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Comparative life cycle assessment of aerobic treatment units and constructed wetlands as onsite wastewater treatment systems in Australia

Giovani Rodolfo Alatrista Gongora, Ray Hangyong Lu and Ali El Hanandeh*

School of Engineering and Built Environment, Griffith University, Nathan, QLD 4111 Australia
*Corresponding author. E-mail: a.elhanandeh@griffith.edu.au

GRAG, 0000-0003-3134-2997; AEH, 0000-0002-1682-0412

ABSTRACT

Life Cycle Assessment was used to evaluate onsite wastewater treatment systems (OWTS): aerobic treatment unit (ATU) with reinforced concrete (C.ATU) and HDPE (H.ATU) tank; and constructed wetland (CW) with three biochar concentrations in the substrate (0%; 10, and 20% v:v), dubbed CW.BC0, CW.BC10 and CW.BC20, respectively. CML 2001 in SimaPro® was used to evaluate the impacts of the treatment of 1 m^3 wastewater. The OWTS were compared on their overall environmental performance scores (OEP). ATUs have higher impacts on human toxicity, eutrophication, freshwater and marine ecotoxicity. The CW.BC20 has the lowest global warming impact (GWP) while CW.BC0 has the highest. Electricity consumption was the largest contributor to the impacts of the ATUs. PVC pipes, coir peat, geomembrane, and electronic devices were the biggest contributors to the impacts of the CWs. The OEP of the CWs were almost a third of the ATUs' (6.07E-03). Changes in electricity sources were tested according to the 2030-Australian targets; increasing renewables share improves the OEP of ATUs by 39%; nevertheless, CWs continue to outperform the ATUs. Variations in biochar biodegradation had a small effect on the OEP of CWs; being relevant only to GWP. This study provides a reference to policy makers for better evaluation of OWTS.

Key words: aerobic treatment unit, domestic wastewater, life cycle assessment, onsite wastewater treatment, vertical subsurface flow constructed wetland

HIGHLIGHTS

- First to present comparative LCA of constructed wetlands vs aerated treatment units for onsite domestic wastewater treatment (OWTS).
- Investigates the consequences of future changes in electricity generation on the environmental efficiency of OWTS.
- Investigates the effect of construction material choice on the overall environmental performance of the systems (OEP).

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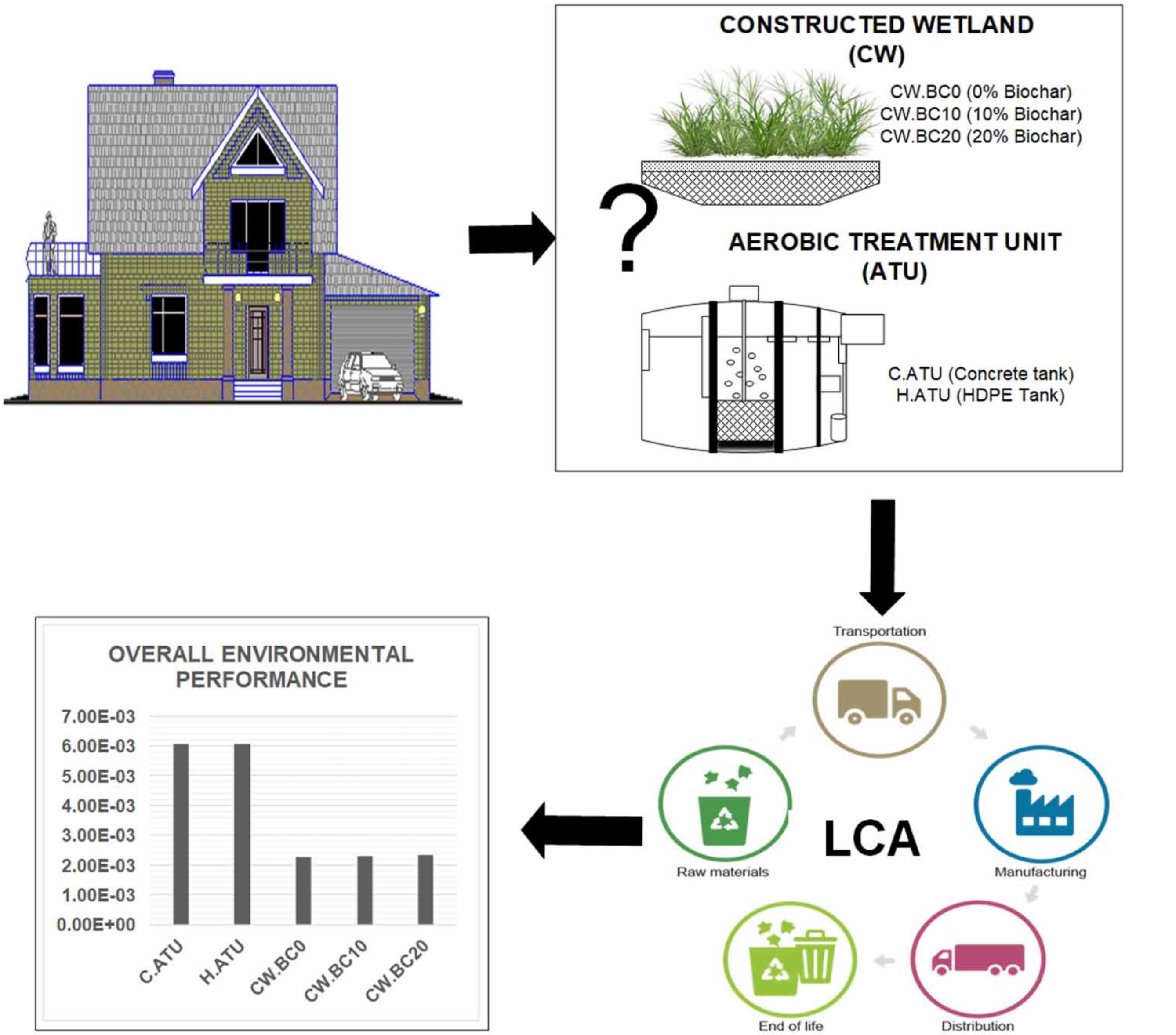
GRAPHICAL ABSTRACT



INTRODUCTION

Wastewater treatment is performed in two types of systems depending on the treatment capacity. Centralized systems are also referred to as conventional wastewater treatment plants (WWTPs), and are usually used in high population density areas where sewerage system is present. Meanwhile decentralized systems or the onsite wastewater treatment systems (OWTSs), are normally applied in low population density areas where a sewerage system does not exist (Leigh & Lee 2019; Ergas *et al.* 2021).

Septic tanks, composting toilets, and the aerobic treatment units (ATUs) are among the most commercialized OWTSs in developed countries (Dubois & Boutin 2018). Particularly, the ATUs are the most popular technologies in Australia. ATUs are technologies that have similarities to centralized systems that use activated sludge or biofilm bed reactors; and have acceptable removal efficiencies according to regulations provided by the Department of Building Property and Development of Queensland, Australia (Government of Queensland 2021). Conversely, nature-based alternatives such as constructed

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wetlands (CWs), are often considered as lower efficiency OWTSs than the ATUs (Magar 2016). Consequently, the CWs are in most cases categorized as a 'polishing' processes associated with septic anaerobic systems; hence these units are used as the last step prior to the release of treated wastewater to the environment (Rodriguez-Dominguez *et al*. 2020; Sylla 2020; Koottatep *et al*. 2021).

Due to the commonly held perception that CWs are likely to produce lower quality discharge than the more advanced technologies, studies comparing different OWTS often assign higher environmental impacts to nature-based systems (Garfí *et al*. 2017; Yang *et al*. 2018). Meanwhile, the ATUs have a contrary perception due its more advanced process for treatment. One example of these criteria for selection of technologies can be seen in Queensland, Australia, where regulatory bodies focus on the effectiveness of the OWTS such as the ATUs, in removing key pollutants from domestic wastewater. However, this selection criteria does not consider the overall environmental impact of the employed technology in any of the reference documents for approval and commercialization (Government of Queensland 2021).

With the purpose of achieving a better comprehensive understanding of the environmental implications of different wastewater treatment technologies, Life Cycle Assessment (LCA) can be used as a tool to provide useful and essential guidance for identifying the potential environmental implications of selecting one or another OWTS technology (Garfí *et al*. 2017). Previous LCA studies have shown that some advanced technologies for wastewater treatment may be outperformed in their environmental performance by conventional or low technological alternatives, despite the improved treated wastewater quality to be discharged (Yıldırım & Topkaya 2012; Corominas *et al*. 2013; De Feo & Ferrara 2017; Diaz-Elsayed *et al*. 2017; Garfí *et al*. 2017; Magar & Magar 2017).

Currently, there is a lack of LCA studies which compare the environmental performance of nature-based systems such as the CWs to ATUs. Energy usage of ATUs and the construction materials for CWs have been identified as significant contributors to the environmental performance of the system (De Feo & Ferrara 2017). Currently, LCA studies in active systems such as the ATUs have shown important changes in the environmental performance when the source of electricity generation (coal, gas, oil, solar) is altered for one or another specific location, this is due to the permanent and high demand of electricity for its operation (De Feo & Ferrara 2017; Mirra *et al*. 2020). Conversely, amendments of the construction material to better target specific pollutants or to improve the quality of the discharge have resulted in inconsistent impacts on the overall environmental performance of the CW systems (Cao *et al*. 2021; Deng *et al*. 2021; Diaz-Elsayed *et al*. 2017; Yang *et al*. 2018; Shen *et al*. 2020; Zhou *et al*. 2020). The availability of new materials that may be used in the substrate can enhance the removal efficiencies of different pollutants in CWs and shrink the gap in the treated wastewater quality currently existing between CWs and ATUs. Concerning this, the use of biochar, a prefabricated material based on carbon, has shown promising results in improving the efficiency of CWs for domestic wastewater treatment, specifically as a substrate material (De Rozari *et al*. 2015; Deng *et al*. 2021). However, it has not yet been established if the improved pollutant removal efficiency will enhance the overall environmental performance. Particularly, it is not clear if the environmental benefits gained through the improved wastewater quality overcome the burdens associated with the manufacturing and use of biochar. LCAs which examined the wastewater treatment systems focused mainly on centralized WWTPs (Tangsubkul *et al*. 2005; Foley *et al*. 2010a, 2010b). Consequently, there is little information about the environmental implication of the use of biochar and how its use in CWs would compare to systems with high removal capacity such as the commercialized ATUs which limits the ability of decision making about the selection of better OWTSs from an environmental perspective. Therefore, the present study aims to evaluate the performance of CWs and ATUs from an LCA perspective in the Australian context and provide an objective assessment of their environmental impacts.

METHODOLOGY

The LCA was conducted using SimaPro v.8.0.4.30 modelling software, based on the International organization for standardization ISO 14040:2006/AMD 1 :2020 (International Organization for Standardization 2020), which includes the following steps:

Scope and goal definition

In the present study, the diagnosis and comparative analysis was performed among the ATUs and the CWs as OWTS alternatives, having two construction variations for the ATUs (concrete and HDPE), and three for the CWs with different biochar contents in the substrate. The ATUs constructed with reinforced concrete as the main material for the structure are referred to C.ATU. ATUs constructed with HDPE as the main material for the structure are referred to as P.ATU. The three

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configurations of CW are: CW.BC0, corresponding to CWs with 0% v:v content of biochar in the substrate; CW.BC10: corresponding to CWs with 10% v:v content of biochar in the substrate; and CW.BC20: corresponding to CWs with 20% v:v content of biochar in the substrate.

The scope includes the inputs and outputs from 'cradle to grave'. The system boundaries are shown in Figure 1. Additionally, the functional unit was set as all the input/outputs associated with 1 m^3 of treated wastewater, considering a lifetime of the treatment systems of 20 years.

Life cycle inventory

For this section, the design and inventory data for both technologies, ATUs and CWs, and their construction types are detailed in the Appendix A (Table A1–A5).

The grouping of the inventory data for both technologies was separated in three subgroups: Construction and installation; operation and supervision; and waste management after 20 years of use (Technology wastes).

Treated wastewater was the only 'product' of concern being used for the diagnosis and comparison between the two technologies. Therefore, allocation procedure was not required due to the lack of co-functionality of the system (Lopsik 2013; Sabeen *et al.* 2018).

Aerobic treatment units

The ATU inventory detailed in Appendix A (Tables A3–A5) was obtained from the analysis of approved certificates for companies installing OWTPs in Queensland, Australia. This included the ATUs as the core treatment unit sized for 10 people equivalent (P.E), also referred to as a 2 m^3/day treatment capacity of domestic wastewater. In Figure 2, the scheme of the processes for the ATUs is presented.

As shown in Figure 2, the core unit of the ATU had four chambers: primary, aeration, settling, and disinfection/pump chamber. After the wastewater is treated, it is sent to the absorption trenches unit for final filtration prior the discharge to the environment, as is required by the local regulations (Government of Queensland 2021). For this purpose, the absorption trenches were sized according to the sizing procedure based on the Australian standards for domestic treatment systems (AS/NZS.1546.1 2008), where the final quality of treated wastewater from the core unit was a determining factor for sizing the filter.

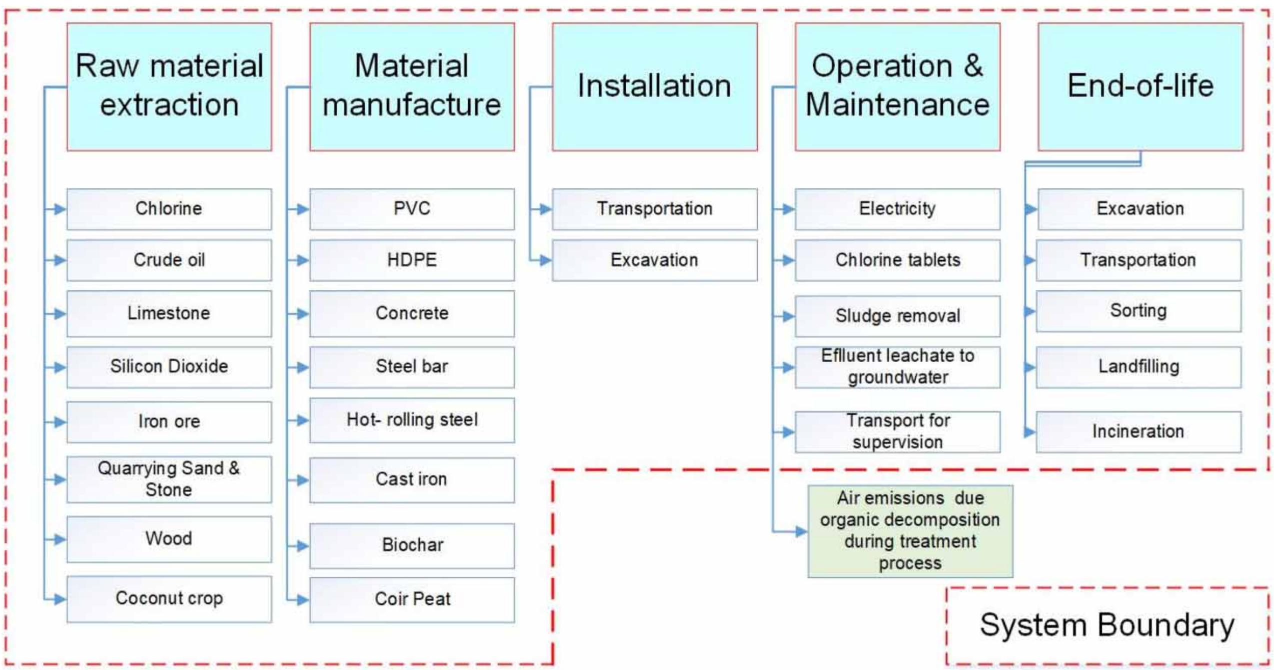

Figure 1 | Boundaries of ATU and CW systems.

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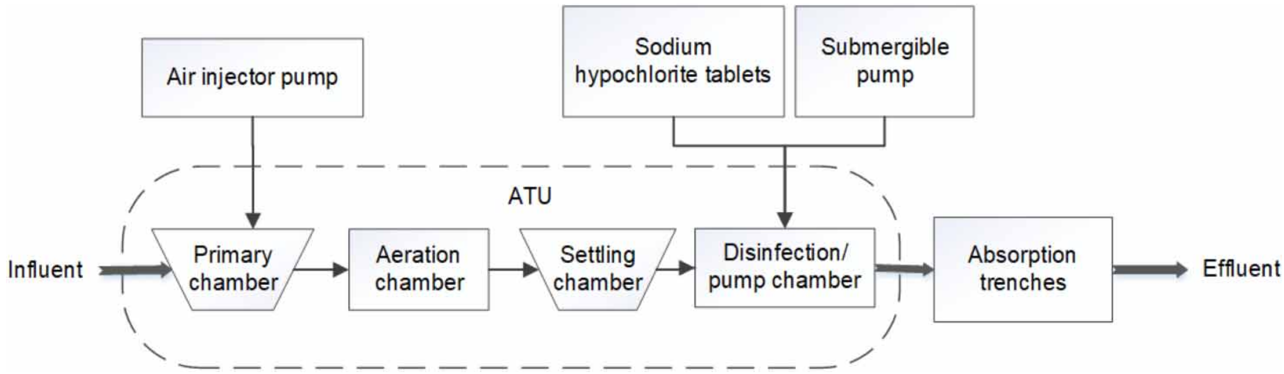

Figure 2 | Schematic of ATU systems.

Constructed wetlands

The CWs inventory data, shown in Appendix A (Tables A3–A5), were set according to designs from pilot scale experiments performed in Queensland, Australia (De Rozari *et al.* 2015, 2018). In brief, the CW system consists of a vertical subsurface flux constructed wetland core treatment unit, with a pre-treatment process performed in a settling/pump chamber. Both process units, as in the design of the ATUs, were sized for 10 P.E.

Once treated, the wastewater was also sent to an absorption trenches unit (Figure 3). The absorption trenches were sized according to the final quality of wastewater following the Australian standards for domestic treatment systems AS/AZN1546.1.

Assumptions

For complementing the LCI for both technologies, 10 assumptions were made as shown in Table 1.

Life cycle impact assessment (LCIA)

LCIA was performed to evaluate the significance of the environmental impacts calculated from the LCI. From the published literature, the experiences across different regions of the world in LCAs for centralized WWTPs and OWTSs show evident variation in the models used for its calculation. In wastewater treatment, the most used LCIA methods in order of higher to lower frequency are the CML 2,001 baseline, Eco-Indicator 99, Eco-Points 97, EDIP, EPS2000, ReCiPe, Impact 2000, and the e-balance method (Corominas *et al.* 2013; Renouf *et al.* 2015; Sabeen *et al.* 2018).

For the present study, the CML 2001 method was selected due to the higher number of experiences and recommendations from LCA guidelines specifically for Australia. CML 2001 includes the following impact categories: Eutrophication potential (EP); Terrestrial Eco toxicity (TETP); Marine eco toxicity (METP); Freshwater eco toxicity (FAETP); Human toxicity (HTP); Acidification potential (AP); Photochemical oxidation (PO); Ozone depletion potential (ODP); Global warming potential (GWP); Fuel depletion potential (FDP); as well as Abiotic depletion potential (ADP) (Corominas *et al.* 2013; Renouf *et al.* 2015).

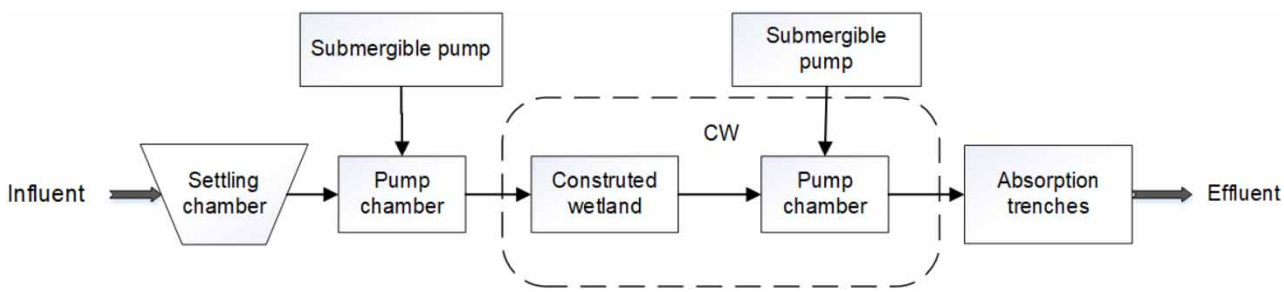

Figure 3 | Schematic of the CW systems.

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Table 1 | Assumptions of LCI for ATUs and CWs used as OWTS in Australia

Assumptions
Soil removed for installation and closure stage was used nearby (25 km maximum distance), not being sent to landfill.
Energy consumption during construction work was not considered.
Transport of materials for construction was assumed as 50 km.
For ATUs and CWs, the replacement of electronic devices was assumed after 10 years of operation.
For the CW.BC0, CW.BC10 and CW.BC20, a total replacement of substrate materials was considered after 10 years of operation.
For CWs, biochar used as substrate material had a 50% sequestration rate of carbon, which is counted as a credit. In other words, only 50% of the carbon contained in the biochar are expected to be released in the form of CO_2 due to the biodegradation of biodegradable carbon over a 100 years period.
Variation in energy demand during the 20 years of operation was assumed as 0%.
The electricity generation source, considered for operation for AWTSs and CWs, was based on Australian Government reports (Australian Government 2019).
Supervision procedures were estimated as the maximum recommended by certification and guidelines (4 times per year for AWTS; two times per year for CWs), and the distance of transport for each supervision visit was assumed to be 20 km.
Transport of the technology wastes from AWTS and CWs to landfill at the end of the lifetime service of 20 years, was considered as 25 km.

Comparative assessment

To compare the OWTS alternatives, the overall environmental performance (OEP) was calculated following the process detailed in LCA experiences from Australia (Lu & El Hanandeh 2019), where one stage normalization was performed (Appendix B).

The procedure for calculating the OEP using the characterization score of each of the 11 impact categories was performed by normalizing the impacts to the Australian context. Each of the category scores was divided by the Australian per capita values for the corresponding impact category (Eq.B1) (Bengtsson & Howard 2010; Lu & El Hanandeh 2019). Normalization results are shown in Table B.1. Then, the 11 normalized scores were summed to obtain a unique score, known as the OEP; lower values of OEP indicate better performance.

LCIA RESULTS

Overall environmental performance

In Table 2 the LCIA characterization results and the OEP for the ATUs and CWs are presented. The ATU systems show a higher OEP score than CW systems. Differences in the OEP score among the three CW systems were observed, being higher as the content of biochar increases in the substrate from 10% to 20%. In this case, the lowest OEP score corresponds to the CW with no biochar in its substrate (CW.BC0).

Contribution analysis

Global warming potential

Table 2 shows that the GWP impact associated with the CWs, specifically the configurations that include biochar as complement of substrate (CW.BC10 and CW. BC20), is lower than the ATUs.

The contribution analysis (Figure 4(a)) revealed that the main contributor to the GWP impact category is the technology wastes generated at the end of life of the CWs after 20 years of operation. The disposal and treatment of spent substrate materials, such as sand and gravel used in high mass quantity, is a particular process contributing to the GWP because of the energy consumption. In contrast, Figure 4(a) also shows that the inclusion of 20% of Biochar in the substrate (CW.BC20) has a negative value of GWP during the construction of CWs. The reason for this is the carbon content in biochar that is stored in the soil, which in turn offsets the net greenhouse gas emissions of the technology.

For the ATUs, the highest contribution to GWP comes from the emissions of CO_2-eq generated by the operation of the systems. Precisely, this is the permanent requirement of electricity for powering pumps and blowers for the treatment process. This is consistent with previous studies which pointed out energy consumption during the operation phase as being a hot spot (De Feo & Ferrara 2017; Mirra *et al.* 2020). This is especially so in Australia because the generation of electricity in Australia

Table 2 | LCIA by CML 2001 method for ATUs and CWs

Impact category group	Impact category	Unit	P.ATU	C.ATU	CW.BC0	CW.BC10	CW.BC20
Climate change	Global warming (GWP100a)	kg CO_2 eq	1.358E+00	1.390E+00	1.459E+00	1.067E+00	6.281E−01
Human toxicity	Human toxicity	kg 1,4-DB eq	8.982E−01	8.933E−01	6.565E−01	6.614E−01	6.674E−01
Ozone layer depletion	Ozone layer depletion (ODP)	kg CFC-11 eq	2.834E−08	3.079E−08	4.243E−08	4.481E−08	4.769E−08
Eutrophication	Eutrophication	kg PO_4^- eq	1.282E−02	1.279E−02	1.173E−02	1.184E−02	1.197E−02
Ecotoxicity	Fresh water aquatic ecotoxicity	kg 1,4-DB eq	8.048E−01	8.059E−01	1.990E−01	2.043E−01	2.104E−01
	Marine aquatic ecotoxicity	kg 1,4-DB eq	2.660E+03	2.671E+03	3.325E+02	3.382E+02	3.450E+02
	Terrestrial ecotoxicity	kg 1,4-DB eq	1.774E−03	1.923E−03	2.081E−03	2.150E−03	2.230E−03
Depletion of abiotic resources	Abiotic depletion (fossil fuels)	MJ	1.970E+01	1.832E+01	1.809E+01	1.910E+01	2.033E+01
	Abiotic depletion	Kg Sb eq	1.755E−06	1.852E−06	2.989E−06	3.034E−06	3.090E−06
Acidification	Acidification	kg SO_2 eq	6.636E−03	6.752E−03	9.899E−03	1.021E−02	1.058E−02
Photochemical oxidation	Photochemical oxidation	kg C_2H_4 eq	2.764E−04	2.767E−04	3.067E−04	3.667E−04	4.360E−04
OEP			6.07E−03	6.07E−03	2.26E−03	2.29E−03	2.33E−03

is highly dependent on fossil fuels (approximately 88% of fossil fuels: 68% coal and 20% natural gas) (Australian Government 2019).

Human toxicity

From Table 2, the two construction types of the ATUs show higher HTP score compared to that obtained from the three types of CWs. Moreover, in Figure 4(b), the operation of the ATUs contributed more impact on HTP, mainly because of the permanent demand of electricity. Conversely, the construction phase is the main contributor to HTP in the case of CW, which is mainly related to material production.

As mentioned in the GWP analysis, coal and natural gas are used in higher proportions for the generation of electricity in Australia. According to the Toxic Release Inventory (TRI), the specific use of coal involves emissions of three principal categories englobing nonvolatile and volatile highly human toxic substances such as, mercury, selenium, beryllium and barium (Rubin 1999; Färe *et al*. 2010; Ozturk & Dincer 2019). In the case of the CWs, this technology showed a higher contribution to HTP in its construction phase, due to the relatively high requirement of materials such as PVC pipes, which involves emissions of carcinogens such as vinyl chloride and dioxins during its manufacture (EPA 2021).

Ozone layer depletion

The CWs (CW.BC0; CW. BC10; CW. BC20) have higher ODP score than the ATUs as shown in Table 2. Figure 4(c) shows that the construction phase of CWs represents the highest contribution for the ODP; further analysis reveals that the transport of construction materials such as sand, gravel, coir peat, concrete, and others, was the input with the higher contribution for this impact category. This result implies that the emissions from the transport of large quantities of materials requires combustion of fossil fuels with the specific release of nitrous oxide, pointed out as one of the gases with evidenced adverse effects for the ozone layer (Chmielewski 1999).

ATUs are much lighter systems and require only the transport of approximately two-thirds of the materials compared to CWs for the construction of the system. Therefore, they result in lower emissions due to transportation and in turn lower ODP impact. However, the electricity used for the operation of this technology represents its highest contribution to the ODP impact of the ATUs. Of particular concern is the use of natural gas for power generation, comprehending the emissions of gases such as chlorofluorocarbons (CFC), halons, and hydrochlorofluorocarbon (HCFC), used as fire suppressants and coolants in the gas pipeline distribution system, being substances with adverse effects for the ozone layer (Purohit *et al.* 2020).

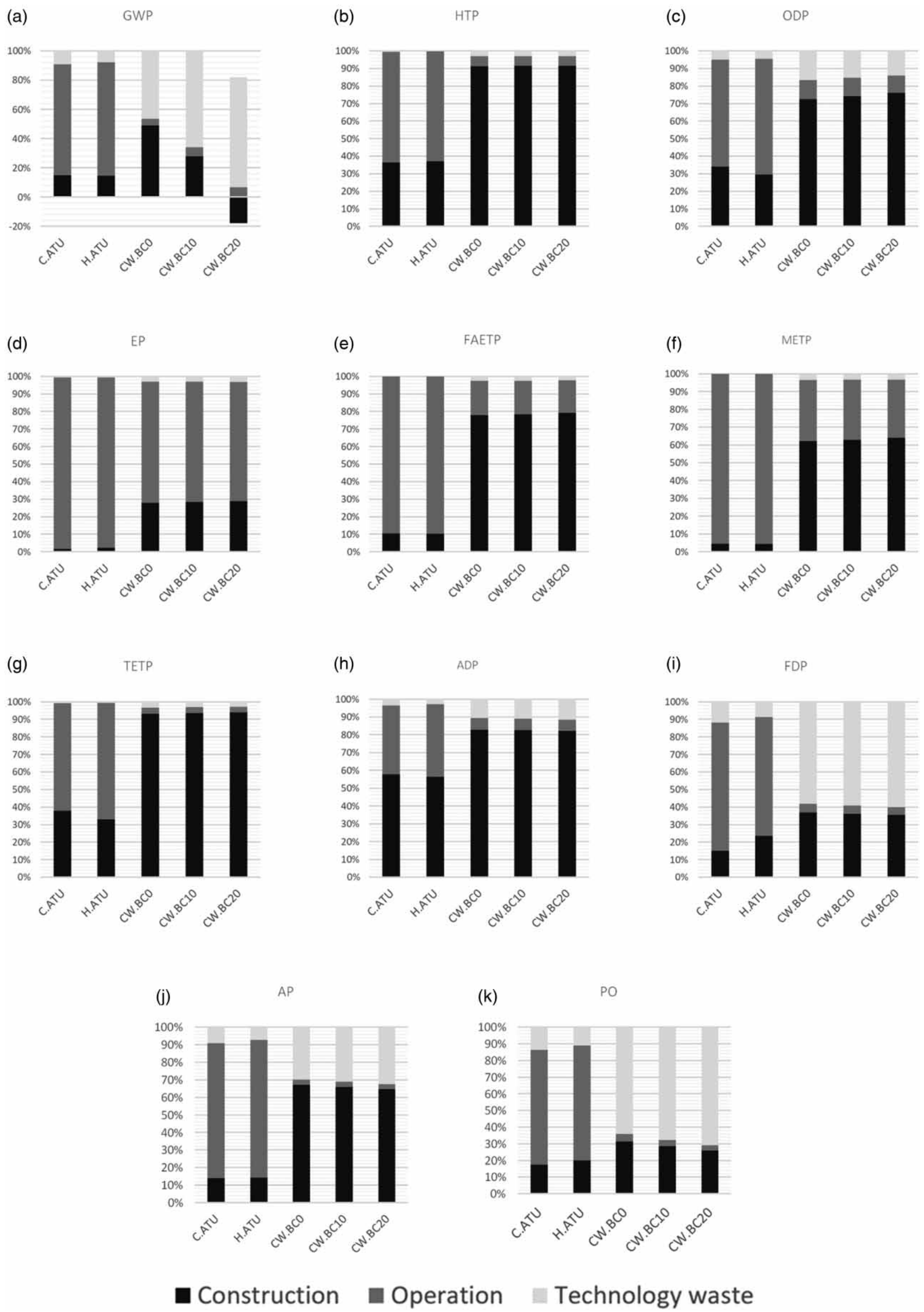

Figure 4 | The contribution of each phase (construction, operation and technology waste) to the impact categories: GWP (a); HTP (b); ODP (c); EP (d); FAETP (e); METP (f); TETP (g); ADP (h); FDP (i); AP (j); and PO (k).

Eutrophication

The EP impact is slightly higher for the ATUs than the CWs, as shown in Table 2. From Figure 4(d), the operation is the main phase that contributes to the eutrophication impact category for ATUs and CWs (approximately 95%). The higher impact found in ATUs is not due to discharge of treated wastewater and its quality, being the same for both technologies and its configurations. In this case, the difference is set as a consequence of NO_2 emission from the use of fossil fuels such as coal and natural gas for the generation of electricity (Zhengfu *et al.* 2010; Korpela *et al.* 2017; Purohit *et al.* 2020). In this context, the emissions from the ATUs are higher due to the continuous demand of electricity.

Ecotoxicity

The ATUs have a higher toxicity impact to aquatic ecosystems (FAETP and METP) than the CWs, as shown in Table 2. Figure 4(e) and 4(f) shows that, in the case of the ATUs, approximately 90% of the ecotoxicity impact can be attributed to the operation phase. The principal cause can be traced to the emissions from mining processes for coal used in power plants, which involves direct and indirect emissions of heavy metals and acid drainage to water bodies which have toxic effects on aquatic ecosystems (Zhengfu *et al.* 2010). Meanwhile, the main contribution for the FAETP and METP in the case of the CWs is the construction phase, with PVC pipes and the pump devices being the main contributors for both impact categories. The emissions from the production of PVC pipes imply fossil fuels as prime matter, having the background emissions from the mining and refinement processes (Saeki & Emura 2002). Furthermore, the emissions related to the mining and refinement of metals, and its further transformation to components of the pump device, includes emissions of substances with heavy metal content reaching water bodies by direct and indirect pathways (Hu *et al.* 2014).

For the TETP (Table 2), the CW technology has higher impact scores compared to the ATUs. Particularly, due to coir peat used as a substrate component in the CW.BC0; CW.BC10; and CW.BC20 (Figure 4(g)). Despite being a 'waste' product made from the coconut crops, coir peat processing for substrate applications involves the use of large quantities of water and chemicals for removal of substances such as salts, tannins and phenolic compounds (Areapeat 2020). In consequence, it may generate a bulk of liquid solutions with variable toxicity that reaches the soil. Furthermore, this material may be transported long distances to the target location, comprising emissions related to fossil fuel utilization (Stucki *et al.* 2019).

In the case of the ATUs, the operation is the phase with the highest contribution to the TETP (Figure 4(g)). Further analysis of the two ecotoxicity impact categories in ATUs and CWs point to electricity consumption as a main contributor due to heavy metal release (mercury, vanadium, cadmium, etc.) during the coal extraction and its subsequent use for power generation (Färe *et al.* 2010; Zhengfu *et al.* 2010).

Depletion of abiotic resources

From Table 2, the three CWs variants show higher ADP impact than the ATUs. The construction phase had the highest contribution to ADP. Figure 4(h) shows the construction phase of the CWs as being the phase with the highest requirement of abiotic resources. The mineral resources to produce pumping devices, followed by the materials used for the fabrication of the geomembrane are identified as inputs with highest contribution to ADP in the case of CWs. The construction phase was also identified as the main contributor to ADP in the case of ATUs, Figure 4(h). Drilling down into the results revealed that pumps and blowers are the highest contributors to ADP in the case of ATUs.

Fossil fuel depletion

The CWs with all variants have a higher FDP impact than ATUs, as shown in Table 2. The FDP worsen with the higher inclusion of BC. Further inspection of the results revealed that the main contributor to this surprising outcome is the waste treatment at the end-of-life stage followed by the construction phase (Figure 4(i)).

Disposal of PVC pipes and geomembrane, as well as other inert wastes require treatment prior to landfilling. For example, spent biochar (after being used) may be considered as hazardous waste due to potential biological contamination and inorganic pollutant loads (such as heavy metals adsorbed by biochar), thus requiring special treatment. Furthermore, transformation of large quantities of waste material where the use of fuels is crucial for powering the processes is a significant contributor. Conversely, the operation phase is the main contributor to FDP in the case of ATUs, due to the permanent requirement of electricity.

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Acidification

CWs has higher AP impact than ATUs as shown in Table 2. Upon inspection of Figure 4(j), the construction phase of the CWs, specifically the production of coir peat, and the PVC pipes for the core treatment unit, causes direct and indirect release of SO_2 and NO_x, agents with a strong acidification effect on the environment. In case of the ATUs, the operation phase is the main contributor to AP with electricity consumption being the main culprit.

Photochemical oxidation

Compared with the impacts caused by the ATUs, the CWs showed higher PO (Table 2), being the scores attributed to the emissions associated with the technology wastes at the end of its lifetime service of the CWs (Figure 4(k)). Materials such as the PVC and geomembrane, manufactured from fossil fuels, may have important relevance together with the solid waste treatment process fed with fossil fuels, generating along with the primary air pollution, a secondary type of pollution, influenced by the sunlight reaction over gases such as nitrogen oxides and volatile organic compounds, generating other pollutants.

For the technology of the ATUs, as in most of the impact categories of the CML 2001 method, the operation remains as the main contributor to the PO (Figure 4(k)), due to the emissions related to the fossil fuels combustion for power generation in Australia.

DATA QUALITY ANALYSIS

Electricity scenarios

The higher contribution in the LCIA for both construction types of ATUs (C:ATU, H.ATU), comes from its operation; specifically, electricity consumption for water pumping and air injection, which contributes at least 50% for most of the impact categories with the exception of the ADP, ODP, and EP. Conversely, the LCIA of the three variants of the CWs (CW.BC0; CW.BC10; CW.BC20), had no evident dependence on the operation of its electronic devices, instead the materials used for the construction weighed more heavily.

As pointed in the contribution analysis for the 11 impact categories, the current generation of electricity in Australia is highly dependent on fossil fuels (Australian Government 2019). The current scenario dubbed 2019 AU, is shown in Table 3 together with two scenarios constructed in line with the objectives for energy supply for Australia; where national policies stipulate that by 2030 the use of renewable energy should increase to 50% of the total supply (Karp 2019).

In detail, the 2019 AU energy scenario represents the business-as-usual case, where sources of renewables account for an approximated 12% (Australian Government 2019). This scenario has already been analyzed and the results shown in the previous comparative OEP section. In this section, the impact of achieving the national goal of 50% renewable energy is investigated. Two pathways to achieve the target are considered: the case of the 2030 AU (50%) scenario assumes an increase of the renewable share to 50% with equal distributed increment among all the renewable sources (hydro energy, ethanol from wood, photovoltaic); and the 2030 S.AU energy scenario, assumes the increase of renewable energy to 50% through increasing the share of photovoltaic electricity.

To evaluate the influence of energy sources on the environmental performance of the ATUs and CWs and compare it with current scenario (2019 AU); the OEP of each OWTS alternative was calculated for the 2030 AU and 2030 S.AU scenarios; the results are presented in Table 4.

Increasing the share of renewables in the electricity grid reduces the environmental impacts of the CWs and the ATU systems. However, it has higher effect on the ATUs. The 2030 AU pathway is likely to return greater environmental benefits than the 2030 S.AU as evident from the percentage reduction of the OEP in Table 4.

Table 3 | Baseline (2019 AU) and future electricity generation scenarios in Australia

ELECTRICITY INPUT	2019 AU	2030 AU	2030 S.AU
Hydro energy	0.05	0.17	0.05
Ethanol from wood	0.03	0.16	0.03
Natural gas	0.2	0.2	0.2
Hard coal	0.68	0.3	0.3
Photovoltaic	0.04	0.17	0.42

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Table 4 | Sensitivity of OEP impact category to changes in electricity source

Energy scenario	C.ATU	P.ATU	CW.BC0	CW.BC10	CW.BC20
2019 AU OEP	6.07E−03	6.07E−03	2.26E−03	2.29E−03	2.33E−03
2030 AU OEP	3.67E−03	3.67E−03	2.16E−03	2.19E−03	2.23E−03
2030 S.AU OEP	3.90E−03	3.89E−03	2.17E−03	2.20E−03	2.24E−03
2030 AU (%)	−39.56	−39.58	−4.25	−4.18	−4.11
2030 S.AU (%)	−35.80	−35.82	−3.84	−3.79	−3.72

A move towards more renewables share can potentially reduce the ATU overall impact by more than 35%. Meanwhile, the change in the OEP of the CWs is less than 5% (Table 4). Despite the significant decrease in the overall impact of the ATUs, the CWs remain as the technology with better environmental performance.

The energy demand is the principal obstacle limiting the ATUs from achieving better environmental performance compared to the CWs. Moreover, operational costs due to electricity consumption is another factor to consider in the selection of technologies. For the present case, the Australian electricity price rates of electricity to residential customers are around \$0.34/kWh (Globalpetrolprices 2020). The electricity consumed to treat 1 m^3 of wastewater in ATUs is estimated to be 1.053 kWh/m^3 which corresponds to an approximate of 0.358 \$/m^3. This cost is higher than the average prices of energy required for centralized WWTPs (0.46 kWh/m^3) (Gu *et al.* 2017), corresponding to approximately 0.156 \$/m^3. Conversely, the electricity requirement for the operation of CWs is less than half of that required by the ATUs (0.42 kWh/m^3) which translates to 0.143 \$/m^3.

Biochar

The percentage of biochar used in the substrate was the principal variant among the three CWs. Due to the high content of carbon in the biochar, its use had a positive influence on balancing the CO_2 emissions from the construction stage. The CW.BC10 and CW.BC20 had lower scores and even negative values (net sequestration) to the GWP when compared with the CWs built with conventional substrate elements (CW.BC0). However, the biochar also influenced adversely and at different magnitude the HTP, ODP, EP, FAETP, METP, TETP, ADP, FDP, AP, and PO impact categories. Most of those adverse effects were identified in the construction and technology waste stages; as a result a slight increase in the OEP scores of the CW.BC10 and CW.BC20 over CWBC0 was noticed as shown in Table 2.

Variation in the production of biochar detailed in Table 5 were assumed to be negligible. However, the fixed carbon in the biochar and its further release in the form of CO_2 to the atmosphere can influence the GWP and the OEP of CWs alternatives; principally, due to the differences in the environmental exposure of the material along its life cycle. Furthermore, as pointed in the LCI assumptions in Table 1, 50% of the carbon in the biochar would be released in the form of CO_2 within 100 years. This means that half of the fixed carbon in biochar remains retained in the soil after its use and disposal of the material. In this context, the sensitivity of the LCIA results in variations in biochar carbon stability was tested for the cases of 100% (all carbon remains locked in the biochar) and 30% assuming 70% of the carbon is released as CO_2 within 100 years.

Table 5 | Biochar production and other wood derivatives (Lu & El Hanandeh 2019)

Wood source	Hardwood plantation from South-east Queensland	
Pyrolysis temperature (°C)		500
Product	Gas yield %	20.9
	Biochar %	22.03
	Bio-oil %	64.2
CO_2 offset	Biochar (kg CO_2 offset)	601
Allocated energy	Bio-oil (kWh)	3849.9
	Gas (kWh)	446.4
Pyrolysis feed	Enthalpy kWh per F.U	236.3

Table 6 | Sensitivity of OEP and GWP impacts to biochar carbon sequestration rate

CW	Decay %	GWP ($kgCO_2$-eq)	OEP
BC0	–	1.46E+00	2.26E−03
BC10	0%	8.92E−01	2.28E−03
BC10	50%	1.07E+00	2.29E−03
BC10	70%	1.14E+00	2.29E−03
BC20	0%	2.54E−01	2.32E−03
BC20	50%	6.28E−01	2.33E−03
BC20	70%	7.77E−01	2.34E−03

As shown in Table 6, the reduction of biochar's carbon decay to 0% resulted in the lowest scores for the GWP for the CW.BC10 and CW.BC20. An increment to 70%, had the contrary effect, increasing the GWP. Despite this potential increase, the two CWs with biochar have lower GWP and OEP scores than the CW. BC0. Changes in carbon decay rates do not seem to affect the ranking of the alternative and CWs continue to return better OEP scores than the ATUs.

CONCLUSIONS

The CWs in any of the configurations of the study have better overall environmental performance than the ATUs. The use of a reinforced concrete tank vs HDPE tank made little difference to the overall environmental performance of the ATU system. The operation phase was identified as the main contributor to the environmental impacts of the ATU systems; from which the electricity required for the aeration process was the principal contributor for all the impact categories, except for ADP, whose principal contributors was the production of electronic devices (blower and pumps), and the EP, depending on the most in the discharge of treated wastewater.

For the CWs, the construction stage has the highest adverse contribution to its environmental performance; in particular, the PVC pipes, coir peat, geomembrane, and the electronic devices were the inputs with the highest contribution to all impact categories with exception of the EP, which was mainly associated with the discharge of treated wastewater during the operation of the technology.

The data quality analysis revealed that the anticipated increase of renewable energy share in the Australian electric grid will improve the environmental performance of the OWTS and particularly the ATUs. Nevertheless, the ATU technology will continue to deliver lower environmental performance than the CWs.

The carbon sequestration rate of the biochar (varied between 0 and 70%) will have very negligible impact on the overall performance of the CW with its effect only felt on the GWP impact category.

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CONFLICT OF 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 STATEMENT

All relevant data are included in the paper or its Supplementary Information.

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