Use of Reclaimed Municipal Wastewater in Agriculture: Comparison of Present Practice versus an Emerging Paradigm of Anaerobic Membrane Bioreactor Treatment coupled with Hydroponic Controlled Environment Agriculture

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Abstract

Advancements in anaerobic membrane bioreactor (AnMBR) technology have opened up exciting possibilities for sustaining precise water quality control in wastewater treatment and reuse. This approach not only presents an opportunity for energy generation and recovery but also produces an effluent that can serve as a valuable nutrient source for crop cultivation in hydroponic controlled environment agriculture (CEA). In this perspective article, we undertake a comparative analysis of two approaches to municipal wastewater utilization in agriculture. The conventional method, rooted in established practices of conventional activated sludge (CAS) wastewater treatment for soil/land-based agriculture, is contrasted with a new paradigm that integrates AnMBR technology with hydroponic (soilless) CEA. This work encompasses various facets, including wastewater treatment efficiency, effluent quality, resource recovery, and sustainability metrics. By juxtaposing the established methodologies with this emerging synergistic model, this work aims to shed light on the transformative potential of the integration of AnMBR and hydroponic-CEA for enhanced agricultural sustainability and resource utilization.

Keywords: Conventional activated sludge, Land/soil-based agriculture, Anaerobic membrane-based wastewater treatment, Hydroponics, Circular agriculture

1. Introduction

Freshwater scarcity can impact food security, nutrition, and livelihoods in agriculture as well as affecting various socioeconomic factors such as income levels, employment opportunities, land values, food prices, and investment in agricultural activities. A dependable alternative non-conventional water source can provide a solution to mitigate these pressures and uncertainties. Given the backdrop of population growth, increasing food requirements, and the strain on water resources, enhancing water productivity in agriculture becomes paramount. On a global scale, about 70% of freshwater withdrawals are associated with agricultural practices (FAO, 2017). Similarly, agriculture presently claims a significant market share of approximately 30% of reclaimed water applications (Connor, 2017). Implementing wastewater reclamation and water reuse has already played a pivotal role in augmenting agricultural output, even in the face of limited access to freshwater resources (Lazarova, 2013).

Water reuse in agriculture involves treating municipal wastewater and using the reclaimed water for diverse agricultural objectives. This approach emerges as a strategy to tackle water scarcity by optimizing water resources and curtailing the discharge of untreated wastewater (especially in underdeveloped and developing countries) and conventionally-treated wastewater into natural ecosystems. Nevertheless, considering factors such as pathogenic microorganisms and emerging organic contaminants, the human health risk associated with treated wastewater remains a significant concern (Al-Hazmi et al., 2023; Dickin et al., 2016).

The role of wastewater reuse in agricultural has been a persistent subject of scrutiny, with several evaluations of its economic, environmental, and human health implications reported in literature (Al-Hazmi et al., 2023; Mainardis et al., 2022; Ofori et al., 2021; Singh, 2021). Despite these considerations, there is a noticeable scarcity of studies addressing the investigation and formulation of policies and regulations that facilitate the safe and effective utilization of treated wastewater in agriculture. This research gap is crucial, as developing such policies is imperative for ensuring the responsible and sustainable implementation of wastewater reuse practices. In this perspective, we evaluate the existing water quality regulations and guidelines governing wastewater reuse in agriculture. We focus on identifying gaps in water reuse practice, particularly in hydroponic controlled environment agriculture (CEA) systems, which differ from traditional land/soil-based agriculture. We also compare the feasibility of integrating anaerobic membrane bioreactor (AnMBR) technology with hydroponic CEA versus conventional activated sludge (CAS) systems used for treating water in land/soil-based crop cultivation. This study aims to recognize and address gaps and challenges associated with providing nutrients from decentralized AnMBR effluents to facilitate reuse in suitable, low land-footprint hydroponic CEA systems. Crucially, addressing regulatory gaps and assessing the treated effluent as a nutrient resource are essential for promoting sustainable agricultural water reuse in hydroponic CEA systems.

2. **Current practice**

2.1. **Conventional wastewater treatment for land/soil agriculture**

2.1.1. ***Irrigated agriculture***

Irrigation in soil-based agriculture involves the controlled application of water to agricultural fields to support plant growth. Irrigation ensures consistent crop yields, especially in regions with irregular rainfall patterns or insufficient soil moisture. Common water sources for irrigation include surface waters and groundwaters, and increasing use of treated wastewater.

Integrating water reuse into irrigated agriculture emerges as a sustainable avenue. Certain wastewater constituents, such as nitrogen and phosphorus, often deemed pollutants in treated effluents, are important plant nutrients.

However, long-term land application of wastewater effluents diminishes productivity due to altered soil physical, chemical, and biological properties, invasive weeds, and soilborne pests and diseases (Durán–Álvarez and Jiménez–Cisneros, 2014). The situation is exacerbated by the suboptimal use of resources, including excessive water, fertilizer, herbicide, and pesticide usage, ultimately leading to detrimental environmental impacts (Aktar et al., 2009; Ayoub, 1999).

2.1.1.1. Agricultural water quality criteria and regulations for reclaimed wastewater

Many regions have regulations and guidelines for irrigation to protect water resources and the environment. Compliance with these regulations is essential to minimizing impacts on public health and preventing water pollution and ecosystem damage. Similarly, different regulations apply depending on the category of reuse. For example, the acceptable levels of fecal coliform are stricter for agricultural reuse involving food crops than for processed and non-food crops (Table 1).

Table 1. USEPA suggested regulatory guidelines for the reuse of wastewater in irrigation (USEPA, 2012).

Reuse Category	Treatment	Water Quality	Monitoring
Agriculture food crops	Secondary + Filtration + Disinfection	pH 6.0–9.0; BOD_5 30 mg/L; TSS 30 mg/L; Fecal coliform 0/100 mL; Cl_2 residual 1.0 mg/L (min)	pH (weekly), BOD_5 (weekly), Turbidity (continuous), Fecal coliform (daily), Cl_2 residual (continuous)
Agriculture processed food crops and non-food crops	Secondary + Disinfection	pH 6.0–9.0; BOD_5 30 mg/L; TSS 30 mg/L; Fecal coliform 200/100 mL; Cl_2 residual 1.0 mg/L (min)	pH (weekly), BOD_5 (weekly), TSS (daily), Fecal coliform (daily), Cl_2 residual (continuous)

Opting for food crops that are less prone to direct contact with treated wastewater or adequately cooked before consumption is often preferable. When proper treatment and safety measures are in place, various food crops can be cultivated using reclaimed water. Examples include leafy greens (crops like lettuce, spinach, kale, and Swiss chard), root vegetables (root crops such as carrots, beets, radishes, and turnips), grains for animal feed (such as corn, sorghum), fruit trees (such as citrus, apples, and pears), nuts (like almonds and walnuts), herbs (such as basil, oregano, and mint), certain vegetables (like tomatoes, cucumbers, and zucchini), and ornamental crops (such as flowers and ornamental plants).

2.1.1.2. Challenges of wastewater reuse in agriculture

Wastewater reuse in agriculture presents both advantages and drawbacks. On the positive side, it contributes to a reduction in the consumption of freshwater for agricultural purposes and may serve as a nutrient supplement. However, various challenges accompany this practice.

Soil quality

Irrigation using reclaimed water can influence the physical properties of soil, including pH, EC, cation exchange capacity, as well as its chemical, and biological aspects, such as enzymes and native organisms (Adrover et al., 2012; Habibi, 2019). For instance, the presence of cations like sodium can potentially disrupt soil structure by affecting soil colloids. This problem is particularly pronounced in coastal regions, where the accumulation of sodium and chloride (Na^+ and Cl^-) is heightened due to the influence of seawater. The subsequent dispersion of soil colloids further compounds the problem, resulting in the loss of soil structure and permeability. Similarly, prolonged irrigation with reclaimed water frequently gives rise to challenges associated with the gradual buildup of contaminants in the soil. This accumulation poses a threat to the long-term quality of the soil environment and can lead to elevated levels of pollutants in both plants and shallow groundwater (Bao et al., 2014; Christou et al., 2014).

Food safety concern

Food crops, especially fresh produce, cultivated using treated wastewater, raise safety concerns due to the potential bioaccumulation of contaminants and/or the presence of pathogenic microorganisms (Kesari et al., 2021; Shrivastava et al., 2022). Safety concerns stand as paramount considerations in the realm of wastewater reuse for agricultural purposes.

Organic micropollutant bioaccumulation

Another challenge associated with required wastewater treatment technologies for agricultural reuse lies in the fate and persistence of emerging organic contaminants (Bolong et al., 2009; Lim et al., 2020). Emerging wastewater-derived organic contaminants such as endocrine-disrupting compounds, pharmaceuticals, personal care products, and antibiotics have not been adequately regulated, monitored, or incorporated into wastewater quality standards. Thus, there are no incentives for wastewater treatment facilities to address them proactively. An illustrative example is the presence of antibiotics in wastewater, elevating the risk of abiotic resistance bacteria (ARB) and antibiotic resistance genes (ARG) (Gupta et al., 2018). These genes could undergo horizontal gene transfer to different bacteria (Gupta et al., 2018). Inadequate removal of other emerging contaminants by current conventional wastewater treatment technologies is also observed (Michael et al., 2013), primarily because these technologies were not designed explicitly for this purpose. The chemical structures of emerging contaminants can exacerbate their resistance to biodegradation or removal during wastewater treatment, as some are intentionally designed to be stable and persistent (Lei et al., 2015). Additionally, these contaminants are often present in wastewater at extremely low concentrations, typically in the range of micrograms per liter or nanograms per liter (Kumar et al., 2022; Samal et al., 2022), a level of concentration for which secondary treatment processes are not optimized for removal.

Given the challenges in completely removing emerging contaminants, trace levels of these substances may persist in treated effluent. Subsequently, these substances can enter the environment or potentially bioaccumulate in crops (Pi et al., 2017). This process may have ramifications for aquatic ecosystems and human health. In some instances, these effluents are employed for agricultural purposes, raising the potential for contamination of crops and, by extension, human health.

2.1.2. ***Conventional wastewater treatment practice and regulatory requirements***

Conventional wastewater treatment methods are widely employed in centralized wastewater treatment plants. The treatment process typically comprises several unit operations: pretreatment, primary treatment, secondary biological treatment, and occasionally tertiary physical/chemical/biological treatment.

Biological wastewater treatment processes utilize microorganisms to remove organic constituents, particularly carbon, as well as nitrogen and phosphorus, from wastewater under various environmental conditions. These biological processes are generally categorized into aerobic and anaerobic treatment methods. Aerobic treatment methods, such as conventional activated sludge (CAS), percolating filters, rotating biological contactors (RBC), sequencing batch reactors (SBR), and membrane bioreactors (MBR), involve the use of oxygen to support microbial activity.

In contrast, anaerobic treatment methods, including anaerobic digestion, anaerobic completely stirred tank reactors, upflow anaerobic sludge blanket reactors, upflow anaerobic filters, anaerobic contact processes, and anaerobic membrane bioreactor (AnMBR), do not require oxygen for the breakdown of contaminants.

A primary goal of some wastewater treatment systems, beyond removing suspended solids and oxygen demand, is to effectively remove both nitrogen and phosphorus to prevent eutrophication. Biological nitrogen removal can be summarized as illustrated in Fig. 1. Nitriding converts organic nitrogen to amino-N and then to ammonia-N. Nitrosation converts ammonia-N into nitrite, whereas nitrification converts nitrite to nitrate under aerobic conditions, while denitrification transforms these oxidized nitrogen forms into nitrogen gas under anoxic conditions.

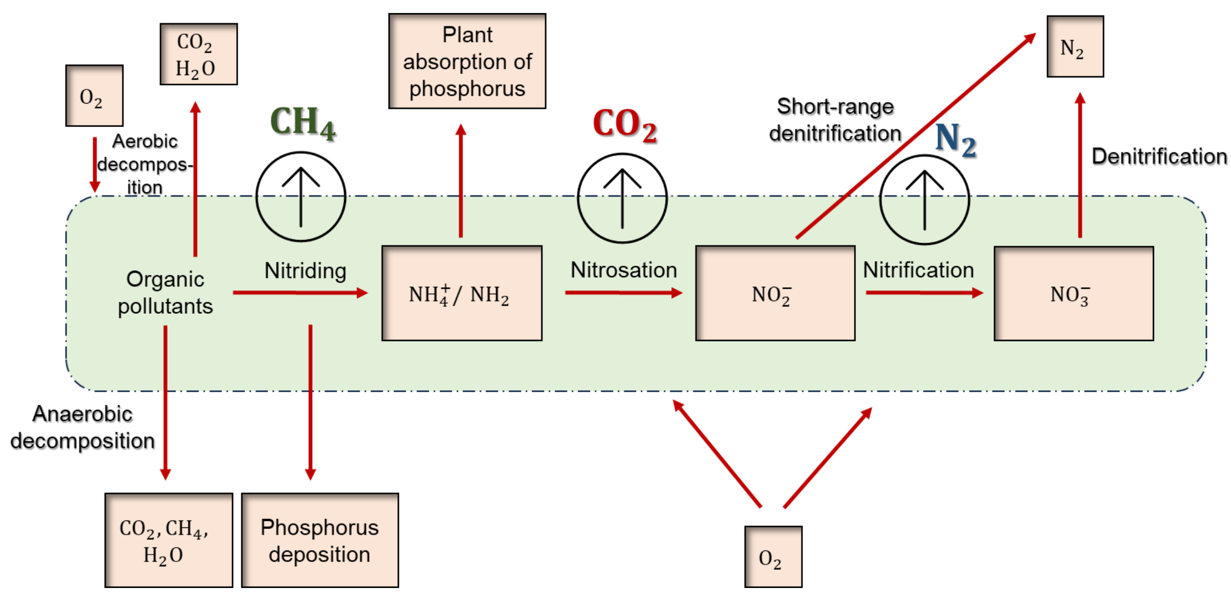


Fig. 1. Illustration depicting the sequential stages of biological denitrification in wastewater treatment (Adapted from (Zhou et al., 2023)).

Phosphorus in wastewater can be either soluble or particulate. Particulate phosphorus is removed through solids adsorption, while soluble phosphorus can be biologically removed by aerobic microorganisms, such as phosphorus-accumulating organisms (PAO). Phosphorus removal is enhanced in the presence of oxygen and PAO. Biological treatment processes liberates phosphate from organic phosphorus, which can then be removed via tertiary chemical precipitation.

In terms of effluent quality, aerobic systems like CAS exhibit higher nitrogen and phosphorus removal rates compared to anaerobic systems. This is primarily due to the higher cell yield coefficient, Y, in aerobic systems. Specifically, the yield of cells per gram of carbon is approximately five times greater in aerobic systems than in anaerobic ones. This difference has two consequences: firstly, aerobic systems produce around five times more sludge, and secondly, they result in approximately five times more nitrogen and phosphorus loss within the aerobic cells.

2.1.2.1.Conventional activated sludge (CAS) process

The CAS process is a widely embraced suspended growth aerobic biological wastewater treatment method. In this process, wastewater is mixed with active microorganisms in an oxygen-supplied reactor. The microorganisms break down bulk organic matter in the wastewater, converting it into biomass and byproducts. Aeration and sedimentation can occur either in a single tank or distinct tanks, with CAS operating in two modes: plug flow and complete mix.

In a plug flow reactor, influent wastewater enters at one end and exits at the other, ensuring uniform residence times for all wastewater constituents. Conversely, a complete mix reactor achieves homogeneity through agitation, resulting in constituents having varied residence times.

The CAS process accommodates diverse variations, with designs influenced by factors such as influent flow rate, organic loading rate, efficiency requirements, the need for secondary settling, and the types of aeration systems. Design criteria encompass kinetic parameters (e.g., cell yield coefficient and endogenous decay coefficient) and operating parameters (e.g., hydraulic residence time (HRT), cell and sludge residence time (SRT), food-to-microorganism (F/L) ratio, mixed liquor suspended solid (MLSS) concentration, mixed liquor volatile suspended solid (MLVSS) concentration, volumetric loading, recycling ratio, mixing power, and oxygen transfer rate). Typically, the floc in CAS, denser than water, facilitates effective settling, yielding nearly sludge-free effluent.

In CAS, three key parameters are pivotal: F/M ratio, SRT/HRT, and the return activated sludge (RAS) flow rate. The F/M ratio is a crucial factor in maintaining equilibrium within the system. It ensures a balance between the organic load, food for microorganisms, and the concentration of microorganisms in the tank. Typically, a desirable range for the F/M ratio in CAS is between 0.2 and 0.4 (Wu et al., 2013). Solids residence time (SRT) refers to the duration microorganisms spend in the secondary treatment train. This parameter is vital for optimizing the treatment process. Simultaneously, the RAS flow rate dictates the flow at which a portion of the settled activated sludge is reintroduced into the aeration tank, contributing to the continuous treatment cycle.

2.1.2.2. Chemical post-disinfection

An obligatory step in wastewater treatment, especially for effluents earmarked for re-use, is post-disinfection using chemical disinfectants, membrane technology, ultra-violet light irradiation, etc. Chemical disinfectants require sufficient contact time provided in contact tanks with baffling. Various chemical post-disinfectants like chlorine, ozone, and chlorine dioxide have been employed over the years. While chlorine has historically proven effective and cost-efficient, its drawbacks include the production of disinfection by-products (DBPs) that are harmful to both humans and the environment, along with storage and safe handling challenges.

2.1.2.3. Greenhouse gas (GHG) footprint

The GHG footprint of CAS processes is a significant consideration when assessing their environmental impact. Various factors influence it, including plant design, operational practices, and the regional energy source. CAS systems contribute to GHG emissions primarily through the energy-intensive processes involved in aeration and the lack of energy and nutrient recovery (McCarty et al., 2011). In conventional aeration tanks, where microorganisms are employed to break down organic pollutants, more than 40% of the total plant energy demand is required to provide the necessary oxygen for microbial activity (Mamais et al., 2015; McCarty et al., 2011). This is estimated to be around 0.298 kg of CO_2 eq/m^3 (derived from the reported U.S. energy carbon footprint of 0.472 kg of CO_2 per kWh) (Yan et al., 2014). The reliance on electricity, often sourced from conventional grids, can result in notable carbon footprints, particularly in regions where the energy mix includes a significant proportion of fossil fuels.

Methane and nitrous oxide, other potent GHGs, may also be produced in CAS systems under certain conditions. Anaerobic zones within the CAS treatment train or during sludge treatment processes such as collection systems, primary clarifiers, influent piping, and grit chambers can lead to methane generation (Monteith et al., 2005; Tumendelger et al., 2019). Denitrification of effluent nitrate can lead to nitrous oxide generation (Tumendelger et al., 2019). Although methane emissions are significantly lower in CAS systems compared to anaerobic systems, they still contribute to the overall GHG profile.

Other sidestream processes of CAS, such as dewatering, transportation, and landfilling also contribute to the GHG footprint. Efforts to mitigate the GHG footprint of CAS systems often focus on enhancing energy efficiency, optimizing aeration processes, and exploring renewable energy sources. Upgrading aeration technologies, adopting energy-efficient equipment, and utilizing cleaner energy sources can significantly reduce the carbon intensity of CAS operations.

3. **Emerging Paradigm**

3.1. **Anaerobic membrane reactor (AnMBR) technology**

There has been a notable shift in wastewater treatment practices in recent years, with increasing emphasis on aerobic MBRs. The aerobic MBR integrates aspects of CAS systems by employing aerobic microorganisms to degrade organic pollutants. However, unlike CAS systems where effluent is separated from sludge in settling tanks, aerobic MBRs utilize membranes for filtration. These filtration membranes enhance CAS efficiency by retaining solids and potentially facilitating compartmentalization for processes like nitrification, denitrification, and biological phosphorus removal.

Nevertheless, aerobic processes are associated with significant drawbacks, including high energy consumption, substantial sludge production, large footprint, and high operational and maintenance costs (Caniani et al., 2015). Thus, there is a growing shift towards anaerobic treatment methods driven by the recognized potential for energy savings. This shift aligns with broader sustainability and resource recovery goals, particularly within the framework of net-zero energy wastewater treatment.

Anaerobic biological processes, which takes place in the absence of oxygen, produce biogas as a byproduct, primarily composed of methane. This biogas can be harnessed as an energy source for electricity generation or heat production, both for internal use within the reactor or as an external energy supply. Utilizing biogas-driven energy can power the treatment facility itself, thereby reducing or eliminating the dependence on external energy sources. This approach not only cuts operational costs but also promotes a more sustainable and self-sufficient wastewater treatment.

There is a similar growing interest in integrating membrane processes with anaerobic wastewater treatment. This interest arises because conventional anaerobic processes typically demand longer HRT (due to combined HRT and SRT) to remove chemical oxygen demand (COD) effectively. AnMBR addresses this issue by decoupling HRT from the SRT. Compared to CAS systems, AnMBRs typically require less energy due to their operation under anaerobic conditions, eliminating the need for a continuous oxygen supply. In CAS system, aeration constitutes a significant portion of the total energy consumption (Au et al., 2013; Sean et al., 2020). The reduced energy requirement of AnMBRs results in lower GHG emissions, especially when fossil fuels are the energy source. Additionally, AnMBRs can generate methane, a potent GHG (if released into the environment). However, this methane can be captured and converted into a valuable energy source. By utilizing captured methane for heating and power generation, AnMBRs can offset emissions that would otherwise be released into the atmosphere if the methane were not captured. Moreover, the reduced sludge production (yield) associated with AnMBRs contributes to a decreased carbon footprint in terms of sludge treatment and disposal. Nevertheless, for a comprehensive and accurate comparison of the actual emissions of wastewater treatment systems, a detailed life-cycle analysis that considers all emissions and energy inputs is necessary. The utilization of biogas also offsets the need for fossil fuels, further reducing the overall carbon footprint of the treatment facility.

The AnMBR system can be configured in several ways, offering flexibility in its design. The membrane can be submerged within the reactor, attached as an external unit, or submerged in an external chamber separated from the reactor. The submerged membrane configuration is known for its relatively lower energy requirement, typically ranging from 0.038 to 5.68 kWh/m^3 (Maaz et al., 2019). In contrast, the energy demand for the external membrane system typically falls within

3 to 7.3 kWh/m^3 (Maaz et al., 2019). However, it's essential to delve deeper into the choice between submerged and external membrane configurations for energy saving. According to Martin et al. (Martin et al., 2011), this decision should be made thoughtfully, considering factors like water flux (Table 2) and the uncertainty surrounding the appropriate gas sparging rate required for sustainable operation of the external mode.

One of the significant aspects of the AnMBR systems is limited nutrient (nitrogen and phosphorus) removal (Fig. 2), which is detrimental for effluent discharge but advantageous for agricultural reuse. Compared to the conventional treatment system utilizing aerobic microbes, where nitrogen and phosphorus are significantly removed, the AnMBR process typically does not include aerobic or aeration zones, resulting in minimal removal of ammonia-nitrogen and phosphorus from the influent. This makes the AnMBR effluent a rich source of nutrients, particularly nitrogen and phosphorus. As a result, nutrient concentrations in anaerobic processes are generally higher than in processes that incorporate aerobic treatment.

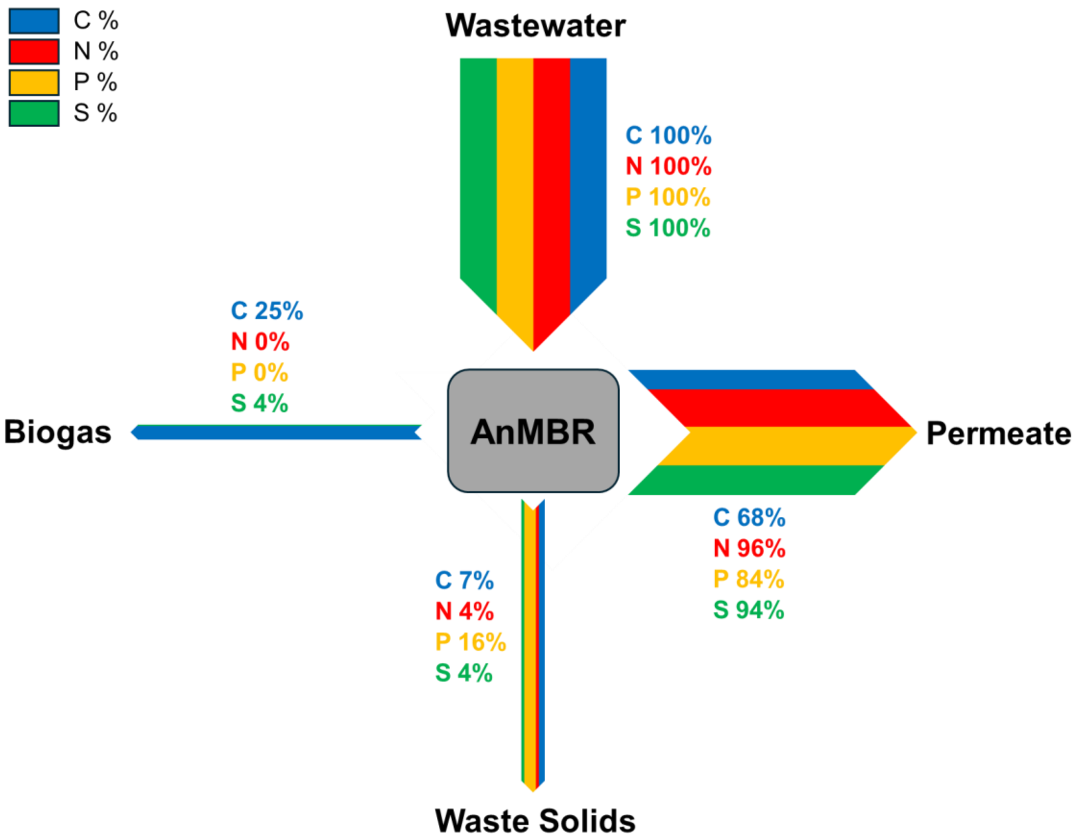


Fig. 2. Schematic representation of an AnMBR, highlighting nutrient mass balance (Du et al., 2022).

Another challenge of the AnMBR system is fouling. Managing fouling in such a setup can be difficult without disrupting the biological system and overall operation. The biogas produced in AnMBR, primarily composed of methane and carbon dioxide, can be sparged through the membranes to dislodge and prevent the buildup of foulants on the membrane surface. AnMBR demonstrates immense potential to offset the energy demand and potential fouling of the system through the production and utilization of biogas.

Overall, anaerobic processes align well with the principles of circular economy by promoting resource recovery. In addition to biogas production, an AnMBR produces a high-quality, nutrient-rich effluent for agricultural reise. Anaerobic digestion produces nutrient-rich biosolids that can serve as organic fertilizers in land-/soil-based agriculture. This closed-loop approach maximizes the value extracted from wastewater, transforming it into valuable products while minimizing waste generation.

3.1.1. ***AnMBR operational parameters***

Various operational and water quality parameters play a crucial role in determining the performance and efficiency of AnMBRs, such as HRT, SRT, organic loading rate (OLR), temperature, pH level, alkalinity and acidity, biomass concentration, nutrient levels, fouling control, permeate flux, mixing and hydrodynamics (Table 2).

One significant challenge with the submerged membrane configuration in AnMBR systems is fouling. Managing fouling in such a setup can be difficult without disrupting the biological system and overall operation. Fouling on AnMBR membranes could be caused by gel/cake layer formation, pore narrowing or pore blocking (Metcalf et al., 1991). Strategies for fouling control, such as biogas sparging, backwashing and chemical cleaning, are important parameters to maintain AnMBR performance.

3.1.2. ***AnMBR effluent water quality***

Several factors, including influent characteristics, operational parameters, membrane type, and system design influence the effluent quality of an AnMBR. Notably, treating low-strength wastewater such as domestic/municipal wastewater, especially at low temperatures, poses a particular challenge for AnMBRs (Kanafin et al., 2021; Song et al., 2018; Verstraete et al., 2009). This challenge arises from the insufficient organic content in low-strength wastewater, which may disrupt the microbial balance, leading to process instability and reduced treatment efficiency. To address this limitation when using AnMBRs for low-strength wastewater treatment, incorporating agricultural waste or food waste could bolster the overall organic load, thereby enhancing AnMBR performance. Additionally, harnessing methane generated within the AnMBR can heat the wastewater, improving its efficiency by providing mesophilic conditions (e.g., 35 ºC).

Nevertheless, effluent quality requirements may vary based on local regulations, intended reuse, or discharge standards. To ensure consistent effluent quality, it is crucial to engage in regular monitoring, adjust operational parameters as needed, and operationally maintain the system. An AnMBR can treat organic-rich wastewaters while simultaneously producing biogas primarily composed of CH_4 and CO_2. Table 2 provides examples of COD removal efficiency and biogas production from AnMBR systems. Although AnMBRs are highly effective at removing total suspended solids (TSS) and COD, their efficiency in removing soluble nutrients, such as nitrogen and phosphorus, is limited. Approximately, 60-80% of these soluble nutrients are retained in the treated water (Jensen et al., 2015). In a related study conducted by Dai et al. (Dai et al., 2015), it was found that around 90% of the influent total nitrogen (TN) remained in the permeate.

The lower biosolid yield in anaerobic processes compared to aerobic processes can primarily be attributed to differences in microbial metabolic pathways, energy efficiency as well as significantly lower cell yield coefficients. In aerobic processes, a greater amount of energy is derived from the organic substrate to produce adenosine triphosphate (ATP), allowing for more substantial microbial biomass production. However, anaerobic processes typically involve anaerobic and fermentation metabolic pathways, which are inherently less energy-efficient, resulting in reduced ATP production per unit of organic substrate consumed. Due to this reduced ATP production, anaerobic microbes generate less biomass compared to their aerobic counterparts. Given that microbial biomass is composed of C, O, H, N, and P, lesser biomass production in the anaerobic processes implies lesser carbon is assimilated into microbial biomass, lesser O and H is taken up, and lesser N and P are used as nutrient by the anaerobic microbes per time compared to aerobic process. Interestingly, this perceived drawback of the AnMBR system can be advantageous as a nutrient source for agriculture (Fig. 2).

Table 2. Treatment efficiency and biogas production of AnMBR systems.

HRT (h)	**OLR kg COD/ m^3.d)**	**SRT (d)**	**MLSS (g COD/L)**	**Temperature (^{0}C)**	**COD removal (%)**	**Flux (LMH)**	**Biogas production (LCH_4/g COD removed**	**Ref.**
6–24			6.1–17.85	25	>90, >95% (BOD)	4.43–15.05	0.25–0.27	(Kong et al., 2021)
2.2	3		10.9	35	87	6	0.12	(Mei et al., 2018)
14.3	0.6–1.0		13–22	20–35	82–90	7	0.23–0.27	(Martinez-Sosa et al., 2011)
6–20		70	6–22	33.3	87	10	0.07	(Giménez et al., 2011)
13–14	1.6–2.0			18	~90	10–14	0.14–0.26	(Gouveia et al., 2015b)
7–17	2–2.5			18	87	10–15	0.18–0.23	(Gouveia et al., 2015a)
16		100	7.7		84	11–12	–	(Martin-Garcia et al., 2011)
4.6–6.8			0.6–1.2	8–30	81–94	4.1–7.5	–	(Shin et al., 2014)
3–6	1.5–3.0			25–30	64–71	100	0.35	(Quek et al., 2017)
7.5		60	12.8	25–30	86–89	13.5–18.1	0.3	(Yue et al., 2015)
12–14			6–10	25	88–90	3.08–11.42	0.15–0.2	(Ji et al., 2020)

1–8	0.8–3.0			20–25	71–77	22.5–180	0.08–0.12	(Yang et al., 2020)
8–14	1.3	60		13–32	88	7–22	0.11–0.18	(Lim et al., 2019)
8–48	0.3–2			25	~95	1.08–6.46	0.28–0.33	(Chen et al., 2017)
6–24				25	~95	2.31–9.26	0.3	(Lei et al., 2019)
57.6–86.4	0.00025–0.00196		3.3–5.6		90.9–95.8	3.5–4.8	0.214–0.322	(Vinardell et al., 2021)
6	1.8	50		18–23	60–73	13	0.37	(Plevri et al., 2023)
25–41	0.6–1.28	70		18–27	87	14–21	0.07–0.169	(Robles et al., 2022)
10–24	0.17		0.767		>90	8.8–20.8	–	(Dhiman et al., 2023)

3.1.3. ***Plant nutrients in AnMBR effluent***

The treated wastewater, which retains soluble nutrients, represent a valuable resource for plant cultivation. In hydroponics, where plants grow in nutrient-rich water rather than soil, the presence of these nutrients in AnMBR effluent can benefit plant growth and production. However, the nutrient composition of AnMBR effluent may exhibit variations depending on the influent wastewater characteristics and the efficiency of the treatment process. For instance, nitrogen may be present in the form of ammonia, and phosphorus may exist as orthophosphate. These nutrient forms must be in a bioavailable state for effective absorption by plants. Consequently, post-treatment processes may be employed further to modify nutrient forms and concentrations in the effluents, ensuring compliance with crop requirements. For example, there are some plants that can use nitrogen in an ammonia form or a subsequent nitrification step (tank or biofilter) can easily convert ammonia to nitrate but with added cost.

3.1.4. ***Energy production (biogas containing CH_4)***

One of the key advantages of AnMBR, compared to CAS, is its ability to generate methane-rich biogas. While in some cases, biogas may contain trace amounts of other gases (Constant et al., 1989), methane is predominantly released in the AnMBR system by methanogens within the reactor. The concentrations of produced methane, however, depend on waste composition, operating conditions, and the abundance of methanogens (Noyola et al., 2006). Due to thermodynamic gas-liquid equilibrium, the concentration of dissolved methane in the reactor and the methane in the headspace is contingent on temperature, partial pressure, and methane solubility. Nevertheless, the potential for energy production in AnMBR is well established. For instance, an estimated 2 kWh of energy can be generated by an AnMBR from the removal of 1 kg COD (Van Zyl et al., 2008). This level of energy production is nearly seven times more than what is required to operate the treatment system (Van Zyl et al., 2008). In a different study, it was found that around 0.32 kWh/m^3 of energy can be generated from an influent COD of approximately 600 mg/L, contributing significantly, about 98%, to the required energy (Zhang et al., 2023). It's crucial to highlight that if the AnMBR effluent cannot be directly utilized, additional post-treatment is necessary, potentially leading to an overall increase in the energy consumption of the treatment process.

Studies have investigated the biogas generation in AnMBR systems under various conditions. For instance, Hu et al. (Hu et al., 2018) reported biogas yields in AnMBR systems, with methane contents ranging from 70 to 80%. Challenges, such as low temperatures (<20 °C) (McKeown et al., 2012) and the presence of high levels of particulate organic substances, which are often non-/slowly degraded (Zhang et al., 2018), can impact microbial activities and result in lower biogas yields. Methane production may also be hindered by sulfate-reducing bacteria, which can compete with methanogens for COD (Giménez et al., 2012). Martin et al. (Martin et al., 2011) found that increasing the COD strength from 0.24 to 10 g COD/L can boost biogas energy generation significantly, from 0.62 to 34.8 kWh/m^3. For instance, in domestic wastewater, Shin and Bae (Shin and Bae, 2018) observed that the theoretical energy potential was proportional to influent COD levels. They analyzed both theoretical and measured energy potentials under various influent COD concentrations, and showed that the measured energy potentials reached approximately 70–80% of theoretical values, particularly under conditions of high influent COD/SO_4^{2-}-S ratios. Moreover, the addition of flux enhancers, such as coagulants for fouling control, can reduce organic content, consequently affecting CH_4 production negatively. In another study by Galib et al. (Galib et al., 2016), assuming an energy conversion efficiency to approximately 40% from heat to electrical energy, net energy benefits between 0.16 kWh/m^3 and 1.82 kWh/m^3 were achieved. These findings highlight the significant potential of AnMBRs to yield energy-positive outcomes.

Despite these challenges, researchers have explored strategies to enhance methane production from AnMBR systems. Gouveia et al. (Gouveia et al., 2015a) utilized turbulence in the membrane module to maintain a consistent temperature in the reactor, enabling a higher methane production in a pilot-scale AnMBR than in a UASB reactor for municipal wastewater treatment. Generally, operating the AnMBR at higher temperatures has been reported to improve methane production (Evans et al., 2018). Nevertheless, elevated turbulence and temperature levels would require increased energy consumption. Therefore, it is essential to optimize these factors to ensure efficient energy utilization. Other studies have investigated the impact of operational parameters, such as HRT, on methane yield, with reduced HRTs favoring the growth and accumulation of specific methanogens, thus maximizing effluent quality and methane yield (Gouveia et al., 2015a).

3.1.5. ***Energy and carbon footprints***

Compared to CAS systems, AnMBRs typically require less energy due to their operation under anaerobic conditions, eliminating the need for a continuous oxygen supply. In the CAS system, aeration constitutes a significant portion of the total energy consumption (Au et al., 2013; Sean et al., 2020). The reduced energy requirement of AnMBRs results in lower GHG emissions, especially when fossil fuels are the energy source. Moreover, the reduced sludge production associated with AnMBRs contributes to a decreased carbon footprint in terms of sludge treatment and disposal. However, AnMBRs can generate methane, a potent GHG.

The consideration of direct and indirect greenhouse gas (GHG) emissions, stemming from methane and CO_2 release in AnMBR applications for wastewater treatment, emerges as a significant concern for large-scale implementation. The GHG footprint of AnMBR systems is subject to variation, contingent upon factors such as the specific characteristics of the treated wastewater, operational conditions, and the effectiveness of methane recovery.

Around 80% of produced methane can be dissolved in the effluent, particularly when AnMBR operates under low temperatures (Cookney et al., 2016; Giménez et al., 2014). This substantial loss poses a dual challenge, contributing to a carbon footprint and reduced system energy efficiency as the released methane eventually enters the environment. Therefore, recovery and efficient utilization of methane from the headspace and dissolved effluent, are important for energy efficiency and environmental impact mitigation.

3.1.6. ***Methane recovery and utilization***

Addressing the challenges associated with dissolved methane in effluent and optimizing its recovery in AnMBR systems have prompted the exploration of various techniques. These approaches encompass aeration, gas stripping, membrane contactors, biological oxidation (e.g., aerobic methane oxidation), and microbial fuel cells. Among these, gas phase membrane-based recovery stands out as a promising method due to its substantial mass transfer area and ease of operation (Rongwong et al., 2018), achieving approximately 99% dissolved methane recovery (Velasco et al., 2018). Subsequently, degassing membrane technology can be employed to recover the retained dissolved methane, with the separation mechanism guided by Fick's law and pressure drop across the membrane (Crone et al., 2016; Gabelman and Hwang, 1999).

Advancements in membrane technology, particularly the modification of membrane surfaces, can enhance methane recovery and mitigating challenges associated with membrane-based methane recovery, such as membrane pore wetting, fouling, stability, etc. (Sethunga et al., 2018). However, the economic viability and energy utilization aspects of these innovations require further exploration and validation to establish their practicality and effectiveness on a larger scale.

The recovered methane can be effectively utilized to enhance the energy efficiency of the AnMBR process, primarily by serving as a source for heating and power generation. Extensive biogas upgrading is often unnecessary when utilizing biogas recovered from the AnMBR process for boiler and combined heat and power (CHP) applications (Petersson and Wellinger, 2009). This not only boosts energy efficiency but also substantially reduces the additional costs associated with biogas upgrading. The ability to efficiently utilize methane in biogas is a key factor in making AnMBR systems sustainable and environmentally friendly.

3.1.7. ***Decentralized/distributed AnMBR facilities***

In contrast to centralized wastewater treatment methods, decentralized wastewater treatment plants offer an advantage by eliminating the necessity for extensive distribution networks and numerous lift stations. This not only results in energy savings but also significantly reduces capital costs (Bernal et al., 2021; Garrido-Baserba et al., 2022). Moreover, decentralized wastewater treatment enables immediate on-site water reuse, further diminishing energy expenses associated with transportation and distribution. Additionally, it bypasses concerns related to microbial regrowth within distribution networks. Although decentralized wastewater treatment has predominantly focused on aerobic membrane bioreactors due to their ability to be operated to produce higher effluent quality, aerobic processes have substantial sludge production, elevated energy consumption, and GHG emissions (Xiao et al., 2019).

In contrast, AnMBR systems provide higher energy efficiency, decreased sludge production, and the potential for nutrient-rich effluent. The prospect of sewer mining in peri-urban areas aligns seamlessly with the principles of a circular economy and water security.

Despite these advantages, decentralized systems, including AnMBRs, do present challenges related to operation and maintenance. Furthermore, each decentralized unit may need to be permitted according to local environmental regulations, which increases the paperwork and monitoring compared to centralized systems. Close collaboration with regulatory authorities would be important to ensure compliance and create a manageable regulatory framework.

3.2. **Hydroponic controlled environment agriculture (CEA) technology**

Hydroponic CEA is becoming popular for its ability to cultivate crops under controlled conditions (e.g., light, temperature) without soil. Compared to land/soil-based agriculture, CEA utilizes less water, requires a much smaller area (i.e., land footprint) for crop cultivation, and utilizes nutrients more efficiently for plant growth (Cetegen and Stuber, 2021). The recirculation of nutrient-rich water through hydroponic modes of operation such as the nutrient film technique (NFT), which could be enabled by automation, allows plants to absorb the necessary elements, reducing the risk of nutrient runoff and its associated environmental consequences. This closed-loop system also reduces the amount of synthetic fertilizers needed, which can mitigate nutrient pollution in surrounding ecosystems. Water reuse in hydroponic systems can contribute to overall resource efficiency (Richa et al., 2020). The reduced need for water lowers the demand on local water sources, making hydroponics a viable option in water-scarce regions. Moreover, the recirculation of nutrient water minimizes the energy required for municipal water pumping and distribution, contributing to energy savings. This synergy between water and energy efficiency is a critical factor in the sustainability of hydroponic systems (Avgoustaki and Xydis, 2020). A hydroponic system offers advantages over conventional agriculture due to the elimination of soil-borne pathogens in the system (Sela Saldinger et al., 2023). Additionally, the hydroponic approach completely avoids the conflicting nutrient balance, where high levels of certain nutrients can lead to the deficiency of other vital nutrients (Abbas et al., 2022) Furthermore, the hydroponic system facilitates nutrients readily available for plant uptake (Dotaniya et al., 2023).

While water reuse in hydroponics offers numerous advantages, there are challenges that must be addressed. One of the primary concerns is the accumulation of plant and human pathogens. As water is continually reused, plant pathogens can proliferate, leading to pest outbreaks. Additionally, the accumulation of salts can negatively impact plant health and growth (Warrence et al., 2002).

It is important to note that media quality is critical to the success of hydroponic systems. The media refer to the nutrient-rich water that circulates through the system and/or inert media that supports the plants. Maintaining a high-quality medium is essential for successful crop production, nutrient efficiency, disease prevention, system longevity, and environmental sustainability. Proper nutrient management, pH and electrical conductivity (EC) control, and effective disinfection measures can ensure that the nutrient solution remains balanced and conducive to healthy plant growth.

*3.2.1. **Modular containers and greenhouses for hydroponic CEA***

Both modular containers are alternatives to traditional farming, addressing challenges in maintaining crop quality from production to consumption. Modular container farms, constructed from repurposed shipping containers, boast high portability and versatility, operating in diverse locations with ease. Prebuilt and requiring minimal setup effort compared to greenhouses, these farms utilize hydroponic systems to grow crops without soil, promoting vertical farming and multi-layer cultivation. High-powered light-emitting diodes and CEA technology further enhance control over the growing environment, regulating humidity, temperature, and carbon dioxide levels.

In contrast, greenhouses, are structured with glass walls and roofing or polyethylene covers (hoop houses), leveraging sunlight as a primary light source (while supplemental lighting is crucial). While larger than modular container farms, greenhouses allow flexibility in cultivation methods and can support hydroponic or soil growth. However, their translucent walls pose challenges in stacking layers without hindering sun exposure to lower levels. While proficient at heat retention, greenhouses may struggle with extreme temperatures, often relying on passive cooling methods.

Both modular container farms and greenhouses offer unique strengths and challenges. Modular container farms, known for portability and compactness, provide placement flexibility, whereas greenhouses, rooted in history and natural light reliance, require larger cultivation spaces. The optimal choice depends on specific needs, emphasizing factors such as size, portability, and the desired level of control over the growing environment. Zhang and Kacira (Zhang and Kacira, 2020) reported that the energy efficiency of modular containers can rival that of greenhouse systems in cold climates. However, in hot climates, greenhouses outperform modular containers significantly in terms of energy usage. The transpiration of plants and the heat generated by the sole source lighting system have a significant impact on the cooling and heating requirements of the heating, ventilation, and air conditioning (HVAC) system in modular containers (Zhang and Kacira, 2020).

3.2.2. ***Rural, urban, and peri-urban CEA***

Location of CEA in urban and peri-urban areas is a potential means to help tackle challenges arising from rapid urbanization and its impact on traditional food production and distribution systems. With the global population increasingly concentrating in urban areas, traditional agriculture encounters difficulties due to diminishing agricultural land and the extended distances involved in transportation. This conventional agricultural approach, characterized by large-scale industrialized methods, contributes to environmental degradation, deforestation, and GHG emissions, particularly from long-distance food transportation (Sarabia et al., 2021).

In response to these challenges, urban and peri-urban CEA has evolved as a transformative solution, reshaping urban areas into self-sufficient and sustainable systems. Urban and peri-urban CEA involves cultivating crops in controlled environment within or on the outskirts of cities, utilizing available spaces. Employing techniques – especially hydroponics – urban and peri-urban CEA optimizes space utilization and resource efficiency, promoting green practices and mitigating the environmental consequences of traditional food systems. Similarly, rural CEA can also be achieved through the utilization of disused buildings on farms or abandoned warehouses in peri-urban areas. This would allow farmers to diversify their income and make the most of the resources at their disposal.

This innovative farming approach not only addresses urbanization challenges but also enhances security and resilience in food supply chains. Traditionally, crops are grown in rural areas, necessitating energy for transportation to urban cities. In contrast, urban and peri-urban CEA minimizes transportation distances by assuming local consumption of produce, while also alleviating *food deserts*. This not only reduces GHG emissions but also fosters local economies through increased employment opportunities.

The significance of urban and peri-urban farming is underscored during times of uncertainty, as seen in the COVID-19 pandemic, where it played a vital role in ensuring food security for low-income communities, supporting community reconstruction, fostering education and empowerment, and contributing to environmental conservation (Feola et al., 2020).

3.2.3. ***Food crops cultivated and nutrient delivery in hydroponic CEA***

Examples of crops suitable for hydroponic CEA include various vegetables (carrots, tomatoes, potatoes, cucumbers, amaranthus, spinach, lettuce, parsley), herbs (aloe vera, mint, basil), and cereals (wheat, rice), among others. Each crop has specific nutritional requirements (Song et al., 2024) and environmental conditions, including temperature, humidity, and carbon dioxide.

In the realm of hydroponic CEA, the strategic application of fertilizers becomes instrumental in optimizing crop growth and maximizing yields. CEA offers efficient application of fertilizers and other micro nutrients to plant roots for uptake. The direct application of these customized solutions to the plant root zone not only enhances nutrient uptake but also minimizes waste, contributing to resource efficiency (Rajaseger et al., 2023). There are alternative modes of nutrient delivery in hydroponic systems, e.g., nutrient film technique (NFT) and deep water culture (DWC), among others.

Fertilizers utilized in hydroponic CEA typically fall into two main categories: chemical/synthetic fertilizers and organic fertilizers (Ahmed et al., 2021). Chemical fertilizers are further categorized into macronutrients and micronutrients. Macronutrients include primary elements like nitrogen (N as NO_3^-, NH_4^+), potassium (K as K^+), and phosphorus (P as $H_2PO_4^-$, HPO_4^{2-}, PO_4^{3-}), as well as secondary elements like calcium (Ca as Ca^{2+}), magnesium (Mg as Mg^{2+}), and sulfur (S as SO_4^{2-}). Micronutrients encompass iron (Fe as Fe^{3+}), manganese (Mn as Mn^{2+}), zinc (Zn as Zn^{2+}), copper (Cu as Cu^{+}, Cu^{2+}), and molybdenum (Mo as MoO_4^{2-}).

3.2.4. ***Energy and GHG footprints***

Hydroponic CEA systems are energy-intensity, attributed to various factors such as lighting, ventilation, cooling, and pumping. A specific example illustrates this; with approximately 15 kWh of energy needed for the cultivation of 1 kg of lettuce in a hydroponic CEA, potentially resulting in up to 17.8 kg of CO_2 emissions (Casey et al., 2022). In another study, Barbosa et al. (Lages Barbosa et al., 2015) compared the energy requirements of lettuce grown using hydroponic systems versus conventional agriculture. The study reported that, despite lower lettuce yield and water efficiency in conventional agriculture, hydroponic agriculture requires 82 times more energy than its conventional counterpart. The environmental impact of hydroponic CEA systems is intricate and is predominantly shaped by energy consumption and the source of that energy. While ongoing efforts to optimize energy efficiency within hydroponic CEA are crucial (Zhang and Kacira, 2020), a pivotal shift towards renewable energy sources is equally imperative. This transition is essential for a comprehensive evaluation of the GHG footprint of hydroponic CEA systems, ensuring a thorough understanding of their sustainability.

The intricate relationship between energy consumption and GHG emissions in hydroponic CEA requires a holistic assessment. Therefore, continual efforts to minimize the energy demands of hydroponic CEA must be coupled with a strategic adoption of renewable energy sources to truly align with sustainability goals.

3.3. **Perspectives on the integration of AnMBR and hydroponic CEA**

The integration of both AnMBR and hydroponic CEA would address land and water scarcity issues, especially in urban and peri-urban areas while simultaneously reducing food transportation distances. Both hydroponic CEA and AnMBR systems aim to maximize productivity within smaller footprints, aligning with circular economy ambitions. For example, AnMBR is modular, readily facilitating scale-up or scale-down in capacity. Similarly, modular containers housing hydroponic systems could be strategically stacked in compact urban spaces near decentralized AnMBR systems for wastewater treatment, creating a synergistic and sustainable urban/peri-urban agricultural and wastewater treatment model.

*3.3.1. **Hydroponic CEA regulatory challenges***

A significant gap exists when it comes to guidelines specifically suited for hydroponic CEA. This deficiency became evident in the study conducted by Boonnorat et al (Boonnorat et al., 2021), where they compared the phytotoxicity of wastewater effluent from CAS, bioreactor, and membrane bioreactor systems. Their research revealed that golden pothos exhibited lower phytotoxicity when cultivated in soil compared to its hydroponic counterpart (Fig. 3). This disparity was attributed to the presence of microorganisms in the soil, which actively participated in the degradation of potentially toxic materials. Furthermore, in a soil-based system, contaminants can adsorb onto the soil particles, reducing their availability to plants. In contrast, in a hydroponic system, plants had direct access to these contaminants, leading to their uptake and consequently resulting in more detrimental effects. A separate study examined the uptake and translocation of di-n-butyl phthalate (DBP) by six different vegetables under hydroponic conditions (Li et al., 2019). DBP was detected in both the roots and shoots of all six vegetables, with concentrations varying depending on the vegetable variety and tissue type. The concentration of DBP in the roots exceeded that in the shoots for all vegetables, indicating poor translocation of DBP from roots to shoots. These findings collectively suggest that DBP is readily absorbed into the roots and shoots of the studied vegetables. Kovačič et al. (Kovačič et al., 2023), investigated the uptake of fourteen contaminants of concern (CECs) and twenty-seven potentially toxic elements (PTEs) in tomatoes cultivated in both soil and hydroponic systems. Their findings revealed the presence of bisphenol S, 2,4 bisphenol F, and naproxen in the tomatoes, with higher levels detected in hydroponically grown tomatoes.

These underscore the necessity of distinct wastewater treatment regulations for hydroponic agriculture, as the conventional guidelines established for soil-based agriculture may not be applicable in this context.

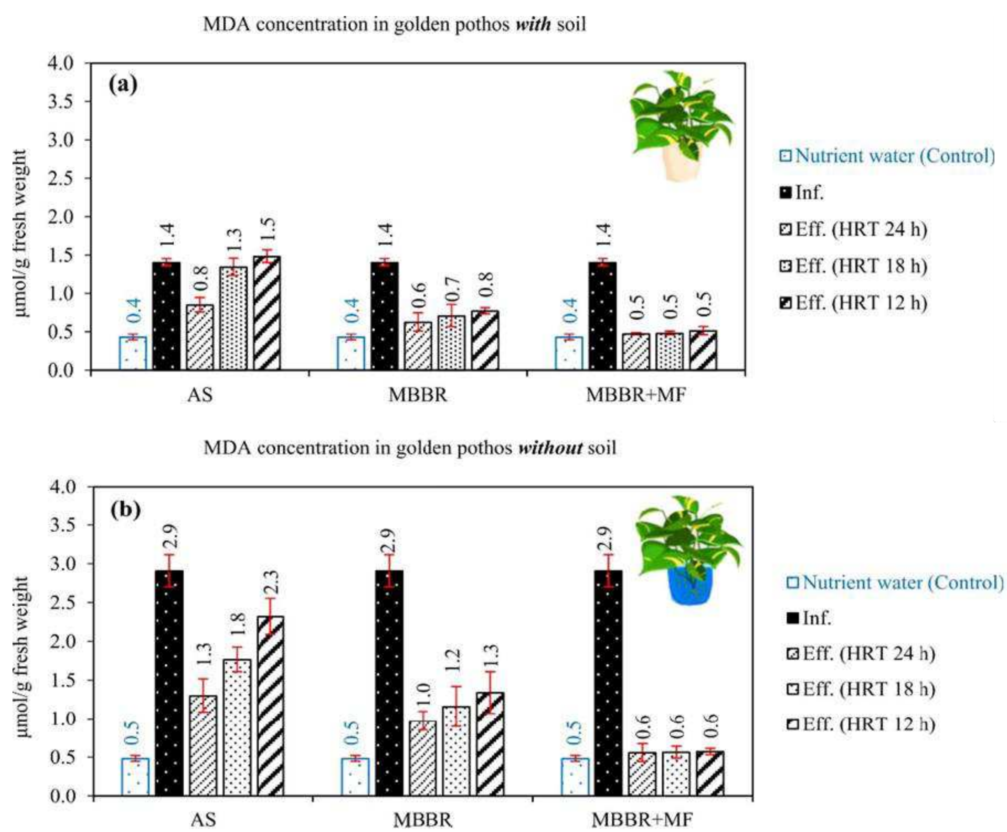


Fig. 3. Malondialdyde (MDA), a biomarker for stress and cell membrane damage in plant, concentrations in golden pothos (Boonnorat et al., 2021). (a) soil-based cultivation, (b) hydroponically grown (Reprinted with permission from Elsevier).

Many wastewater reuse regulations were designed with traditional soil-based agriculture in mind. Therefore, one of the primary challenges faced by hydroponic CEA is that conventionally treated effluent is not suitable for use due to the presence of contaminants.

To use treated wastewater effluent for hydroponic agriculture, there is a pressing need for stringent regulations that specifically address pathogen and contaminant levels. In addition to regulatory challenges, public perception and attitudes can pose challenges. Addressing these perceptions and demonstrating the safety of such practices is essential for gaining consumer acceptance and regulatory approval. Another concern is the limited requirement for comprehensive testing of emerging contaminants, such as pharmaceuticals, and personal care products under existing regulations. This regulatory gap may raise concerns about the potential presence of these substances in hydroponic CEA water.

Complying with regulatory requirements for wastewater treatment and maintaining water quality in hydroponic CEA can be a complex and costly endeavor. Hence, growers bear the responsibility of rigorously testing their water sources for potential contaminants that could pose risks to both crops and human health. Smaller or less well-funded hydroponic operations may struggle to invest in the necessary infrastructure and expertise required for compliance, potentially limiting their ability to use treated wastewater effectively. Furthermore, local zoning and land use regulations may not always align with the goals of hydroponic CEA operations, including the use of treated wastewater. Zoning restrictions and land use regulations can present additional barriers to implementing hydroponic systems that rely on unconventional water sources. Regulations governing wastewater reuse may lag advancements in hydroponic CEA technology and wastewater treatment methods. As CEA practices evolve, regulations must adapt to ensure the safety and sustainability of these systems. This requires proactive collaboration among industry and consumer stakeholders, regulatory agencies, and research institutions to keep regulations up-to-date and relevant.

Addressing these regulatory gaps and limitations necessitates a collaborative effort involving hydroponic CEA operators, wastewater treatment facilities, regulatory agencies, and research institutions. Developing clear and science-based guidelines specifically tailored to hydroponic CEA, along with comprehensive monitoring and reporting requirements, can promote the safe and sustainable use of treated wastewater in this context. Additionally, public education and outreach efforts can help address concerns and build confidence in the safety of hydroponically grown produce, fostering greater acceptance of this environmentally responsible cultivation method.

3.3.2. ***Water quality of biological systems for hydroponic CEA***

Biological treatment processes have been extensively studied for their wastewater treatment efficiency and have consistently yielded excellent results (Khalidi-Idrissi et al., 2023). However, certain biological processes face challenges in efficiently removing emerging contaminants from both the effluent and biosolids.

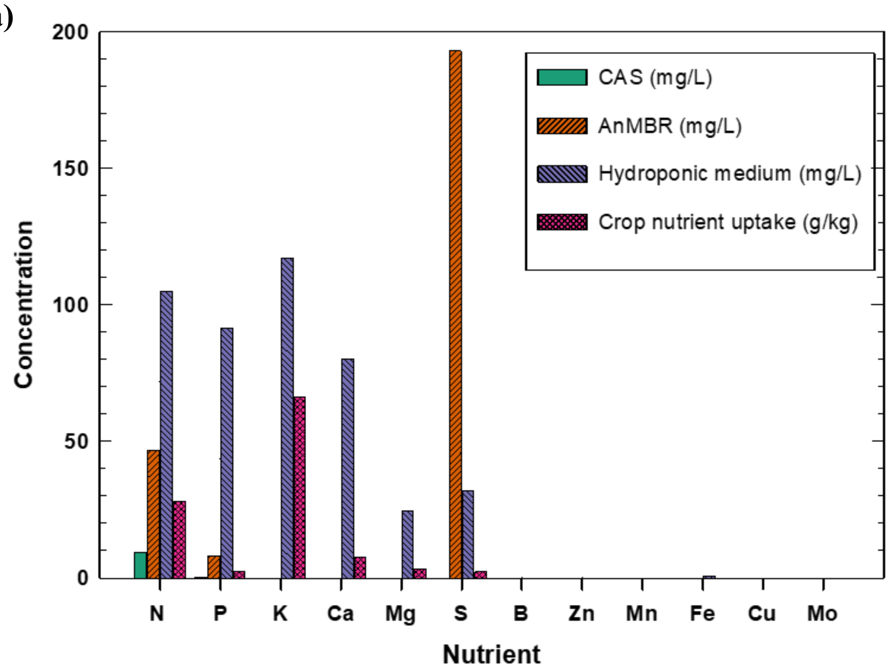
A study by Kreuzig et al. (Kreuzig et al., 2021) investigated the efficiency of various wastewater treatment processes in removing organic micropollutants, such as acesulfame, caffeine, carbamazepine, diclofenac, ibuprofen, sulfamethoxazole, acetyl-sulfamethoxazole, 1 H-benzotriazole, and 4/5-methylbenzotriazole, from effluents used in the hydroponic cultivation of lettuce. The results were indicative that conventional wastewater treatment methods, including the use of expanded granular sludge bed reactors (EGSB), sequencing batch reactors (SBR), and biological activated carbon filtration reactors (BACF), were often ineffective in the removal of micropollutants. Some of these treatment processes displayed as low as 0% removal efficiency for certain micropollutants. Inefficiencies in removing micropollutants like carbamazepine led to the uptake of these substances by lettuce, subsequently translocating within the plant's system. Another study further demonstrated that alfalfa, cultivated under hydroponic conditions, could uptake and translocate sulfamethazine (Kurwadkar et al., 2017). Nonetheless, the integration of membranes with biological processes has played a crucial role in mitigating risks in hydroponically grown crops. Notably, when a membrane system was combined with a biological system, it effectively eliminated the risks associated with crop stress and cell membrane damage that were previously linked to inadequate removal of micropollutants in treated wastewater effluents from CAS and biological systems (Fig. 4) (Boonnorat et al., 2021). However, it is important to mention that biodegradation varies according to aerobic, anoxic, and anaerobic conditions.

Another significant concern in biological treatment processes is the presence and impact of ARB and ARGs especially due to the rising concentrations of antibiotics in wastewater (Abdul and Lloyd, 1985). This concern arises because microbes involved in biological treatment processes may naturally possess ARGs or acquire them through prior exposure to antibiotics in humans or animals. Consequently, if precautions are not taken, some biological wastewater treatment processes may inadvertently become hubs for ARB and ARGs production. While ARB can be removed relatively easily, ARGs tend to persist due to their smaller sizes and non-living properties. Furthermore, there is a risk that some of these ARB and ARGs may end up in the effluent or biosolids, posing a significant threat to human health. The introduction of membrane processes, including microfiltration and ultrafiltration, has significantly improved the removal efficiency of ARB, ARGs and emerging contaminants in the effluent compared to conventional wastewater treatment processes (Radjenović et al., 2009). However, it's important to note that these contaminants may still accumulate in the wasted activated sludge, which could potentially find its way back into the environment (Fijalkowski et al., 2017).

3.3.2.1.Water quality of AnMBR effluent for hydroponic CEA

AnMBRs have emerged as a promising wastewater treatment process because they generate minimal (in large-scale system) to no sludge (in small-scale system with no sludge wastage), allowing for long-term operation without interruptions. As mentioned earlier, in addition to providing water suitable for agricultural purposes, AnMBR exhibits the capacity to supply essential nutrients like nitrogen, phosphorus, and other nutrients crucial for supporting plant growth. (Song et al., 2018). While there are undeniable benefits, challenges may arise concerning nutrient balance and concentrations. The nutrient concentrations for lettuce cultivation in hydroponic media is often higher than that in CAS and AnMBR effluents (Fig. 4(a)). This is attributed to the need to supply sufficient nutrients to the crops and the desire to promote more rapid plant growth and yield. However, the disparity between hydroponic media and actual crop needs highlights inadequate nutrient management within the hydroponic system. For example, in a zero-discharge hydroponic system, only 10.9% of phosphorus and 13.5% of nitrogen were assimilated into the lettuce crop, while 15% phosphorus and 77.6% nitrogen were lost (unaccounted for) (Fig. 4(b)) (Yang and Kim, 2020). Nitrogen loss occurred through denitrification into the atmosphere, while phosphorus was lost via precipitation (Yang and Kim, 2020). Addressing this significant nutrient disparity requires effective nutrient management practices. Furthermore, adjustments and augmentations to AnMBR effluent are also essential to meet the specific nutrient requirements of hydroponically grown crops.

(a)



(b)

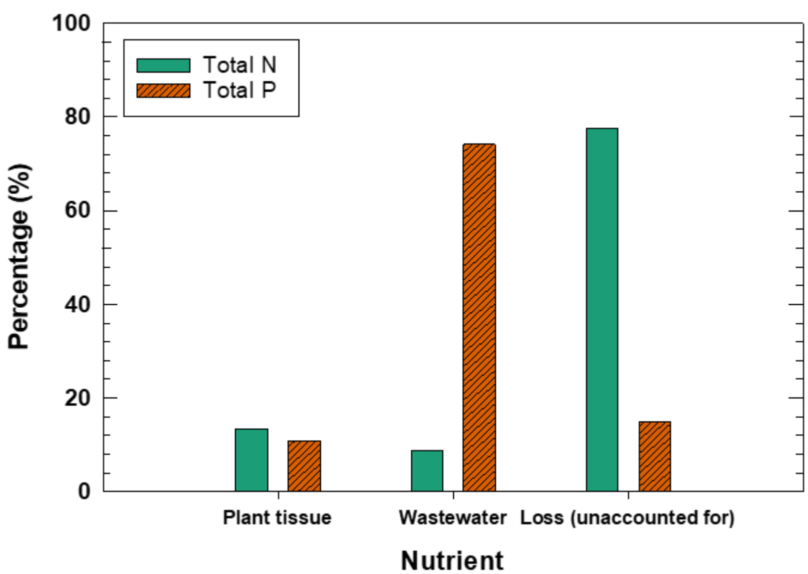


Fig. 4(a) Comparison of nutrient content in effluent water from AnMBR and CAS, along with the nutrient requirements for crisphead lettuce and the nutrient uptake of lettuce leaves at harvest. Data from (Ahmed et al., 2021; Bertanza et al., 2017; Robles et al., 2022; Sublett et al., 2018), (b) The proportional allocation of overall nitrogen and phosphorus in a hydroponic system used for cultivating lettuce (data from (Yang and Kim, 2020)).

The microbial communities within anaerobic digestion can adapt to degrade a wide range of organic substances over time, and the coupled membrane technology can help retain emerging contaminants adsorbed onto the biomass and biological-based contaminants within the reactor until they are fully degraded, preventing their release into the effluent. For example, AnMBR has exhibited impressive log_{10} removal values of 5.2 and 6.1 for *Escherichia coli* and enterococci, respectively (Wong et al., 2009). Additionally, a log removal value of 3.6 has been reported for viruses (Zhang et al., 2022). Studies have demonstrated the efficiency of AnMBRs in removing emerging contaminants (Abargues et al., 2012; BouNehme Sawaya and Harb, 2021). This enhanced efficiency is not limited to emerging contaminants but also extends to ARB and ARGs. In fact, AnMBRs have been reported to exhibit higher ARG removal compared to their aerobic MBR counterpart. The superior performance of AnMBR over aerobic MBR was validated by Harb et al. in their study, where ARG concentrations were log 1-2 lower in AnMBRs compared to aerobic MBR (Harb et al., 2016). Monsalvo et al. (Monsalvo et al., 2014) investigated the efficacy of removing 38 organic micropollutants (OMPs) in an AnMBR system, revealing that 9 OMPs exhibited removal rates exceeding 90%. This outcome is unsurprising, given that the degradation of OMPs in AnMBR occurs through co-metabolism. In this process, enzymes generated during the metabolism of other compounds are utilized for the biodegradation of OMPs (Hazen, 2010). Co-metabolism involves the utilization of external organic matter as carbon and energy sources to degrade OMPs partially or completely in wastewater. AnMBR demonstrates a higher efficiency in the biodegradation of antibiotic-type OMPs compared to its aerobic MBR counterpart. This is attributed to the lower prevalence of antibiotic resistance genes in anaerobic conditions (Amha et al., 2019; Harb et al., 2019; Harb et al., 2016). The removal efficiency of OMPs in AnMBR can be enhanced through the integration of post-treatment technologies like membrane processes and advanced oxidation processes (AOPs). These methods enable the removal of OMPs that may not have been biodegraded, employing additional physicochemical approaches for effective elimination. However, there is still the concern of potential plant uptake/accumulation of OMPs.

The growing interest in AnMBR wastewater treatment for agricultural purposes is driven by the global shift towards achieving net-zero emissions and promoting a circular economy. In an AnMBR system, net-zero emissions and circular economy goal can be achieved by efficiently capturing and utilizing methane-rich biogas, often through combined heat and power (CHP) generation, as well as capturing nutrients in treated water for agricultural reuse. Additionally, energy consumption can be minimized through optimized operation and energy-efficient infrastructure.

However, utilizing AnMBR effluent for hydroponic CEA introduces a potential challenge: the presence of dissolved methane and hydrogen sulfide in the effluent. It is noteworthy that while methane at low levels has been reported to play a role in regulating plant physiology, promoting root development, and delaying senescence and browning (Kou et al., 2018; Li et al., 2020), and low concentrations of hydrogen sulfide can assist plants in responding to abiotic stresses such as heavy metals, salinity, drought, and extreme temperatures (Huang et al., 2021; Kou et al., 2018; Li et al., 2022; Singh et al., 2020). However, high levels of these dissolved gases may negatively impact crops, especially primary root growth (Branscombe, 2016; Jia et al., 2015; Zhang et al., 2017).

In the AnMBR effluent, a significant amount of methane is dissolved, with the concentration depending on the operating temperature (as shown in Fig. 5). Previous studies have indicated that, at a temperature of 30 °C, the theoretical dissolved methane level in AnMBR effluent can represent up to 45% of the methane produced in the system. (Li et al., 2021; Liu et al., 2014). While increasing the operational temperature can reduce the concentration of dissolved methane in the effluent, it's important to note that the effects of methane on hydroponically grown crops at high concentrations are not well known.

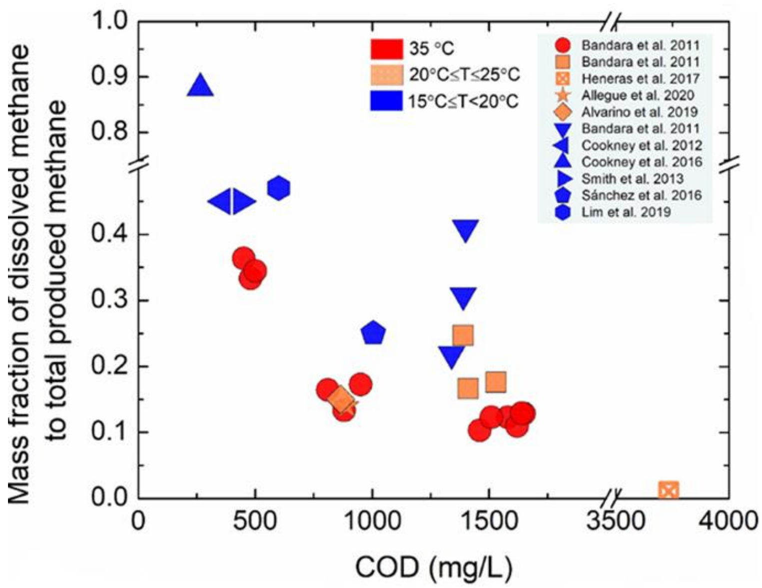


Fig. 5. The proportion of methane dissolved in relation to the overall methane production (Li et al., 2021) (Reprinted with permission from Elsevier).

The predominant inorganic nitrogen form found in AnMBR effluent is often ammonium rather than nitrate. Different crops exhibit preferences for either ammonium or nitrate, and these nutrients have distinct effects on plant growth and physiology (Coruzzi and Bush, 2001; Lee et al., 2021; Miller and Cramer, 2005). In a theoretical context, ammonium appears to be a more efficient alternative to nitrate due to its metabolic efficiency, requiring less energy for assimilation (Middleton and Smith, 1979; Song et al., 2022a). However, the response to these nitrogen forms can vary among plant species and varieties, with some being highly sensitive to ammonium and others favoring ammonium over nitrate as their primary nitrogen source (Lee et al., 2021). Paradoxically, it has been observed that most plants cannot thrive on ammonium alone as their sole nitrogen source and typically require some nitrate supplementation (Song et al., 2022a). Importantly, elevated levels of ammonium can have adverse effects on certain plants, leading to stunting and leaf chlorosis (yellowing) and necrosis (tissue death) (Song et al., 2022b). This underscores the need for careful management of nutrient levels in hydroponic systems utilizing AnMBR-treated wastewater. Achieving the right balance between ammonium and nitrate, along with other essential nutrients, is critical to ensuring optimal plant health and growth. In practice, understanding the specific nutrient requirements of the crops being cultivated in hydroponic systems and monitoring and adjusting nutrient levels accordingly is essential for maximizing crop yields and quality while minimizing the risk of nutrient-related issues.

3.3.3. ***Management of effluent dissolved GHG***

Elevated levels of methane in the AnMBR effluent could potentially affect nutrient uptake efficiency in plants, especially for essential nutrients like phosphorus. It's crucial to recognize that different plant species and varieties may respond differently to the presence of methane in hydroponic solutions, with some being more tolerant and others more sensitive to methane exposure. Therefore, it is essential to continually monitor methane concentrations in the effluent and investigate their effects on plant growth. However, adopting proactive and sustainable approaches involves options such as recovering dissolved methane from the effluent, employing advanced oxidation processes for its removal, or converting it to CO_2 for the benefit of plants. These strategies contribute to advancing the AnMBR system towards greater sustainability.

Similarly, hydrogen sulfide is a volatile, toxic, and malodorous compound. It is imperative to reduce the concentration of hydrogen sulfide in the AnMBR effluent below phytotoxic levels before its utilization in hydroponic systems. Effective measures must be taken to ensure that the potential benefits of using AnMBR effluent for hydroponic CEA are not compromised by the presence of these gases at harmful concentrations.

3.3.4. ***Use of recovered AnMBR energy in hydroponic CEA***

The energy recovered from the AnMBR system holds the potential for multifaceted applications, not only benefiting the wastewater treatment process itself but also contributing significantly to the energy efficiency and sustainability of hydroponic CEA. Beyond the intrinsic benefits discussed earlier, such as energy self-sufficiency in the AnMBR system, this recuperated energy can be harnessed in various ways within hydroponic CEA, amplifying its overall ecological impact.

One prominent avenue for utilizing the recovered energy, often in the form of methane, involves combustion to release both CO_2 and heat. This strategic deployment of methane-derived energy serves several crucial purposes within the hydroponic CEA framework. Firstly, the released CO_2 and heat can be directed towards CO_2 fertigating and heating the CEA environment and wastewater to provide mesophilic conditions, ensuring optimal conditions for both plant growth, particularly during colder periods and wastewater treatment. This not only fosters a conducive atmosphere for plant development but also helps to regulate and maintain the desired temperatures essential for hydroponic cultivation. Additionally, the produced CO_2 from the combustion of methane emerges as a valuable resource in the hydroponic CEA context. This enriched CO_2 can be strategically introduced into the growing environment to enhance photosynthesis.

Moreover, the recovered methane can be effectively employed to modulate the temperature of the nutrient solution, a critical element in hydroponic systems. By utilizing the energy generated from methane, the nutrient solution can be kept within the optimal temperature range, promoting an environment conducive to nutrient absorption and plant health. Nevertheless, elevated sulfate concentrations in the feedwater have the potential to diminish methane production (Giménez et al., 2012).

A further innovative application involves supplementing the COD in the AnMBR to levels sufficient to produce ample biogas (Hu et al., 2020). This supplementation can be achieved using plant harvest waste (after grinding) sourced from the hydroponic CEA system. This biogas can then be converted into electricity through CHP systems or generators or upgraded to produce biomethane by removing other components such as carbon dioxide, air, hydrogen sulfide, volatile organic compounds, etc. This electricity can be channeled to power diverse components of the CEA facility, ranging from lighting systems to nutrient solution pumps and temperature control mechanisms. By integrating this electricity into the CEA infrastructure, a self-sustaining and energy-efficient cycle is established, reducing external energy dependencies.

Fig. 6 illustrates the synergistic integration of hydroponic CEA and the AnMBR system, highlighting key challenges in the process.

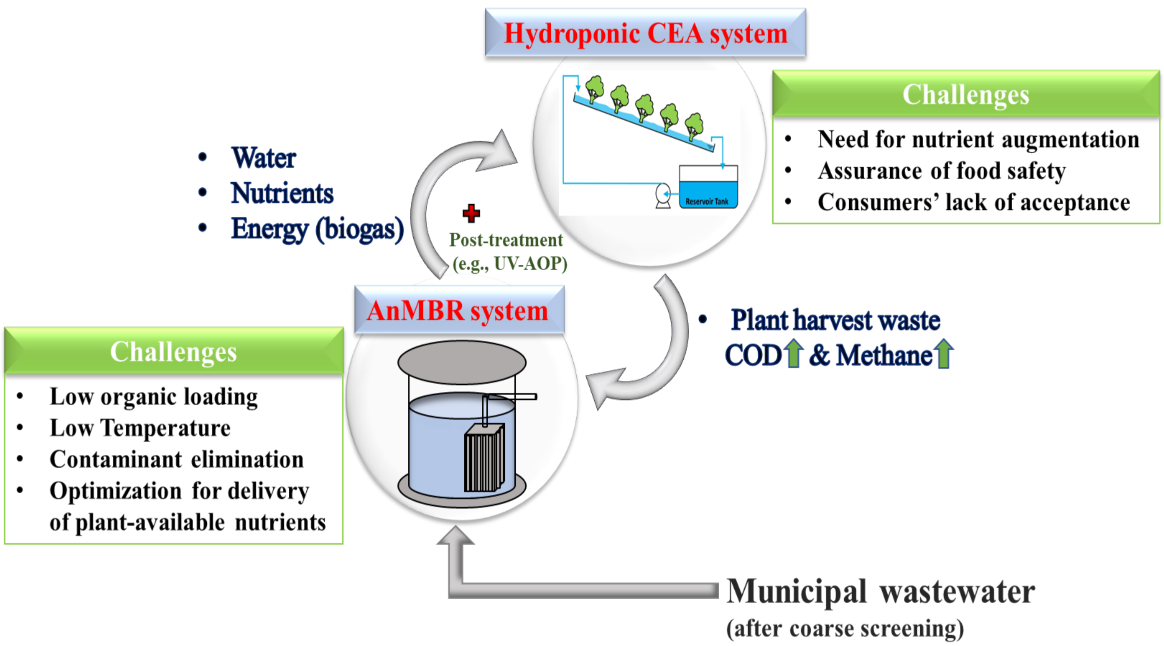


Fig. 6. Integration summary: hydroponic CEA and AnMBR systems synergistically aimed at maximizing productivity within compact footprints, in alignment with circular economy ambitions.

4. Future research efforts

While AnMBR treated wastewater can be a valuable resource for hydroponic CEA, there are potential challenges that need further studies because of their unknown long-term effects on human health, such as residual chemicals and pharmaceuticals, emerging contaminants, and bioaccumulation of contaminants in crops. For instance, most existing literature on AnMBR has limitations in accurately assessing pathogenic risks due to the reliance on culture-based indicator bacteria. This issue is particularly significant considering the diverse microbial communities within AnMBR systems across regions. To address this, future studies should adopt a comprehensive approach, integrating culture-based, molecular-based, and quantitative microbial risk assessment (QMRA) analyses. This multifaceted strategy will provide a more thorough understanding of the pathogenic and antibiotic resistance tendencies of AnMBR effluent.

Investigating the potential bioaccumulation of emerging contaminants in crops cultivated with AnMBR effluent is also imperative for a holistic health assessment.

AnMBR effluent may have insufficient levels of nutrients essential for optimal plant growth. The AnMBR process can remove some nutrients from wastewater, resulting in effluent with suboptimal nutrient content for plant development. Although supplementing with fertilizers is common to meet crop nutrient demands, future studies should explore optimizing operational parameters and conditions in the AnMBR process or the augmentation of AnMBR feed stream organic loading with plant harvest wastes. Optimizing operational parameters/conditions and incorporating plant harvest waste for the enhancement of high-quality effluent, rich in beneficial plant nutrients, has the potential to decrease the requirement for supplementary fertilization, consequently lowering costs.

AnMBR predominantly produces NH_4^+ as the nitrogen form, which may not be directly beneficial to plants unless converted to the NO_3^- form. In hydroponic CEA, where adequate soil microbial communities for complete nitrification may be lacking, there's a risk of NH_4^+ persistence or partial conversion to NO_2^-, which can be toxic to some crops in high concentrations. Future research should focus on identifying high-value NH_4^+/NO_2^--tolerant crops or developing appropriate technologies to convert the nitrogen form in AnMBR effluent to forms most useful for crop growth.

While the environmental impact of biogas, particularly concerning dissolved methane and headspace methane, has been discussed in previous section, the influence of dissolved methane in AnMBR effluent on crop growth remains inadequately understood. The potential benefits of low concentrations of methane on crops and the potential toxicity or interference with oxygen uptake by plant roots at high concentrations are uncertain. Consequently, further research is essential to gain a more comprehensive understanding of the correlation between dissolved methane in AnMBR effluent and plant growth.

5. Conclusions

In conclusion, the synergistic integration of decentralized wastewater treatment systems with food production operations holds promise for concurrently supplying water, energy, and nutrients, especially with the adoption of innovative technologies like AnMBR and hydroponic CEA systems in decentralized applications, e.g., peri-urban areas. This integration would not only tackle the urgent challenges of wastewater management but also promotes sustainable water use. AnMBR technology ensures efficient and eco-friendly wastewater treatment, while hydroponic systems optimize resource utilization by employing treated wastewater as a nutrient-rich solution for plant cultivation. Nonetheless, the challenges underscored in this study must be addressed to propel the advancement of AnMBR for hydroponic CEA, thereby fostering the development of closed-loop, environmentally conscious systems.

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