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Techno-environmental and economic assessment of color removal strategies from textile wastewater

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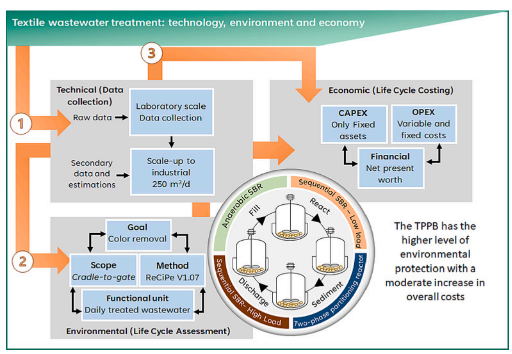
HIGHLIGHTS

- The type of dye, its concentration and removal efficiency are parameters that influence ecotoxicity.
- The two-phase partitioning bioreactor performs better in terms of ecotoxicity.
- The effects of lower pH and the use of blowers are offset by higher dye removal.
- Chemical dosing control should be tighter for higher loading rates.
- Polymer prices are expected to increase investment costs by 0.6 to 8.3 %.

GRAPHICAL ABSTRACT



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ABSTRACT

The textile industry is one of the most chemical-intensive processes, resulting in the unquestionable pollution of more than a quarter of the planet's water bodies. The high recalcitrant properties of some these pollutants resulted on the development of treatment technologies looking at the larger removal efficiencies, due to conventional systems are not able to completely remove them in their effluents. However, safeguarding the environment also implies taking into account indirect pollution from the use of chemicals and energy during treatment. On the other hand, the emerged technologies need to be economically attractive for investors and treatment managers. Therefore, the costs should be kept under control. For this reason, the present study focuses on a comparative Life Cycle Assessment and Life Cycle Costing of four scale-up scenarios aiming at mono and di-azo reactive dyes removal from textile wastewater. Two reactors (sequencing batch reactor and two-phase partitioning) were compared for different reaction environments (i.e., single anaerobic and sequential anaerobic-aerobic) and conditions (different pH, organic loading rates and use of polymer). In accordance with the results of each scenario, it was found that the three technical parameters leading to a change in the environmental profiles were the removal efficiency of the dyes, the type of dye eliminated, and the pollutant influent concentration. The limitation of increasing organic loading rates related to the biomass inhibition could be overcame through the use of a novel two-phased partitioning bioreactor. The use of a polymer at this type of system may help restore the technical performance (84.5 %), reducing the toxic effects of effluents and consequently decreasing the environmental impact. In terms of environmental impact, this is resulting into a reduction

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of the toxic effects of textile effluents in surface and marine waters compared to the homologous anaerobic-aerobic treatment in a sequencing batch reactor. However, the benefits achieved for the nature comes with an economic burden related to the consumption of the polymer. It is expected that the cost of investment of the treatment with the two-phase partitioning bioreactor rises 0.6–8.3 %, depending on market prices, compared to the other analyzed sequential anaerobic-aerobic technologies. On the other side, energy and chemical consumption did not prove to be limiting factors for economic feasibility.

1. Introduction

Increasing industrialization resulting from rapid population growth is having numerous negative consequences on ecosystems (i.e., eutrophication, acidification and toxicity) and human health (i.e., cancer, nervous problems and dermatitis) (Al-Tohamy et al., 2022). Among this industrial network, textile factories have some of the most chemical and pollutant intensive processes caused mainly by the activities performed during the dyeing and finishing stages (Drumond Chequer et al., 2013). As a result of its operation, there is a heavy demand for resources and emissions released into the environmental compartments. It is estimated that the sector represents the fourth highest pressure category after food, housing and transport in resource consumption, it is responsible for 10 % of global carbon emissions and for polluting 20 % of global water resources (European Parliament, 2023; Greer et al., 2015). This is because in some of the major textile countries such as China, India, Pakistan, Brazil, Bangladesh and Malaysia discharge their effluents directly into rivers. Globally, it is estimated that 52 % of the global average of wastewater produced is treated, although this percentage in South Asia decrease to 16 %, which is especially problematic taking into account this region also has the highest population share (31 %) and the textile industry is a key sector (Jones et al., 2021). The continuous discharge to the environment of complex wastewater containing aromatic hydrocarbons, heterocyclic dyes, high suspended solids, heavy metals (Cu, Cr, As and Zn), sulphates, oils, surfactants and chlorides arises as a major environmental issue (Drumond Chequer et al., 2013; ECWRTI, 2020; Wang et al., 2022). The degradation of non-biodegradable compounds, such as dyes, is particularly complicated due to their solubility in water and their recalcitrant nature, which hinder their elimination by conventional methods (Lellis et al., 2019).

To increase the level of protection of the water bodies, a wide range of wastewater treatments are nowadays available including physical, chemical, biological and advances oxidation methods (Christian et al., 2023). Their selection depends mainly on the characteristics of the wastewater, although other aspects such as space limitations, treatment capacity, process flexibility, economic sustainability, commercial availability, energy and chemical demand, inhibitory effects and final waste management can also be taken into account. Among them, biological processes have gained attention since they are claimed to be economically viable, environmentally friendly, suitable for a wide range of dyes and industrial scale application (Holkar et al., 2016). Despite drawbacks such as inhibition in the presence of toxic substances, long treatment times and possible foaming, biological treatments are capable of removing dyes that physicochemical methods cannot such as acid, reactive and direct dyes (Adane et al., 2021). Besides, many methods such as adsorption, chemical precipitation or membrane filtration concentrate pollutants into other streams instead of contributing to their complete mineralization. Other disadvantages can be the need for dewatering, pH modification, high energy cost and sludge generation (Ceretta et al., 2021; Zakaria et al., 2023).

As far as advanced oxidation methods are concerned, the removal of non-degradable compounds is an advantage shared with biological methods. However, and despite the higher flexibility of the process, they might have (depending on the technology) mass transfer limitations, higher resource demand for operation and more costly implementation (Zhang et al., 2021). Although the aforementioned generic advantages may open a possibility for the use of biological methods, the efficiency of the methods would depend on many other specifics such as dye structure and concentration, carbon/nitrogen sources, temperature, pH, dissolved oxygen (anaerobic, aerobic or facultative conditions), but also on the type of microorganism (fungus, bacteria, yeast or algal) or enzymes used in particular on their activity and adaptability (Jamee and Siddique, 2019).

Anaerobic treatment of wastewater streams has reported to be a sustainable and circular practice due to the transformation of the methane generated into electricity and/or heat (Obaideen et al., 2022). Besides, it is a more energy-efficient alternative and produces less biological sludge (Laizer et al., 2022). Nevertheless, its stand-alone use in textile wastewater treatment has been limited by the diluted concentration of influents and formation of intermediary aromatic compounds (Işik and Sponza, 2008). Therefore, even if the color can be anaerobically removed from the textile industry effluents, the toxicity hazard of streams discharged to aquatic environments is a critical issue. The combination of anaerobic and aerobic steps has been verified as promising solution capable of achieving decolorization, dye mineralization as well as reduction of energy demand (Lourenço et al., 2000). In terms of reactor configurations, membrane bioreactors, Sequencing Batch Reactor (SBR), baffled reactors, up-flow anaerobic blanket reactors, moving bed biofilm reactors and continuous stirred tank reactors have been used for the anaerobic/aerobic stages (Azimi et al., 2021). Among them, the SBR is characterized by high operational flexibility, easier design adaptability to meet effluent discharge standards, and low construction and maintenance costs (Kathawala et al., 2021). Tomei et al. (2016b) have demonstrated that the anaerobic-aerobic sequential operation of the SBR in treating textile bath wastewater is capable of achieving dye removal efficiencies of 70–80 %.

However, increasing organic loading rates may expose the biomass to toxic effects. The addition of an amorphous polymer can effectively reduce the self- inhibitory effects of the toxic substrate. In this case, the bioreactor functions as a two-phase partitioning bioreactor (TPPB) in which the polymer is capable of retaining high amounts of contaminants, which are gradually released into the bulk phase of the bioreactor following a process driven by bacterial metabolism (Amsden et al., 2003). This configuration will avoid/reduce the inhibitory and toxic effects exerted on the biomass by xenobiotic compounds such as dyes.

As important are the direct emissions of pollutants from a facility as the environmental loads associated with its treatment/removal process and therefore a connection between the two must be established to minimize the overall effects (Grösser et al., 2017). To achieve this goal, Life Cycle assessment (LCA) is one of the most recurrently used methodologies among others such as Environmental Impact Assessment, and Net Environmental Benefit Analysis (Rashid et al., 2023). Although in the wastewater sector, LCA applications were mainly directed to domestic treatment, technologies that favored the improvement of industry effluent quality were also subjected to evaluation. In this regard, several dye treatment options were considered for evaluation with the LCA methodology i.e. a fungal biological process with activated carbon (Gabarrell et al., 2012), the use of a $ZnCl_2$ modified bio-adsorbent in a batch system (Maiti et al., 2023), conventional activated sludge process with activated carbon filters and filtration/reverse osmosis units (Nakhate et al., 2020), Fenton-based processes followed by aerobic sequencing batch reactors (García-Montaño et al., 2006), electrochemical and electrocoagulation systems (Ahangarnokolaei et al., 2021; Álvarez et al., 2020), membrane capacity deionization with reverse osmosis (Cetinkaya and Bilgili, 2019) and moving bed biofilm reactor-membrane biorreactor system (Yang and López-Grimau, 2021).

It should be noted that standard LCA practices only consider one of the three pillars (environmental, economic, and social) of sustainability. In order to provide a robust assessment towards a sustainable operation, the technical modeling and data collection performed for LCA can also partially be a source of information to complete an economic assessment (Rashid et al., 2023). Previous studies on the economic assessment of technologies for textile wastewater treatment have been mainly focused on chemical oxidation processes (El-Dein et al., 2006), membrane biorreactor, moving bed biofilm reactor (Yang et al., 2020a), electrochemical processes, and solar electrophoto-Fenton and membrane processes (Ranga and Sinha, 2023).

To our knowledge, there are no previous publications on LCA and Life Cycle Costing (LCC) of sequential anaerobic-aerobic processes and TPPBs for dye removal. With the aim of contributing to the current state of knowledge, in this study we evaluated the environmental and economic performance of four scenarios for biological mono and di-azo dye removal in SBRs and TPPBs. First, we considered the comparative evaluation of anaerobic and sequential anaerobic-aerobic operation, then the impacts of increasing organic loading rate on the sequential process, and finally, the efficiency of a TPPB operated in sequential mode. Since laboratory data have been collected from previous research and adapted to full-scale operation, prospective LCA and LCC approaches were considered in order to have a broad overview of the full-scale performance of the different technological solutions that are ready for application.

2. Methodology and methods

2.1. Scale-up and proposal of scenarios

The present study presents a comparative environmental and economic evaluation of wastewater treatment scenarios aiming at biological color removal from textile industry effluents. The research is based on the results obtained from previous reports (Tomei et al., 2016a, 2016b) on a segregated stream mixture containing reactive dyes such as Remazol Black 5, Remazol Yellow RR and Remazol Brilliant Red 21 originated from the effluent from the dyeing bath of an Italian textile factory. The quantitative information gathered from these studies, as shown in Table 1, has been scaled-up and completed with other primary data to create the inventories corresponding to the LCA and LCC methodologies. In this regard, prospective analyses aimed at determining the future impacts of the tested technologies have been provided. For such reasoning, the study should be replicated after full-scale implementation. In the reference plant, dyes are removed by an SBR operating on a 24-hour cycle and including feeding, reaction, settling and effluent discharge. Four scale-up scenarios are proposed according to the operational characteristics listed in Table 1. In the four scenarios analyzed, the SBRs have been operated under different reaction environments (i.e., single anaerobic and sequential anaerobic-aerobic) and operating conditions (different pH, organic loading rates and use of polymer). This change in performance between scenarios was intended to answer the following hypotheses:

- Based on the better dye degradation claimed to be offered by TPPB, its application will result in better environmental protection compared to other technologies such as anaerobic and sequential anaerobic-aerobic treatments.
- The lower energy consumption of the anaerobic system provides a strong environmental benefit compared to anaerobic-aerobic SBR and TPPB. This hypothesis is based on the results of typical domestic wastewater treatment comparing anaerobic and aerobic operations and the relevance of energy use in the environmental profile. Examples could be the studies of Ranieri et al. (2021) and Mishra et al. (2021).

Table 1
Equipment operational pre-fixed parameters for the design of the treatment facility.

Technical parameters	Units	AnS	SLS	SHS	TPPB
Temperature	°C	27.00	27.00	27.00	27.00
Initial pH	–	7.00	7.00	7.00	4.57
Exchange ratio	–	0.30	0.30	0.30	0.30
Organic loading rate	$kg_{COD}/m^3{\cdot}d$	0.13	0.13	0.16	0.16
Organic loading rate for the dye	$kg_{COD}/m^3{\cdot}d$	0.005	0.005	0.01	0.01
Removal efficiency of COD	%	95.69	96.00	96.75	98.80
Removal efficiency of dye	%	65.47	90.58	71.47	84.50
Biomass concentration	mg VSS/L	3970.00	3800.00	3100.00	2655.00
Feeding time	min	20.00	20.00	20.00	20.00
Anaerobic reaction time	min	1360.00	680.00	680.00	680.00
Aerobic reaction time	min	0.00	680.00	680.00	680.00
Settling time	min	40.00	40.00	40.00	40.00
Effluent discharge time	min	20.00	20.00	20.00	20.00

Acronyms: Anaerobic SBR reactor (AnS), Chemical Oxygen Demand (COD), Sequencing anaerobic-aerobic operating with High dye load in an SBR reactor (SHS), Sequencing anaerobic-aerobic operating with high dye Load in an SBR reactor (SLS) and Two-Phase Partitioning Bioreactor (TPPB).

- Since the increase in organic loading rate results in biomass inhibition, the removal efficiency of the dye is likely to decrease and thus dye emissions to the environment will increase. This raises the question of whether or not the influent concentration of the dye increases the environmental impact proportionally.
- The optimized operation of TPPB is conducted at pH 4 to ensure the maximum sorption rate of the dye through the polymer, which means that more chemicals are needed for pH adjustment. Therefore, the increased indirect impacts associated with chemicals may lead to reduced environmental sustainability of TPPB.
- The increased energy consumption in the aerobic phase, the use of more chemicals to reduce the pH in TPPB and the use of a polymer to control the ability to release the contaminants to the biomass represents an impact on the economic viability of the new technology.

Based on the above assumptions and in order to confirm the hypothesis considered, an industrial scale facility was designed to treat 250 m^3/d of dye effluent of a textile factory. The flow rate was selected among the existing capacities of Italian wastewater treatment plants constructed within the sector (Abiri et al., 2017; Negri et al., 2020; Rozzi et al., 1999). To avoid heights of reactor vessels higher than 10 m, the configuration of each scenario is constituted by a system of four 208.33 m^3 parallel bioreactors performing all the stages of operation from feeding to discharge (Turton et al., 2018). A schematic representation of the plant is reported in Fig. 1.

Since the physical configuration of the scenarios is analogous, the difference between them is basically related to the reaction phase, which depends on the process kinetics, organic load, and polymer addition. The first scenario is named as AnS and works with an anaerobic SBR with 1360 min reaction phase and is fed with 0.005 kg COD/m^3·d dye load, while the second option or scenario SLS has been designed for the same load but with the sequential alternative use of anaerobic and aerobic treatment. Another scenario (SHS or “sequential anaerobic-aerobic operating with high dye load”) has been defined with similar configuration than SLS but doubling the organic loading rate of the dye in the influent. The last scenario consists of a TPPB operated with Hytrel polymer (Du Pont). All above-mentioned scenarios were compared in pairs and thus, a univariate analysis was performed. Therefore, the

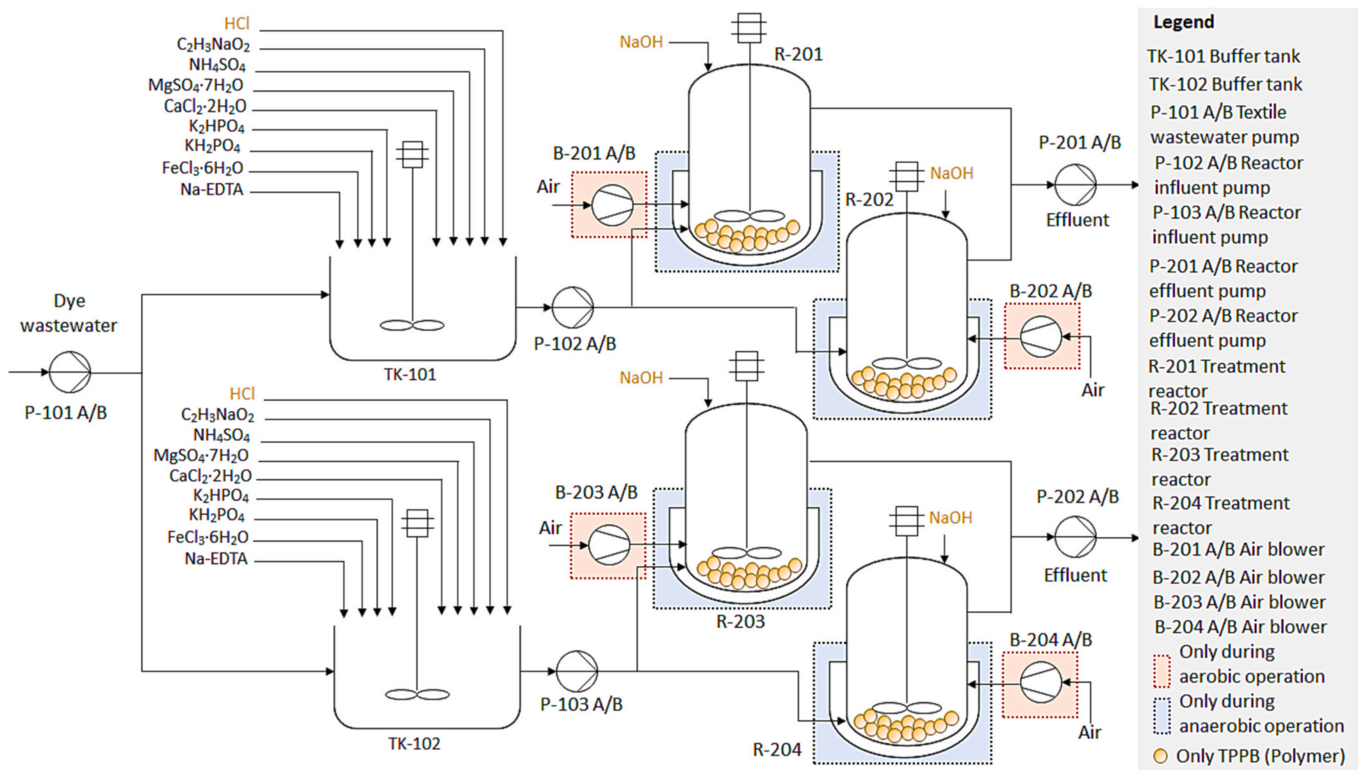

Fig. 1. Generic scheme of the proposed alternatives under assessment.

effects derived from the use of oxic and anoxic reaction environments, the loading rate and polymer addition were progressively and independently analyzed through environmental and economic perspectives.

A remark on the size of the bioreactors should be done since their working volume comprises not only the volume required for the influent stream but also the residual volume derived from the exchange ratio of 0.3. Each pair of SBRs is fed with 125 m^3 of liquid volume from a storage tank, which provides adequate mixing (60 s for typical rapid mixing operations) of the chemicals added. Besides, this previous storage of the influent is also necessary to ensure a minimum residence time to ensure continuous operation of the facility despite the batch performance of the reactor (Metcalf and Eddy, 2014). These storage tanks are named as TK-101 and TK-102 in Fig. 1 and are filled by a single continuously operating centrifugal pump, while discharged by two independent batch pumps. It should be noted than two more are being considered as a backup for malfunction and maintenance situations. Similarly, two other pumps were used to discharge the liquid from the bioreactors at the end of each work cycle. The energy consumption of the five centrifugal pumps was estimated according to the guidelines of Sinnot and Towler (2020) assuming a distance between operation units of 5 m (CCPS, 2018) and no height difference. Other physical devices such as heating jackets, blowers and stirrers were included. The jackets ensured the temperature maintenance in the anaerobic conditions up to 27 °C. The reactor was assumed to be well insulated and, therefore, the estimated energy demand is associated with the temperature rise and not with losses through the vessel walls. Blowers would supply air to ensure that the reactor medium had oxidative metabolism and, unlike heating and air supply, stirring was used during both reaction stages. Stirrer and blower requirements were determined based on the calculation procedure from Kresta et al. (2016), McCabe et al. (2007) and Metcalf and Eddy (2014). In the latter case, a growth yield coefficient for heterotrophic bacteria of 0.1 g VSS/g COD_{used} was considered as well as an endogenous decay coefficient of 0.075 g/g·d (Farabegoli et al., 2010; Feng et al., 2010).

2.2. LCA framework

2.2.1. Goal and scope

The guidelines of ISO 14040 and 14044 were followed to provide an attributional LCA approach with "*cradle-to-gate*" boundaries, which were set according to the International Reference Life Cycle Data System (European Commission et al., 2010; ISO, 2006a, 2006b). These boundaries included the consumption of chemicals and energy (for electrical and heating needs) although a zero-burden assumption has been adopted for the wastewater influent using a cut-off by classification allocation for a waste with no economic value. In accordance with this information, Fig. 2 is subdivided into two bubbles or semi-spheres. The left side of the image is intended to provide knowledge about the background processes, in other words, all the above-named consumables. Their production process has been taken from the Ecoinvent 3.8 database and thus they are out of the control of the LCA practitioner (Wernet et al., 2016). The right side, on the other hand, encompasses the modelled scenarios belonging to the foreground. As indicated by the colorful flows of the left semi-sphere of Fig. 2, the background consumables feed the foreground and constitute indirect impacts coming from the Technosphere or human-made domain. On the other hand, direct emissions, such as methane produced during the anaerobic stages or dyes remaining in the effluent, originating from the process under study are part of the foreground. Consequently, methane and dye emissions are directly released to the environment or ecosphere (shown in yellow in Fig. 2).

In addition, the LCA has only focused on the operational phase and construction, demolition and maintenance have been left out of the system boundaries. This is because the construction of wastewater treatment facilities involves data collection on materials (e.g., concrete, steel, or plastics) and activities (e.g., use of machinery) that entail detailed design, which is therefore outside the scope of this study (Rashid et al., 2023).

Regarding the functional unit, two main groups can be differentiated within the state of the art of LCA and textile wastewater treatment:

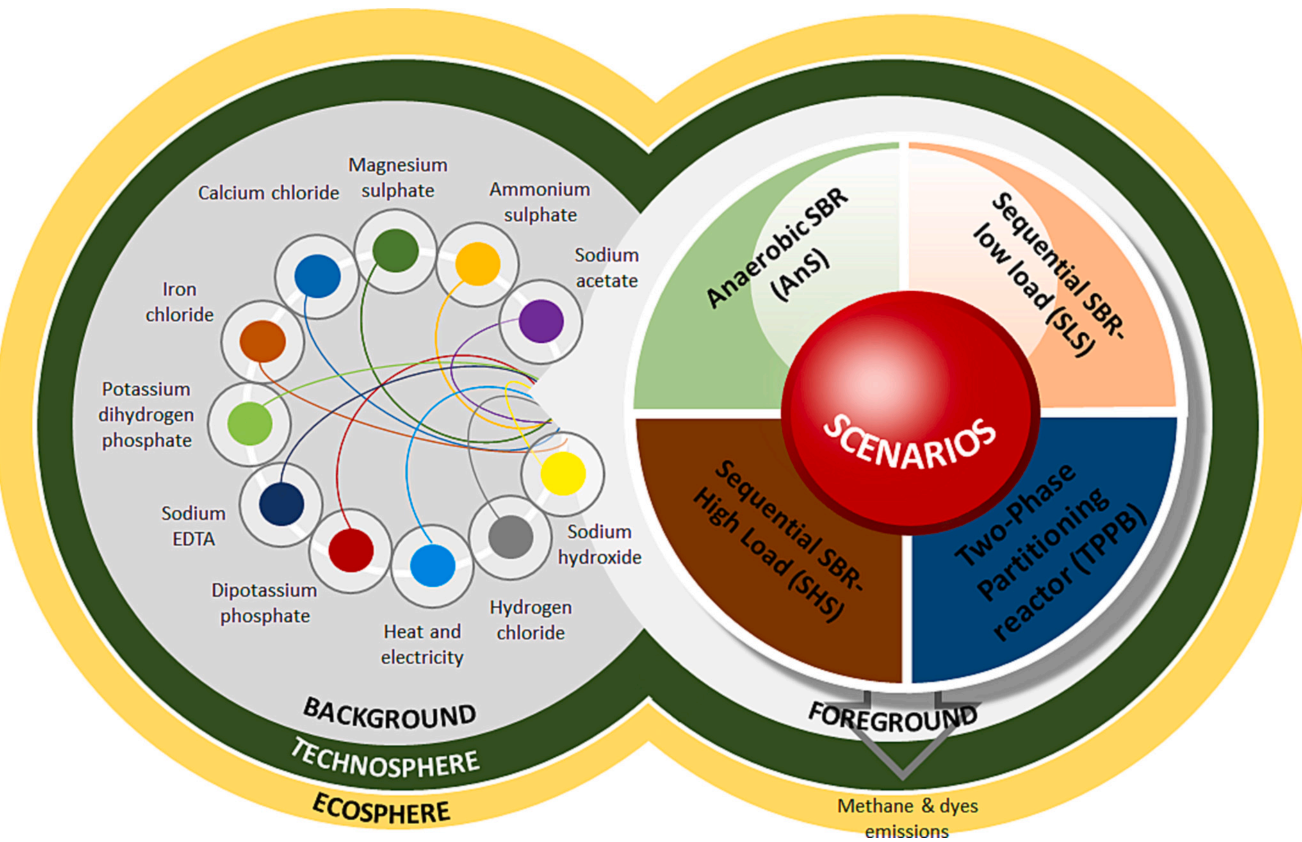

Fig. 2. Definition of the Life Cycle Assessment system boundaries.

volume of wastewater treated and pollutant removal efficiency. This last term been expressed in terms of percentage with respect to the color/pollutant concentration in the influent. Both functional units are a quantitative representation of the function of the systems under analysis and are useful as reference in the estimation of the environmental impact during the Life Cycle impact assessment stage (Arzoumanidis et al., 2020).

In line with recent reports shown in Table 2 and the main goal of the study, the designated functional unit is the "*daily treated wastewater*" of the 4-batch parallel reactor systems. It differs from those already reported in the literature, but taking into account that the presented scenarios were scaled with the same capacity and influent composition, it is useful when establishing synergies between LCA and LCC (as costs are usually expressed per unit time). Apart from the functional unit, Table 2 also presents some other LCA characteristics for different technologies applied to textile wastewater treatment such as the categories analyzed, the method and supporting program used for the impact assessment stage.

2.2.2. Environmental life cycle inventory

A process-based Life Cycle Inventory was created for each of the scenarios following a "bottom-up" approach, as shown in Table 3 per functional unit. The inventoried input and output flows were classified in relationship to their origin: the human domain (Technosphere) or the environment (Ecosphere). The interpretation of Table 3 can be supported also on the descriptions of Fig. 2 since the chemical and energy resources used come from the Technosphere, while emissions are released into the Ecosphere or nature. The Technosphere compounds were also subdivided in Table 3 into chemicals and energy. Table 3, for example, indicates that sodium acetate is a chemical coming from the Technosphere (as it is a material coming from other manufacturing process) used in all the four analyzed scenarios. Direct emissions affect both the atmosphere and aquatic environments, as they are released in the form of CO_2, CH_4 and COD due to the degradation of the dye and its persistence in the effluent.

It is important to highlight that the CO_2 emitted to the atmosphere from the biological degradation of the organic compounds of the wastewater can be assumed either biogenic or non-biogenic. This is because the origin/production of some wastewater components, such as sodium acetate and textile dyes, can be either produced by synthetic or biogenic-based processes. The Intergovernmental Panel of Climate Change or IPCC guidance states that a corresponding analysis of the influent sources should be made for wastewater treatment plants to determine whether the greenhouse emissions derived from their processes can be categorized as biogenic (Ye et al., 2022). Accordingly, the CO_2 generated for the analyzed scenarios were considered biogenic, as commonly assumed for domestic wastewater treatment plants. However, IPCC concerns were addressed by a subsequent sensitivity in which these emissions were managed as non-biogenic.

An observation must be made with respect to dyes emitted to water courses, as the inventoried SimaPro® chemical databases are still insufficient to provide definitive LCA impact assessments when it comes to reactive dyes. For this reason, the effects of four different dyes (acid violet 49, metanil yellow, methylene blue and Orange II) were compared to identify the magnitude of profile changes with the type of dye removed. Considering this, the suitability of the herein studied technologies should be further analyzed at in the future at large-scale after other investigations expand the database of LCA characterization factors for reactive dyes.

2.2.3. Life cycle impact assessment

The quantitative input-output flows of the scenarios in Table 2 are then transformed into a magnitude of contribution to the environment. For this purpose, two of the most widely used methods among the LCA community dealing with textile wastewater treatment were selected (Table 2): the ReCiPe MidPoint (H) V1.07/World (2010) and EndPoint

Table 2
LCA characteristics of the studies related to the removal of textile wastewater pollutants.

Technology	Categories	Method	Programme	Functional unit	Authors
Activated carbon adsorption and *Trametes versicolor*	CC, OD, HT, POF, TA, FE, ME, TET, FET, MET, MD, FD.	ReCiPe Midpoint	SimaPro 7.2.2	The removal of 90 % of the color from 1 m^3 of simulated effluent with 150 mg/L of the dye Grey Lanaset G.	Gabarrell et al. (2012)
$ZnCl_2$ modified bio-adsorbent in batch reactor	CC, FD, OD, HT, PMF, TA, FET, MET, MD and WD	ReCiPe Midpoint	OpenLCA 1.11.0	Not specified	Maiti et al. (2023)
Aeration, clarifier, activated carbon filters, ozonation, ultrafiltration and reverse osmosis	AD, AP, HTP, CC, FET, EP, MET and TET	CML Midpoint and ReCiPe Endpoint	GaBi 8.7	Two-fold functional units of 1500 m^3 and 1200 m^3	Nakhate et al. (2020)
Artificial light photo-Fenton process, solar driven photo-Fenton process and artificial light photo-Fenton coupled with aerobic sequencing batch reactor	ARD, CC, OD, HT, FET, MET, TET, POF, AP and AE.	CML Midpoint	Not specified	The removal of 80 % DOC from 1.2 L of 250 mg/L Cibacron Red FN-R synthetic effluent	García-Montaño et al. (2006)
Electrochemical and ultraviolet combined system	CC, OD, TA, FE, ME, HT, POF, PMF, TET, FET, MET, IR, ALO, ULO, NLT, WD, MRD and FD	ReCiPe Midpoint and ReCiPe Endpoint	SimaPro 7.3.3	6240 kg of wet cotton fabric dyed with reactive dyes in a Jet process	Álvarez et al. (2020)
Electrocoagulation and ozonation	CC, OD, IR, OFH, PMF, OFT, TA, FE, ME, TE, FE, ME, HCT, HNCT, LU, MD, FD, WD	ReCiPe Midpoint and ReCiPe Endpoint	SimaPro 9.1.0.8	1 L of treated dye wastewater	Ahangarnokolaei et al. (2021)
Membrane capacity deionization and Reverse Osmosis	AD, CC, OD, HT, MET, TET, POF, TA, AE, CN, NCN, RI, IR, OD, RO, AET, TET, TA, LO, AA, AE, CC, NREP and MEX	IMPACT 2002+ and CML	SimaPro	Not specified	Cetinkaya and Bilgili (2019)
Gingko biloba-wood membrane	HH, TA and AE	ReCiPe Midpoint	Gabi	30 × 30 × 5 mm^3, and the operational volume of the wastewater was 1 L, with an MB concentration of 50 mg/L	Niaz et al. (2020)
Photo-Fenton	CC, OD, TA, FE, ME, HT, POF, PMF, TET, FET, MET, IR, ALO, ULO, NLT, WD, MD, and FD	IPCC and ReCiPe Midpoint	OpenLCA 1.6	Decolorization (PD = 97 %), biodegradability (BOD_5/COD > 0.4) of a 100 mg/L of AO5 1 m^3 of wastewater	Belalcázar-Saldarriaga et al. (2018)
Electrochemical treatment assisted by UV irradiation.	CC, HH, OD, HT, POF, PMF, IR, TA, FE, ME, TET, FET, MET, ALO, ULO, NLO, WD, MD and FD.	ReCiPe Midpoint and ReCiPe Endpoint	SimaPro 7.3.3	1000 m^3 of uncoloured effluent	Buscio et al. (2019)
Hybrid Moving Bed Biofilm Reactor—Membrane Bioreactor and activated sludge treatment	CC, HH, OD, HT, POF, PMF, IR, TA, FE, ME, TET, FET, MET, ALO, ULO, NLO, WD, MD and FD.	ReCiPe Midpoint and ReCiPe Endpoint	SimaPro	1 m^3 of treated effluent	Yang and López-Grimau (2021)
Fenton process	TA, CC, FD, MD, WD, FET, MET, TET, FE, ME, HT, IR, ALO, NLT, ULO, OD, PMF, POF	IPCC and ReCiPe Midpoint	Not specified	1 m^3 of contaminated wastewater	Grisales et al. (2019)
Conventional activated sludge, membrane biorreactor and moving bed biofilm reactor	CC, HH, OD, HT, POF, PMF, IR, TA, FE, ME, TET, FET, MET, ALO, ULO, NLO, WD, MD and FD.	ReCiPe Midpoint and ReCiPe Endpoint	SimaPro	1 m^3 of treated effluent	Yang et al. (2020b)

Acronyms: Abiotic Depletion (AD), Abiotic Resource Depletion (ARD), Acidification Potential (AP), Agricultural Land Occupation (ALO), Aquatic Acidification (AA), Aquatic Ecotoxicity (AET), Aquatic Eutrophication (AE), Biological Oxygen Demand (BOD), Carcinogens (CN), Climate Change (CC), Chemical Oxygen Demand (COD), Dissolved Organic carbon (DOC), Eutrophication Potential (EP), Fossil Depletion (FD), Freshwater Ecotoxicity (FET), Freshwater Eutrophication (FE), Human Carcinogenic Toxicity (HCT), Human Health (HH), Human Non-carcinogenic Toxicity (HNCT), Human Toxicity (HT), Ionizing Radiation (IR), Land occupation (LO), Marine Ecotoxicity (MET), Marine Eutrophication (ME), Metal Depletion (MD), Mineral Extraction (MEX), Mineral Resource Depletion (MRD), Natural Land Occupation (NLO), Natural Land Transformation (NLT), Non-carcinogens (NCN), Non-Renewable Energy Primary (NREP), Ozone Depletion (OD), Ozone Formation-Human Health (OFH), Ozone Formation-Terrestrial ecosystems (OFT), Particulate Matter Formation (PMF), Photochemical Oxidant Formation (POF), Respiratory Inorganics (RI), Respiratory Organics (RO), Terrestrial Acidification (TA), Terrestrial Ecotoxicity (TET), Urban Land Occupation (ULO) and Water Depletion (WD).

(H/H) V1.07/World (2010). Besides, Microsoft® Excel and SimaPro® software version 9.4.0.2 were used (Huijbregts et al., 2017; Oele et al., 2022). The six most relevant midpoint categories of the ReCiPe method were defined after a comparison of the normalized results for each of the scenarios (see supplementary material): Human carcinogenic toxicity (HCT), Freshwater toxicity (FET), Marine toxicity (MET), Freshwater eutrophication (FE), Ionizing radiation (IR), Fossil resource scarcity (FRS) and Global Warming Potential (GWP).

In addition to normalization, weighting (the other optional LCA stage of the Life Cycle Impact Assessment step) was considered with a twofold rationale. On one hand, the midpoint outcomes were formulated in economic terms and, secondly, the analogous endpoint categories were aggregated into a single environmental damage score (easier to understand for non-LCA practitioners). The use of shadow prices of de Bruyn et al. (2010), which translate environmental revenues and drawbacks into external costs, allowed internalizing the environmental effects of specific actions/specific characteristics of each scenario in the LCC analysis.

2.3. LCC framework

2.3.1. Construction of the economic analysis

The LCC, a technique used in predicting the cost profile of assets, projects or processes, was performed in agreement with the requirements of the ISO standard 15686-5:2017 (ISO, 2017). The economic research performed for each of the scenarios can be classified

Table 3
LCA inventory expressed per FU (daily treated wastewater) for the operation of textile wastewater treatment scenarios.

Life Cycle Inventory materials	Units	AnS	SLS	SHS	TPPB
Inputs from Technosphere					
Chemicals					
Sodium acetate	kg/d	84.00	84.00	120.00	120.00
Ammonium sulphate	kg/d	36.00	36.00	96.00	96.00
Pentahydrate magnesium sulphate	kg/d	7.20	7.20	19.20	19.20
Calcium Chloride	kg/d	3.60	3.60	9.60	9.60
Dipotassium phosphate	kg/d	7.92	7.92	21.12	21.12
Potassium dihydrogen phosphate	kg/d	6.12	6.12	16.32	16.32
Hexahydrate iron chloride	kg/d	0.14	0.14	0.38	0.38
Ethylenedinitrilotetraacetic acid disodium salt	kg/d	0.14	0.14	0.38	0.38
Hydrochloric acid	kg/d	–	–	–	$6.67 \cdot 10^{-3}$
Sodium hydroxide	kg/d	–	–	–	$9.01 \cdot 10^{-3}$
Energy					
Stirrer for the reactor	kWh/d	607.62	607.62	607.62	607.62
Stirrer for the storage	kWh/d	0.15	0.15	0.15	0.15
Blower	kWh/d	0.00	$3.60 \cdot 10^{-3}$	$5.60 \cdot 10^{-3}$	$1.12 \cdot 10^{-2}$
Heating	kWh/d	2167.41	2167.41	2167.41	2167.41
Influent pump	kWh/d	2.28	2.28	2.28	2.28
Reactor feeding pump	kWh/d	0.52	0.52	0.52	0.52
Effluent pump	kWh/d	0.52	0.52	0.52	0.52
Outputs to the Ecosphere					
Emissions					
Methane	kg/d	24.71	24.71	30.41	30.41
Carbon dioxide[a]	Kg/d	45.20	45.20	55.63	55.63
Dye[b]	kg/d	0.18	$4.37 \cdot 10^{-2}$	0.30	0.11

Acronyms: Anaerobic SBR reactor (AnS), Sequencing anaerobic-aerobic operating with High dye load in an SBR reactor (SHS), Sequencing anaerobic-aerobic operating with high dye Load in an SBR reactor (SLS) and Two-Phase Partitioning Bioreactor (TPPB).

[a] The carbon dioxide emissions were only incorporated to the analysis when the carbonaceous compounds of the wastewater were assumed to come from fossil resources.

[b] Although the amount of dye remains invariable, acid violet 49, metanil yellow, methylene blue and Orange II are the four species used for the sensitivity analysis.

primarily as conventional or financial, since it predominantly attends to investment costs rather than external costs, those related to environmental or social issues (Hoogmartens et al., 2014). In this regard, LCC includes not only the operational phase, but also construction and maintenance and thus reflects both the day-to-day costs and the effects of facility sizing. End-of-life costs related to decommissioning/demolition of the plant were left out of the system boundaries, as the financing of the infrastructure was estimated for its 30-year lifetime (Spiller et al., 2015).

The economic implications of the environmental impact of the facility were also addressed by integrating the LCA outcomes into the LCC. However, the study cannot be entirely classified as full environmental life cycle costing since the assumptions of the LCA analysis mentioned in Section 2.2.1 restrict the monetization of environmental costs related to the construction phase. Therefore, and in accordance with the above, the analysis integrated aspects such as capital expenditure, operational expenditure, working capital and operational environmental costs. The estimation of capital expenses was based on the cost estimation framework for chemical industries published elsewhere such as Green and Perry (2008), Turton et al. (2018) and Sinnot and Towler (2020). Thereupon, the goal of the scenario benchmarking was to gain insights on the development of economics on technologies with a low technology readiness level. In fact, the costs estimation included in this manuscript can be classified according to the Recommended Practices (RP) No. 17R-97, and RP No. 18R-97 of the Advancement of Cost Engineering International within the group 5, project level definition of 0–2 % and expected cost accuracy of 4–20 % (AACE, 2020a, 2020b).

Factor methods are among the most recurrent practices for low-level definition projects and use the historical delivered or purchased cost of equipment. Consistently, the initial investment was estimated from the delivered base costs from Woods (2007) which were conveniently adapted with material, temperature, pressure, timeframe (using the Chemical Engineering Cost Plant Index), and Wroth factors (referring to the installation of the equipment on-site). The aggregation of the costs of each of these constitutes the fixed assets of the initial investment and includes equipment devises such as reactors, storage tanks, pumps, blowers and stirrers. Considering that the non-fixed assets associated with the patenting of the technology, or the software used for process monitoring have the same impact in all scenarios and that already published data are available (and therefore there is no possibility of patenting them), they were left out of the system boundaries.

Although not part of the initial investment, capital expenditure has also covered the nominal acquisition cost of equipment to be replaced every 10 years over the lifetime of the treatment plants (i.e., pumps, agitators and blowers), as well as their depreciation. In this matter, a constant rate of 15 % was assumed with an end-of-life residual value of zero (EY, 2014; Zhou et al., 2021). On the operating side, costs were broken down into variable and fixed costs. While the first group covers the chemical, electrical, and environmental costs, the second one is incorporating to the analysis the labour, equipment maintenance, insurance, company-paid salary overhead, supervision, land rent, and plant overhead. Since the facilities are expected to be in operation for a 30-year period of time, the increase in variable and fixed costs has been established as a nominal annual rate of increase dependent on the inflation factor or Consumer Price Index. To consider this price increase, the constant average Consumer Price Index of 2.07 % for Europe over the last 20 years has been used (European Central Bank, 2023a).

Finally, in order to be able to operate successfully, the working capital or cash flow to be paid to meet the short-term financial liabilities was determined. The storage of chemicals before their use, the pending payment to material suppliers and the operating cash flow related to the management risks assumed by the company while operating were the aspects considered for their estimation. All of them are directly linked to both variable and fixed costs. It was assumed that one month storage time of materials and delay in payment to suppliers was sufficient to meet the minimum short-term liabilities of the facility (Environmental Protection Agency, 2000).

Apart from all this, the following two particularities of the study should be taken into consideration. First, if the treated wastewater is not recovered, there would be no marketable products and, therefore, no revenues. On the other hand, and in relation to Section 2.2.3, the current set of weighting factors from de Bruyn et al. (2010) was applied to translate the environmental impacts of the LCA to costs. However, the use of this dataset has a limitation: not all categories have been mapped. Emission factors have been provided for climate change, ozone depletion, acidification, photo-oxidant formation, particulate matter formation, marine eutrophication and freshwater eutrophication but the effects to the categories of freshwater and marine ecotoxicity cannot be found.

2.3.2. Economic inventory

The Life Cycle Inventory for the LCC methodology for the operational phase of the facility is analogous to the one shown in Section 2.2.2 for the LCA. Other important input parameters associated with the construction phase and unit costs required to complete the LCA are shown in Tables 4 and 5 (i.e., equipment size, financial data, factors and chemical use), whose assumptions and calculation procedure has been specified through Sections 2.3.1 and 2.3.3.

2.3.3. Economic viability analysis

The capital expenditure, operational expenditure and other economic parameters estimated according to the procedure described on Section 2.3.1 were used to develop an income statement showing the future annual expenditures of the facility. Following the Net Present Value or NPV method, monetary flows have been discounted with a 7.61 % weighted average cost of capital to the present; classified by typology into initial investment, variable operational expenditure, fixed operational expenditure, working capital and replacement of equipment; and aggregated to determine the total investment (both as a lump sum and annualized).

The weight average cost of capital was estimated considering the financing of the facilities within each scenario with own resources (66 %) and external debt (34 %), which is the most conventional financing model of Italian companies (Pwc, 2016). In the latter case, the average European tax rate of 14.2 % has been used to adjust the capital cost of the debt (EUTAX Observatory, 2021).

Other profitability tools such as return of investment, internal rate of return, minimum acceptable rate of return or payback period were not addressed since the main function of the facility is the removal of dye and economic benefits from product recovery from biogas or reclaimed water were not considered. Moreover, the tools mentioned serve different purposes and, in this case, the research is focused on the determination of the value of the investment. However, when studying NPV, some weaknesses must be taken into account. For example, NPV

Table 4
Unitary prices selected for the chemicals/energy included as variable costs.

Parameter process	Units	Value	References
Electricity	€/kWh	0.26	Eurostat (2023a)
Heat	€/kWh	$7.8{\cdot}10^{-2}$	Eurostat (2023b)
Sodium acetate	€/kg	0.9	Echemi.com (2023a)
Ammonium sulphate	€/kg	0.42	Chemanalyst (2020)
Pentahydrate magnesium sulphate	€/kg	0.34	Intratec (2023)
Calcium Chloride	€/kg	0.29	Intratec (2023)
Dipotassium phosphate	€/kg	1.13	Echemi.com (2023a)
Potassium dihydrogen phosphate	€/kg	1.35	made-in-China (2023)
Hexahydrate iron chloride	€/kg	0.45	Lama et al. (2016)
Ethylenedinitrilotetraacetic acid disodium salt	€/kg	3.02	Chemanalyst (2020)
Water	€/m^3	1.46	EurEau (2021)
Polymer	€/kg	2.40	Alibaba.com (2023)
Sodium hydroxide	€/kg	0.51	Echemi.com (2023b)
Hydrochloric acid	€/kg	0.10	Chemanalyst (2020)

Table 5
Major costs analysis parameters needed for the construction of the LCC.

Parameter process	Units	Value	References
Stirrer size of TK-101 and TK-102	kW	4.6875	–
Stirrer size of R-201, R-202, R-203 and R-204	kW	6.981	–
Blower size of scenario SLS	m^3/s	$2.63{\cdot}10^{-5}$	–
Blower size of scenario SHS	m^3/s	$4.10{\cdot}10^{-5}$	–
Blower size of scenario TPPB	m^3/s	$2.04{\cdot}10^{-4}$	–
Pump size of P-101 A/B	kW	0.09	–
Pump size of P-102 A/B and P-103 A/B	kW	0.40	–
Pump size of P-201 A/B and P-202 A/B	kW	0.40	–
Chemical Engineering Cost Index	–	800.6	Maxwell (2020)
Location factor	–	0.79	Green and Perry (2008)
Temperature factor	–	1.00	Smith (2016)
Pressure factor	–	1.00	Smith (2016)
Wroth factor for stirrers and reactors	–	4.1	Green and Perry (2008)
Wroth factor for blowers		2.5	Green and Perry (2008)
Worth factor for storage tanks		3.5	Green and Perry (2008)
Worth factor for pumps		7.0	Green and Perry (2008)
Material factor (for SS type 316)	–	2.6 from CS	Woods (2007)
WACC	%	7.61	Altavilla et al. (2021) and KPMG (2019)
Taxation rate	%	14.16	EUTAX Observatory (2021)
Amortization rate coefficient	%	15.00	EY (2014)
Cost price index	%	2.07	European Central Bank (2023a)
Land rent	%	1 % of investment	Sinnot and Towler (2020)
Maintenance of equipment	%	3 % of investment	Sinnot and Towler (2020)
Insurances	%	1 % of investment	Sinnot and Towler (2020)
Labour	€/person·year	37,135	Economic Research Institute (2023)
Direct salary overhead	%	30 % of labour and supervision	Green and Perry (2008)
Supervision	%	15 % of labour	Green and Perry (2008)
General plant overhead	%	65 % of labour, direct salary overhead and supervision	Sinnot and Towler (2020)
Lifetime of the facility	years	30	Rufí-Salís et al. (2022)
Useful life of blower, pumps and stirrers	years	10	Environmental Protection Agency (2022) and Zhou et al. (2021)

treats any investment as if it were a single opportunity under certain assumptions (Andrée et al., 2017).

Other disadvantages may be the failure to capture the exponential growth of the economy and the updating of legislation, which drives continuous technological substitutions within the sector. Due to this, there is a great deal of uncertainty in terms of determining cash flows. Despite the static nature of the LCA (as the selected approach was attributional) and the LCC, a sensitivity analysis was performed to capture how NPV changes under other market conditions. The following modifications were introduced for some hotspot parameters: a 25 % change in fixed costs, sodium acetate consumption and initial investment. The Weighted Average Cost of Capital was also reduced from 7.61 % to 5 %. Finally, a paradigm shift in accounting for price changes was included in the study. During the development of the LCC, inflation was addressed with the use of a constant average European Consumer Price Index. However, this index is a lagging indicator of inflation. The Producer Price Index is a more volatile and sometimes less favorable factor for the producer. Therefore, NPV results were also given for the average (1.84 %), the most unfavorable (14.83 % in 2021) values of this index and the known results for the year 2022 (7.84 %) (European Central Bank, 2023b).

3. Results and discussion

3.1. Impact environmental profiles

As mentioned on Section 2.1, the four scenarios (AnS, SLS, SHS and TPPB) were compared pairwise. AnS was taken as the reference case study within this section and, therefore, subsequent changes in the environmental profile of the other scenarios were described using this as a reference. Accordingly, Fig. 3 shows the relative contribution of each of the main parameters affecting the environmental performance of the anaerobic treatment process. Fig. 3 is a 100 % staked semi-radial bar chart and, despite its circular shape, can be interpreted similarly to a conventional stacked bar chart. In this figure, each bar in the semicircle corresponds to an impact category, which is further subdivided into different levels representing the inputs and outputs of the LCA inventory.

As shown in Fig. 3, energy consumption is the main environmental factor in three (IR, FE and FRS) of the seven midpoint categories analyzed, with a representativeness of over 64.9 % in the case of FE. The main effect of this energy consumption is in the IR category, with 83.1 % of total direct and indirect emissions. The reason for this hotspot is the use of stirrers in the reactors R-201, R-202, R-203 and R-204, which operate continuously with a power of 6.98 kW during the 1360 min of the reaction phase. Therefore, the use of electricity in the stirrers is indirectly exposing humans to radioactive emissions from the use of nuclear energy in the European electricity mix or from resource extraction activities from non-nuclear energy technologies (United Nations, 2021).

Reactor heating to reach the desired operating temperature (27 °C) competes with the stirrer as major concern in two (29.3 % for GWP and 54.2 % for FRS) of the seven categories highlighted. The share of other electrical devices is negligible, accounting for <1 % of the profile. As for chemicals, FET (52.5 %), MET (52.8 %) and HCT (53.6 %) are the most affected categories, with sodium acetate and ammonium sulphate being the components of most concern. However, the unitary environmental impact of each of them as it is inventoried in the Ecoinvent database is not the greatest when compared to the other compounds. This means that per kg of chemical product the most relevant species are the sodium salt of ethylenediaminetetraacetic acid and potassium phosphate. Finally, direct emissions to the environment are detrimental to the GWP, MET and FET categories. The first is strongly influenced by methane emissions with a share of 44.2 %, while the other two are affected by the non-degraded dyes in the effluent. Despite not making a large contribution to the profile (3.0 % and 0.2 % respectively), these emissions are significant for scenario comparison. This is because their similar physical configuration results in profiles with identical impact on most electrical devices except the blower.

Fig. 4 shows in pairs the comparative profile of the scenarios analyzed. From an environmental point of view, anaerobic treatment is analogous to sequential anaerobic-aerobic operation, except for two

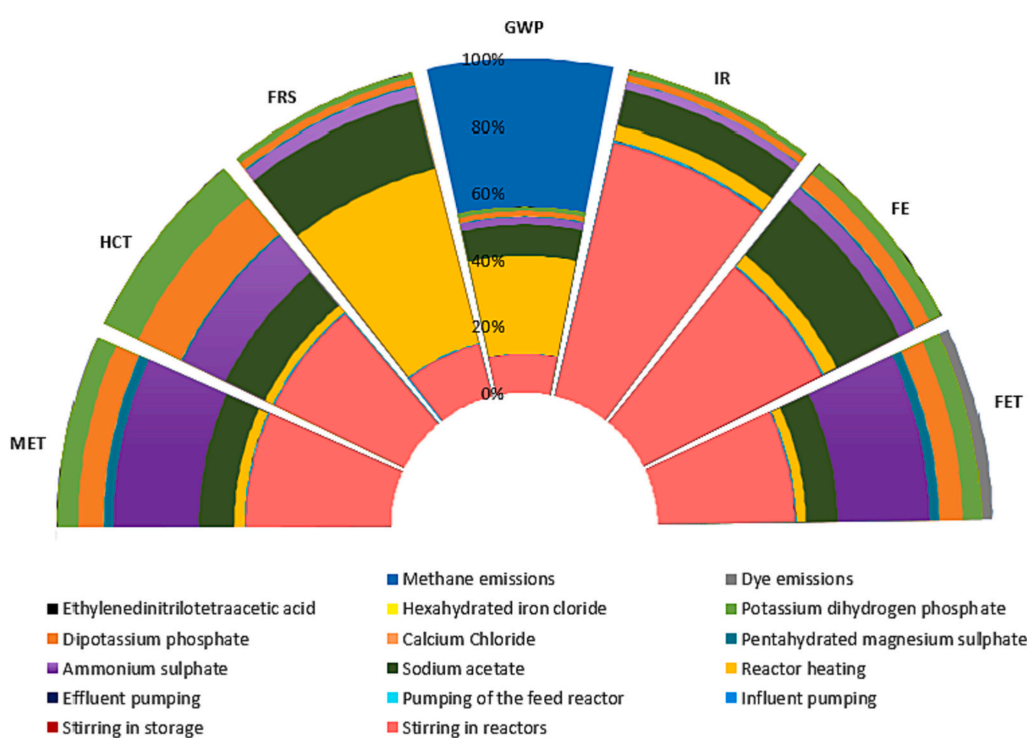

Fig. 3. Midpoint environmental impact relative contribution of the AnS (Anaerobic SBR reactor) scenario. Global Warming Potential (GWP), Fossil resource scarcity (FRS) Freshwater eutrophication (FE), Freshwater toxicity (FET), Human carcinogenic toxicity (HCT), Ionizing radiation (IR) and Marine toxicity (MET).

Fig. 4. Midpoint comparative profile of the scenarios (a) AnS and SLS, (b) SLS and SHS and (c) SHS and TPPB. Anaerobic SBR reactor (AnS), Sequencing anaerobic-aerobic operating with High dye load in an SBR reactor (SHS), Sequencing anaerobic-aerobic operating with high dye Load in an SBR reactor (SLS) and Two-Phase Partitioning Bioreactor (TPPB). Global Warming Potential (GWP), Fossil resource scarcity (FRS) Freshwater eutrophication (FE), Freshwater toxicity (FET), Human carcinogenic toxicity (HCT), Ionizing radiation (IR) and Marine toxicity (MET).

midpoint categories (FET and MET). Under the same organic loading rate and process capacity, the differences between the AnS and SLS scenarios are only the electricity consumed by the blower in charge of the oxic atmosphere keeping during the aerobic phase and the concentration of dye in the effluent (see Table 2). However, the blower is not one of the hot spots in the system since both scenarios differ minimally in their COD removal efficiency, from 95.7 % to 96 %. Therefore, the LCA results are transformed in the GWP by an absolute order of magnitude of 10^{-6}. Similarly, the extension of these results for other categories like IR, FE, and HCT reached a maximum of 10^{-5}. In contrast to the total COD removal efficiency, wastewater color removal caused by the dye mineralization is much higher (from 65.5 % in AnS to 90.6 % in SLS).

Therefore, the maximum environmental enhancement (2.3 %) in SLS can be detected in freshwater ecotoxicity. However, this improvement is directly related to the impact of a specific dye species. While methanil yellow or dye assumed as reference case study has a representativeness of 0.77 % in FET, the environmental contribution of acid violet provides a share of 0.05 %, methylene blue 0.56 % and orange II 0.05 %. In this regard, the better performance of the AnS and SLS could be related to the level of degradability and to the type of dye found in textile industry influents. Therefore, the relationship is not only associated with the color removal but also with the toxic implications of the remaining dye in the aquatic environment. It is, however, worth mentioning that the change type of dye only leads to a maximum impact difference between categories and dyes of 2.25 %. For such reasoning, the selection of the dye is not a main hotspot for the system, but other aspects related to chemical and energy consumption (as shown in Fig. 3).

As the organic loading rate increases, the following aspects undergo changes: the concentration of the chemicals added, blower consumption, and air and water emissions. All of these variables result in a change in environmental performance ranging from 18.5 % (for FRS) to 45.0 % (for FET). Since most of the equipment depends on the wastewater flow being processed, the energy demand does not change between SLS and SHS. Nevertheless, the relative contribution of chemicals is increasingly relevant compared to the reference scenario (AnS).

Regarding the above impact categories, the impact of chemicals rises from 52.5 % to 72.4 % in FET, from 52.8 % to 73.0 % in MET and from 53.6 % to 73.0 % in HCT. In this context, the relationship of the environmental profile with the demand of chemicals becomes as much as stronger as the organic load does. Because of this, the control of the dosing of chemicals becomes more stringent for a highly concentrated operation in SBRs. On the other hand, and from a technical point of view, a higher organic loading rates implies a lower dye removal rate. The estimations indicate a reduction from 90.5 % in the SLS scenario to 71.5 % in the SHS scenario. Therefore, the relative impact of this scenario is the highest predicted so far in terms of dye toxicity: 2.3 % for AnS, 0.77 % for SLS and 2.85 % for SHS.

To improve this situation, a polymer was fed into the reactors of the SBR treatment system (TPPB scenario). The goal was to avoid the inhibitory effects associated with the higher concentration of toxic compounds and, consequently, to improve the dye removal efficiency up to a 13 %. Three main differences can be found between SHS and TPPB associated with the environmental impact results: the use of pH controlling agents such as sodium hydroxide and hydrochloric acid, a higher electricity demand in the blower (due to the consequent improvement in COD removal) and the reduction of dye emissions. Although TPPB has a slightly worse environmental profile in five of the seven categories analyzed, the differences are not significant (<1 %). However, the positive effect of the reduction of the remaining dye in the effluent is much more relevant (0.12 % for MET and 1.79 % for FET compared to SHS). Taking a second look at Fig. 4A and B it can be seen that at the same organic loading rate there are no appreciable differences in impacts between the anaerobic, sequential and two-phase partitioning reactors except for two of the categories under study (MET and FET). These two categories have a strong relationship with the removal efficiency of the treatment since marine and freshwater ecotoxicity are strongly affected by the discharge of dyes into water courses.

It appears that an improvement in the removal efficiency of total COD and color brings a slightly better environmental profile for TPPB when it comes to the treatment of more concentrated wastewater in terms of CO_2 emissions as biogenic and methane released directly to the atmosphere. However, the implementation of a biogas blower and a heat and power unit for the production of electricity and heat from the generated methane can decrease the impact of the reference scenario in the range of 7.9 %–17.5 % (FRS and IR, respectively). In this regard, future research can be conducted to determine the technical feasibility of biogas recovery. Although differences may be found between fully anaerobic and sequential anaerobic-aerobic systems depending on the composition and degree of biogas recovery, CO_2 and CH_4 emissions to the environment would depend on COD and total color removal. Therefore, a sensitivity analysis is provided considering this type of emissions as non-biogenic rather than biogenic. In the AnS scenario, CO_2 non-biogenic emissions would denote 2.24 % of the new profile and this share would differ in the other scenarios by <1 %. Independently of being or not biogenic, the CO_2 emissions from the AnS scenario are not significant. The key factor is the maximum methane recovery. This is because the methane emission factor is 36 times higher than CO_2 and, thus, becomes the main hotspot in the CC category (Balcombe et al., 2018).

3.2. *Damage environmental profile*

While Section 3.1 showed the potential environmental impacts (midpoint) of each of the scenarios, this section highlights the contribution of a damage assessment process (endpoint). The results are expressed as a single score in Pt or points (reference unit for environmental comparison of processes/products) rather than a multitude of environmental categories. Apart from a single score, the endpoint analysis draws attention to the end stage of the cause-effect chain. It highlights the consequences of actions on humans, ecosystems and resources rather than focusing on the relative importance of an emission. The outcomes reached by the four scenarios are similar with the scores of 28.27 Pt, 28.29 Pt, 35.38 Pt, 35.28 Pt for AnS, SLS, SHS and TPPB respectively. Because of this similarity, Fig. 5 only shows the disaggregated results for the TPPB scenario, which is the new technology highlighted in this publication. The most impactful aspect of the profile for the TPPB is the methane emissions produced during the anaerobic reaction phase of the treatment process with around 41 % of the overall score. Heating follows closely with a representativeness of 25 % and the electricity demand of the stirring has a share of about 12 %. While in the midpoint analysis stirring had a more relevant role in the profile than heating, the aggregated endpoint analysis shows a very different trend. In this particular case, the damage generated by heating doubles that of stirring activities. This is because the global warming endpoint damage category is the largest contributor in two (human health and ecosystem quality) of the top three endpoint aggregation levels: human health, ecosystems and resources. According to the midpoint GWP category (as shown in Fig. 3) the hotspots are arranged in the following order: methane emissions, heating and electricity from the stirring of the reactors. This sequence matches the already displayed for the endpoint analysis. Compared to the results of Fig. 5, two items energy items vary

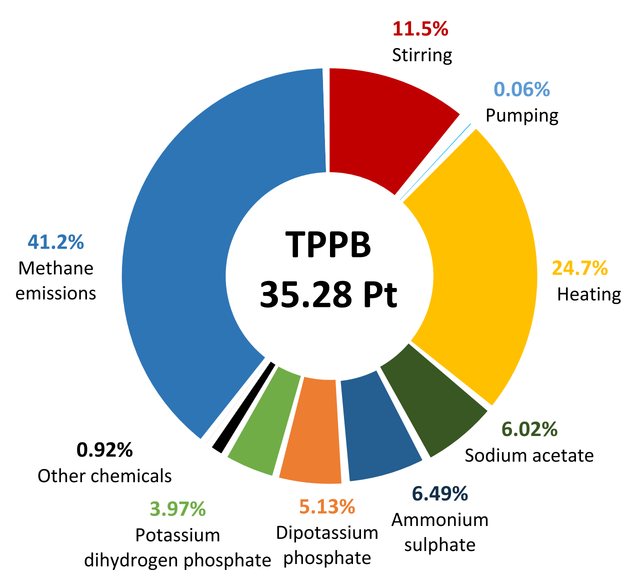

Fig. 5. Endpoint relative profile for the Two-Phase Partitioning or TPPB scenario.

noticeably between scenarios which are the stirring from 14.3 % (AnS) to 11.5 % (TPPB) and the heating from 30.8 % (AnS) to 24.7 % (TPPB).

Regarding to the influence of chemicals, the conclusions reached are similar to those already specified in the midpoint analysis. The main deviation in the profile from SLS and SHS can be found in the increase of their contribution. For example, the sodium acetate percentages varied from 5.26 % to 6.0 %. On the other hand, and considering a more global approach, the increase of the organic loading rate is the responsible of the worst single score outcomes in scenarios SHS and TPPB. However, the damage analysis is strongly defined by global warming potential and not by ecotoxicity category. Because of this, a clear win-win solution cannot be distinguished.

3.3. Economic profile

The TPPB process results in a 46.3 M€ total investment through the lifespan of the designed facility. The cost breakdown for the investment differentiated the following categories: initial investment, initial working capital, variable costs, fixed costs, variables of working capital and acquisition of assets in time. All of them can be differentiated in Fig. 6. Although outcomes have been displayed hereunder for TPPB, the proportions of cost items are similar also for AnS, SLS and SHS since the deviation of the respective shares is no larger than 2 %. The global investment profile is dominated then in TPPB by the fixed costs, which make up 69.5 % followed by the initial capital investment in a 21.5 % (or 9.9 M€). As hotspot, the fixed costs are proven to be very sensitive to changes. This suggests that an overall reduction of 25 % in the two most important economic parameters (operating and maintenance personnel and plant overhead, at 30.3 % and 29.5 %, respectively) associated with them is accompanied by an overall investment decrease for the construction of the treatment facility of about 17 %. Because of this, the operational expenditure is representing 73.6 %–75.8 % of the total costs. Apart from fixed costs, another critical parameter is the weight average capital cost when used as discount rate. This is because, the rate used to discount the future cash flows from an investment to the present value from 7.61 % to 5 % implies a 22.2 % higher total investment.

Regarding the results of the initial capital investment, the main cost driver is the construction and installation of the reactors with around 1.9 M€ each. The second largest cost generating component is the mixing tanks, with ~11.6 % of the investment. However, the initial investment does not encounter improvements as effective as those of the fixed costs when the same reduction is applied in the sensitivity analysis (6.5 %). The relevance of variable costs shows a similar trend to the results of the LCA analysis. Reactor heating and the use of reactor stirrers are the two main components with percentages of 34 % and 32 % respectively. The use of sodium acetate accounts for 15.2 % of the variable costs, followed closely by external environmental costs associated with the operation of the facility (11.1 %). Although these have been included in the study to highlight their importance should the company have to pay for them, toxicity effects are not reflected and therefore scenarios such as TPPB using polymer to increase disposal efficiency are being penalized. As far as the change of the Producer Price Indexes is concerned, the total investment varied from the 48.1 M€ to 47.2 M€ (at 1.84 %), 244.2 M€ (at 14.83 %) and 87.8 M€ (at 7.84 % in 2022).

Four aspects determine the better economic performance of TPPB: the relative difference between the dye removal efficiency between the two scenarios, the market costs of the polymer, the penalties imposed by the government in relation to environmental protection (a dissimilarity of 0.8 % in environmental costs can be found when the environmental weighting factor for toxicity reaches a price of 10 €/kg of 1,4-DCB) and the durability of the polymer (the replacement of the spoiled polymer by a fresh feed is done only once per year of operation of the facility). Regarding to the second above-named aspect, the investment costs of the scenarios SHS and TPPB reduced from 8.3 % to 0.6 % by changing the polymer prices from 2.4 €/kg to 0.1 €/kg. The relationship between the modifications in the investment costs and the prices were linear with a slope of the equation of 0.035.

The total annual equivalent cost of the AnS scenario is 3.6 M€, which expressed per functional unit translates into 10,985 €/d and per cubic meter of treated water into 43.94 €/m^3. The cost of SLS were similar to the AnS scenario with a negligible difference between them, which is related to the purchase of the blowers needed to operate under oxic conditions. As expected, and in agreement with the LCA outcomes, the increase of the organic loading rate from scenario SLS to SHS also raised the prices (~1 %) because of the larger use of chemicals and environmental impact costs. The TPPB had a 48.1 €/m^3 total specific investment in agreement to the above-mentioned key points. It reflects how a moderately higher investment could ensure a larger level of protection to the environment.

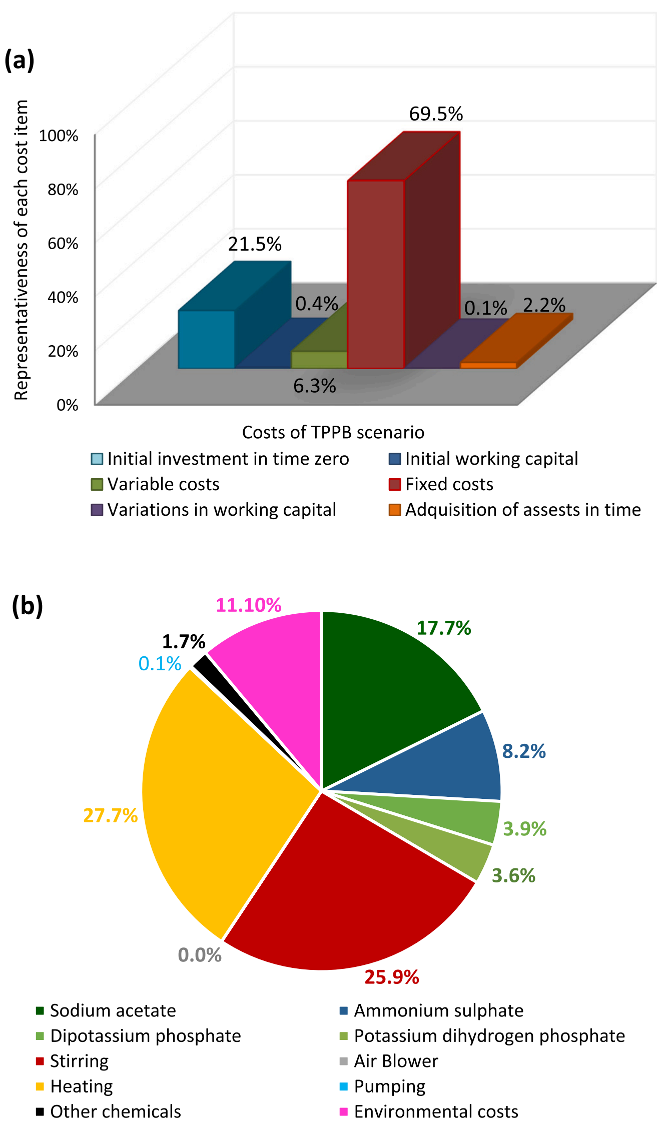

Fig. 6. Economic outcomes of the TPPB scenario. (a) Breakdown per updated cost item; (b) Share of the variable costs of operation.

4. Conclusions

The selection of the anaerobic or sequential anaerobic-aerobic media conditions, the organic loading rate, the type of dye eliminated, and the use of amorphous polymers has demonstrated to influence the environmental and economic profiles while running textile wastewater treatment processes using SBRs and TPPBs. The first three technical parameters have led to a significant change in the environmental outcomes of the four analyzed scenarios. The underlying rationale is a considerable influence of marine and terrestrial ecotoxicities (a maximum of 2.3 % was recorded) caused by an increased removal efficiency of the systems, while the differences in the other impact categories more related to chemical and energy consumption are almost negligible (well below 1 %).

Comparison with TPPB is only possible for high organic loading rates since its design was intended to reduce the biomass inhibition caused by dyes and thus overcome the lower removal efficiency found for anaerobic-aerobic treatment for high organic loading rates. In this circumstance, an increase in this parameter led to a considerable increase in the impacts on the environmental profile created mainly by the increase in dye emissions to water as a result of the decrease in removal efficiency and the increased use of chemicals, to maintain the optimal level of carbonaceous substrate and microelements necessary for biomass growth (20 % increase in the profile). Overall, TPPB performed worse than anaerobic-aerobic treatment in all impact categories except marine and terrestrial ecotoxicity. However, except in these two cases, the difference between the impacts of the categories cannot be considered statistically significant, as it is well below 1 %. It is important, however, to note that an improvement in removal efficiency led to a much better performance in terms of toxicity.

Although from an environmental point of view, the sequential anaerobic-aerobic process resulted in better outcomes than the single step anaerobic treatment and the TPPB outperformed both, the economic results restrict the feasibility of this scenario to the market prices of the polymer, its durability and the assigned value of the environmental costs of ecotoxicity. Polymer consumption is expected to increase the cost investment by 0.6–8.3 % of TPPB compared to anaerobic-aerobic treatment for a price variability per weight of the polymer ranging from 0.1 to 2.4 €/kg. In contrast to the initial hypothesis proposed, energy and chemical consumption did not prove to be limiting factors for economic feasibility.

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CRediT authorship contribution statement

Sofía Estévez: Conceptualization, Formal analysis, Investigation, Methodology, Visualization, Writing – original draft. **Domenica Mosca Angelucci:** Conceptualization, Investigation, Supervision, Writing – review & editing. **María Teresa Moreira:** Supervision, Visualization, Writing – review & editing. **M. Concetta Tomei:** Conceptualization, Supervision, Validation, Writing – review & editing.

Declaration of competing interest

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

Data availability

Data will be made available on request.

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