sink through plant photosynthesis during wastewater treatment and, thus both internal and external mitigation measures can be implemented (as described in Section 2.5). For the other facilities (ALF, MIR, and GAVA), the only option could be offset greenhouse gas emissions through active cropland or forest management (or other projects) that remove equivalent CO_2 emissions elsewhere. Using the same calculations as for CENTA, this would require 16.7 ha for ALF, 138.5 ha for MIR, and 566.2 ha for GAVA each year.

Table 4
Carbon balance on the CENTA WWTP for the E.F. 3.0 method and the ReCiPe 2016 Method.

Parameter	Unit	Method	Value
Carbon sequestration capacity for CENTA CWs	kg CO_2/y	–	−6037.4 (1)
Absorbed CO_2 emissions by vegetation	t CO_2/y	–	−6.04 (1)
	g CO_2 eq./m^3	–	−457.38 (1)
Direct CH_4 emissions to air	g CO_2 eq./m^3	ReCiPe	344.25
		E.F. 3.0	372.60
Direct N_2O emissions to air	g CO_2 eq./m^3	ReCiPe	142.4
		E.F. 3.0	142.4
Total carbon footprint of CENTA	g CO_2 eq./m^3	ReCiPe	580.4
		E.F. 3.0	621.1
Net balances carbon footprint (with capture of direct emissions)	g CO_2 eq./m^3	ReCiPe	29.27
		E.F. 3.0	57.62
Net balances carbon footprint (with capture of total emissions)	g CO_2 eq./m^3	ReCiPe	123.02
		E.F. 3.0	163.72
Carbon Footprint (as indicated in Eq. (6), EQF as 2.5 and only operation)	gha	ReCiPe	0.79
		E.F. 3.0	1.05
Carbon Footprint (as indicated in Eq. (6), EQF as 0.46 and only operation)	gha	ReCiPe	0.15
		E.F. 3.0	0.19
Carbon Footprint (as indicated in Eq. (6), EQF as 0.37 and only operation)	gha	ReCiPe	0.12
		E.F. 3.0	0.16
Carbon Footprint (as indicated in Eq. (6), EQF as 1.28 and only operation)	gha	ReCiPe	0.41
		E.F. 3.0	0.54

Note: (1) The negative values indicate a removal of the compounds to the environment.

4. Discussion

4.1. *The carbon footprint of agricultural irrigation*

Wastewater reclamation is increasingly seen as a pathway toward a more sustainable and circular economy. Indeed, using treated wastewater directly in agriculture can reduce environmental impacts by about 33 % compared to diluting untreated wastewater in surface water before use (Miller-Robbie et al., 2017). However, most research in this field remains qualitative, with few studies quantifying the environmental impacts. As shown in Table 5 existing studies fall into two main groups. Some focus on the sustainability of irrigation systems using drip, sprinkler, surface, or furrow methods. Others examine the environmental impacts of wastewater reuse, including treated urban effluents or raw wastewater under regional water policies.

However, in such analyses, the environmental burden of wastewater treatment itself is attributed to reclamation, with impact values varying depending on the treatment method applied—for instance, 1.034 kg CO_2eq./m^3 for secondary treatment and up to 7.0 kg CO_2eq./m^3 for more advanced treatment processes (Moretti et al., 2019; Pan et al., 2019). These figures are broadly consistent with the findings of the present study, which reported impacts ranging from 0.11 to 0.58 kg CO_2eq./m^3 for ALF and CENTA, respectively.

4.2. *The performance of constructed wetlands in terms of carbon capture*

At the CENTA facility, the operational carbon footprint of CWs was estimated at 0.58 kg CO_2eq./m^3. LCA studies show wide variation in CW carbon footprints, from 0.05 to 5.30 kg CO_2eq./m^3 of treated water (San Miguel et al., 2023). For instance, Yin et al. (2025) reported 0.18 kg CO_2eq./m^3 for a single CW, while Andreo-Martínez et al. (2021) found values between 0.10 and 0.21 kg CO_2eq./m^3. Higher impacts were observed by Flores et al. (2020): 0.9–1.2 kg CO_2eq./m^3 for winery wastewater, and by Garfí et al. (2017): 0.6–1.3 kg CO_2eq./m^3.

Apart from the number of CWs and wastewater composition, there are also differences based on the type of vegetation used in the CWs. Polyculture CWs have been claimed to have highest impacts, up to 12.2 kg CO_2eq./m^3, compared with 1.3–8.1 kg CO_2eq./m^3 for monocultures. This is caused by a larger quantity of CH_4 and N_2O emissions, which have originated from the presence of more networks with the roots and thus, the creation of anoxic zones that favor the presence of methanogens (Carrillo et al., 2023). Other studies, however, defended that there are not significant difference in CH_4 emissions between monocultures and polycultures (Maucieri et al., 2017). While there are controversies among studies regarding with the selection of the type of cultures, direct air emissions (principally N_2O, CO_2, and CH_4) seem to be a critical hotspot, also for this study. These may be influenced through targeted manipulation of microbial communities involved in nutrient cycling and organic matter removal (Liu et al., 2025). In addition to microbial inoculation, pollutant degradation pathways are conditioned by substrate and media composition (e.g., biochar, activated carbon), the supplementation of external carbon sources such as plant-derived materials and biodegradable polymers, and the incorporation of materials supporting autotrophic denitrification (Tao et al., 2025; Yang et al., 2025).

When incorporating the effects of the carbon capture, CENTA reported a net global warming potential of 0.12 kg CO_2eq./m^3. This suggests that the facility with CWs as nature-based solution does not act still as carbon sink caused by the large quantity of direct emissions to be offset. Similarly, Wang et al. (2018) found that CO_2 fixation is small

Table 5
Climate change quantification of strategies for irrigation in agricultural activities based on Life Cycle Assessment studies results.

Reference	Stage	Objective	Climate Change (Original FU)	Climate Change (per m^3)
Lukkoor et al. (2025)	Agricultural irrigation systems	Comparison of flood and drip irrigation using surface and groundwater	Drip irrigation: 0.42 kg CO_2eq./kg in-shell pecans Flood irrigation: 0.07 kg CO_2eq./kg in-shell pecans	Drip irrigation: 0.154 kg CO_2eq./m^3 Flood irrigation: 0.013 kg CO_2eq./m^3
Batoukhteh et al. (2025)		Comparison of furrow, sprinkler, and drip irrigation	Furrow irrigation: 798 kg CO_2eq./t wheat Sprinkler irrigation: 1749 kg CO_2eq./t wheat Drip irrigation: 623 kg CO_2eq./t wheat	Furrow irrigation: 0.51 kg CO_2eq./m^3 Sprinkler irrigation: 0.93 kg CO_2eq./m^3 Drip irrigation: 0.86 kg CO_2eq./m^3
Parada et al. (2021)		Benefits of fertigation management practices with drip irrigation	Traditional drip irrigation: 0.38 kg CO_2eq./kg tomato Drip irrigation with drained water recirculated to the system: 0.334 kg CO_2eq./kg tomato Drip irrigation with water reduction and recirculation: 0.323 kg CO_2eq./kg tomato	Open system: 5.04 kg CO_2eq./m^3 Recirculated control system: 6.38 kg CO_2eq./m^3 Recirculated reduction system: 6.65 kg CO_2eq./m^3
Mehmeti et al. (2016)		To assess the "Sinistra Ofanto" irrigation scheme	Micro-sprinklers with diesel engine pumps: 89,296,134 kg CO_2eq./y Drip irrigation: 88,430,503 kg CO_2eq./y Subsurface drip irrigation: 88,430,503 kg CO_2eq./y Micro-sprinklers with solar powered pumps: 72,315,546 kg CO_2eq./y Micro-sprinklers with smart technologies: 86,704,855 kg CO_2eq./y	Not available
Canaj et al. (2021a)	Reuse of wastewater in agriculture	Comparison of reuse with and without tertiary treatment	Direct secondary wastewater and groundwater mixture: 0.21 kg CO_2eq./m^3 Reuse after tertiary treatment: 0.41 kg CO_2eq./m^3	Direct secondary wastewater and groundwater mixture: 0.21 kg CO_2eq./m^3 Reuse after tertiary treatment: 0.41 kg CO_2eq./m^3
Pan et al. (2019)		To evaluate three wastewater treatment alternatives for irrigating crops	Secondary treatment and then reused: 4.0 kg CO_2eq./m^3 Secondary effluents with nutrient removal: 4.25 kg CO_2eq./m^3 Secondary effluents with tertiary treatments: 7.00 kg CO_2eq./m^3	Secondary treatment and then reused: 4.0 kg CO_2eq./m^3 Secondary effluents with nutrient removal: 4.25 kg CO_2eq./m^3 Secondary effluents with tertiary treatments: 7.00 kg CO_2eq./m^3
Moretti et al. (2019)		To assess treated municipal wastewater and surface water	Treated municipal wastewater: 0.209 kg CO_2eq./kg nectarines Surface water: 0.0196 kg CO_2eq./kg nectarines	Treated municipal wastewater: 1.034 kg CO_2eq./m^3 Surface water: 0.051 kg CO_2eq./m^3
Miller-Robbie et al. (2017)		Comparison of untreated wastewater diluted in surface streams with wastewater treatment with direct reuse in agriculture	Uncontrolled released: 5.50 kg CO_2eq./y Release to stream: 3.80 kg CO_2eq./y Effluent reuse in existing farms: 3.80 kg CO_2eq./y Effluent reuse in surrounding land: 2.00 kg CO_2eq./y	Uncontrolled released: $8.37 \cdot 10^{-7}$ kg CO_2eq./m^3 Release to stream: $5.78 \cdot 10^{-7}$ kg CO_2eq./m^3 Effluent reuse in existing farms: $5.78 \cdot 10^{-7}$ kg CO_2eq./m^3 Effluent reuse in surrounding land: $3.04 \cdot 10^{-7}$ kg CO_2eq./m^3
Arzate et al. (2019)	Tertiary treatments for wastewater reuse	Three scenarios were considered for the comparison of	No tertiary treatment with groundwater supply: 0.372 kg CO_2eq./m^3 Ozonation with sand	No tertiary treatment with groundwater supply: 0.372 kg CO_2eq./m^3 Ozonation

(continued on next page)

Table 5 (*continued*)

Reference	Stage	Objective	Climate Change (Original FU)	Climate Change (per m^3)
		wastewater reclamation	filtration: 0.235 kg CO_2eq./m^3 Photo-Fenton with sand filtration: 0.5 kg CO_2eq./m^3	with sand filtration: 0.235 kg CO_2eq./m^3 Photo-Fenton with sand filtration: 0.5 kg CO_2eq./m^3

compared to electricity-related emissions, leading to a net positive impact too.

5. Conclusions

An LCA conducted on four case studies in the Iberian Peninsula yielded the following conclusions: (1) Energy consumption during the operational phase is the main contributor to most environmental impact categories, regardless of whether the system is technology- or nature-based; (2) Direct emissions substantially influence specific categories, particularly climate change and freshwater eutrophication; (3) The ALF scenario shows the most favorable environmental performance, mainly due to reduced energy demand and lower indirect water demand; (4) Results for freshwater eutrophication are highly sensitive to local characterization factors, as they are strongly modulated by direct aquatic emissions; and (5) among the alternatives, CENTA, which represents the only nature-based solution, shows superior potential for achieving carbon neutrality, as it requires only 0.79 ha of cropland to offset operational emissions, in contrast to the 566 ha required by GAVA. Based on these results, future research should focus on expanding the system boundaries to determine whether the wastewater recovery performance of the facility is maintained when agricultural impacts are integrated in the evaluation.

CRediT authorship contribution statement

Helena Feijoo: Writing – original draft, Validation, Methodology, Investigation, Formal analysis. **Sofía Estévez:** Writing – original draft, Visualization, Methodology, Investigation, Formal analysis. **Sara González-García:** Writing – review & editing, Validation, Supervision. **María Teresa Moreira:** Writing – review & editing, Validation, Supervision.

Declaration of competing interest

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

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Data availability

Data will be made available on request.

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