Journal of Cleaner Production 472 (2024) 143496

Contents lists available at ScienceDirect

Journal of Cleaner Production

journal homepage: www.elsevier.com/locate/jclepro

Comprehensive life cycle assessment of garden organic waste valorisation: A case study in regional Australia

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ARTICLE INFO

Handling Editor: Panos Seferlis

Keywords:
Climate change
Biochar
Waste management
Pyrolysis
Life cycle assessment

ABSTRACT

This study evaluates the environmental impacts associated with the conversion of garden organic waste into value-added products, namely compost or biochar, employing various processes. Three distinct scenarios are considered: composting garden organic waste followed by screening of oversized materials (CBP), pyrolysis of oversized screenings of compost into biochar AP(I), and in-situ conversion of garden organics into biochar AP(II). A comprehensive Life Cycle Assessment (LCA) was conducted using OpenLCA software and life cycle impact assessment was conducted using Recipe 2016 midpoint methodology. The environmental ramifications of each scenario were assessed, optimising transport distances in AP(II) to achieve a functional unit of one tonne of biochar produced within a cradle-to-gate system boundary. For the first time, this study offers a holistic exploration of the benefits of soil biochar application, extending its scope to climate change mitigation, incorporating the optimisation of transport distance and its influence when scaling up the technology. The results revealed that global warming was increased from 125 kgCO_2 eq during composting of garden waste to 232 kgCO_2 eq where oversized screenings of compost is converted to biochar at an off-site facility. However, direct conversion of the oversized organic waste to biochar, without composting, showed reduced global warming impact of 56 kgCO_2 eq, and is thus the most favourable scenario to limit climate impacts of this fraction of organic garden waste. However, among 18 environmental impact indicators studied, eight indicators were either not influenced or did not significantly increase by transport distance to an off-site pyrolysis facility, while the magnitude of 10 impact indicators increased with transport distance. The insights and methodologies presented in this study hold global relevance, based on an actual case study in regional Australia, offering valuable recommendations for sustainable waste management practices and establishing biochar as a carbon-neutral or carbon-negative solution. The findings contribute to existing waste management knowledge and provide guidance for accessible carbon dioxide removal and soil carbon sequestration technologies.

1. Introduction

Around 1.4 billion tonnes of organic waste is generated globally (Kaza et al., 2018) and of this, 5 million tonnes is generated in Australia per year (DELWP, 2023). Better management of this waste is required to reduce the amount of organic waste going to the landfill. Composting of garden organic waste is commonly used for valorisation of this waste in an easy and cost-effective method (Kumar et al., 2011). Compost can be used as fertilizer, and soil conditioner by adding stable carbon to soil. However, not all garden waste is suitable for composting, particularly the oversized screenings (OS) of the composting process. Oversized screenings are particles >20 mm in size, which has been through all the phases of composting but cannot be sold as a compost (López et al., 2010) and is screened out and discarded as waste. For example, of 350 tonnes of garden waste collected by City of Greater Geelong, Australia, per week, ~300 tonnes of dry waste are recovered, of which ~150 tonnes is processed to marketable compost, with another ~150 tonnes of OS that remain as waste. Alternative management of the OS wastes are vital given to maximise valorisation of garden wastes and reduce decomposition and global warming potential of emissions.

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https://doi.org/10.1016/j.jclepro.2024.143496
Received 30 March 2024; Received in revised form 8 July 2024; Accepted 25 August 2024
Available online 28 August 2024

Compost is generally produced using open-air windrows (processing of garden organic waste in an open-air environment, in the form of piles where the material can break down in the presence of oxygen) on a large area of land (Serafini et al., 2023), producing leachate, and composting gases (Kumar et al., 2011). The control, capture, and treatment of the emissions from decomposition of waste materials is difficult. More than 97% of the emission are released to the atmosphere (Steiner et al., 2014) and approximately half of the carbon pollutants present in the leachate and sludge during composting process are released in the environment through water and soil (Wen et al., 2019). In terms of emissions contributing to climate change, it is reported that the composting of garden waste released emissions equivalent to 67 kg CO_2 eq (Lu et al., 2020), 130 kg CO_2 eq (Oviedo-Ocaña et al., 2023) to 150 kg CO_2 eq (ten Hoeve et al., 2019) for every 1000 kg of garden waste composted. Hence there is a requirement to understand and control emissions related to composting (Wen et al., 2019).

Recently biochar has attracted great interest in the waste management sector (James et al., 2022; Patel and Panwar, 2023). Biochar is made from the pyrolysis of waste biomass and is used for multiple co-benefits to water, soil, air, and environment (Kumar Mishra et al., 2023; Xia et al., 2024). It is a value-added product made from thermal treatment of waste that improves the soil quality and is also considered as a negative emission technology (IPCC, 2018; Shoudho et al., 2024). The life cycle benefits, and environmental impacts of biochar production process depends on the feedstock as well as the scope and complexity of the technology used. However, generally biochar has high pH, carbon content, cation exchange capacity CEC (Adhikari et al., 2023b,c), and nutrient availability with high surface area and porosity (Adhikari et al., 2023). Biochar produced from waste plant biomass exhibited pH of 7–11 (Ji et al., 2022), carbon content of 40–75% (Mao et al., 2019), CEC of 1.3–10.8 $cmol_c kg^{-1}$ (Domingues et al., 2020) and surface area of 1–440 m^2/g (Mao et al., 2019). Additionally, biochar reduces the emission of atmospheric CO_2 and provides opportunities for carbon sequestration (Joseph et al., 2021). Converting the OS of composting process to biochar is an alternative end-of-life management of the waste. This will provide a carbon negative and sustainable pathway of waste management, soil amendment and reduce global warming.

Life cycle assessment (LCA) is a technique to analyse and explore the potential environmental impacts (positive or negative) of any product (Patel and Panwar, 2023). Recent studies have used LCA to analyse the carbon reduction potential of agro-residues (Dai et al., 2020; Tisserant et al., 2022; Zhu et al., 2022), using different pyrolysis processes, and indicated the suitability of agro-residue biomass for sustainable production of biochar. Application of biochar made from organic waste derived from Norwegian barley crops showed a significant net negative emissions (2–8 tonnes CO_2 eq ha^{-1} yr^{-1}) (Tisserant et al., 2022), depending on the use scenarios for pyrolysis products. Biochar used as fertilizer, with bio-oil sequestration provided maximum sequestration (~8 tonnes CO_2 eq ha^{-1} yr^{-1}), followed by biochar used as fertilizer with combined heat and power (~5 tonnes CO_2 eq ha^{-1} yr^{-1}). Soil only application of biochar provided least sequestration (~2 tonnes CO_2 eq ha^{-1} yr^{-1}). In all the scenarios the emissions from the process were balanced by the negative emissions, indicating a net negative emission. Therefore, biochar was demonstrated as a potential negative emission technology depending on the feedstock type, production process and end use. More than 50% of biomass carbon that would be released to the atmosphere if the waste biomass was left to decompose is converted into stable carbon that is locked in biochar for more than 100 years (Joseph et al., 2021; Steiner et al., 2014). Emissions are thus reduced, removed, and avoided by the sourcing of feedstock, self-sustained production process, heat and electricity generated from the process, sequestration of bio oil (Tisserant et al., 2022) and use of biochar in soil (Xia et al., 2024). After a review of more than 200 articles on open burning and pyrolysis of organic waste, Patel and Panwar (2023) concluded that quantifying the benefits and limitations of soil application of biochar is yet to fully account for complex processes. Although numerous studies have delved into the LCA of biochar systems, utilizing diverse feedstocks, with a focus of biochar soil application, there are only few studies using garden organics or crop/composting residues. A net negative emission of ~920 $kgCO_2$eq per tonne of biochar from crop residue was observed by (Yang et al., 2021), however other studies with different methodological approach and integration of more damage categories is required the strengthen the findings because the results are based on data from literature and only GWP was used as the damage category (Patel and Panwar, 2023). Conducting a credible and valid LCA for composting of organic waste with a comprehensive interpretation is complex (Blengini, 2008). Saharudin et al. (2024) provided an LCA with evidence that high temperature biochar had lower overall impacts compared to the low temperature biochar that was prepared from palm and rice residues at 300–600 °C. Results from LCA studies can be significantly affected by different impact assessment methods used (Garcia et al., 2020; Matuštík et al., 2022) and also depends on the sensitivity of factors such as system boundary conditions, transport distance in the overall system, impact indicators evaluated, and impact interpretation method used as well as the type of biochar used (Kumar Mishra et al., 2023). The results from LCA if integrated into a larger context can enhance the holistic understanding environmental impacts at a global scale. Conversion of waste to biochar and its soil application includes biochar characterisation and benefits and risks of its application in soil. Even though there is growing interest in conversion of waste to biochar and its soil application on a large scale, there are challenges and knowledge gaps hindering the comprehensive and prospective understanding on benefits of biochar application to soil (Luo et al., 2023).

Evaluations using LCA of composting of garden organics showed that combining centralized and on-site waste management systems minimizes environmental impact by reducing the waste volume for collection and transport, though transport distances were not optimized (Rotthong et al., 2023). Before 2023, there were only 25 reproducible and comparable studies on the LCA of garden waste composting (Oviedo-Ocaña et al., 2023), with clear components like functional units and system boundaries. However, just 16 compared composting to other technologies, highlighting the need for more research with well-defined life cycle inventory, functional units, and impact assessment methods for better standardization and comparability. Additionally, from 1995 to 2022, out of 1370 documents in Web of Science and Scopus, only 56 articles met criteria for coherence, detailed inventory, applications, and LCA mention (Serafini et al., 2023) with only three studies from Australia, indicating a need for more research in this context. Furthermore, studies comparing composting with on-site and off-site pyrolysis using primary data are scarce. Previous studies on biochar production have focused on techno-economic analysis and environmental impacts using secondary data (Matuštík et al., 2022; Nie et al., 2021; Ong et al., 2020). However, the environmental performance of converting organic substances to biochar with primary data remains unevaluated, with limited use of laboratory and field data (Amoah-Antwi et al., 2020; Matuštík et al., 2022). Even though the studies inform about the impacts, the magnitude of the impacts and how can it be leveraged in different damage categories have not been discussed in literatures yet. Additionally, comprehensive studies on soil-biochar interactions and carbon sequestration are lacking (Lade et al., 2020; Luo et al., 2023).

Therefore, to address the knowledge gaps mentioned above, this research was conducted with the aim of a comprehensive evaluation, including LCA of pyrolysis-biochar system for OS derived biochar prepared using flaming pyrolysis technology in Australia. Specifically, the objectives of this research were to (1) evaluate the potential environmental impacts of the composting process followed by screening of oversized (OS) materials as a current best practice (CBP) and conversion of OS to biochar AP (I) or direct garden waste to biochar AP (II) without composting, as an alternative best practice, (2) explore the influence of on-site production and optimise the transport distance according to different impacts assessed and (3) account for the holistic benefits of soil application of organic waste derived biochar.

2. Materials and methods

The standard methods ISO 14040, 2006 (ISO, 2006a) and ISO 14044, 2006 (ISO, 2006b), provides a key framework of a standard LCA (Finkbeiner, 2014; Serafini et al., 2023). The key features such as goal and scope definition, functional unit, impact assessment (methods and indicator used) are clearly stated in the text for the compliance with the standard method (ISO, 2006a). Additionally, the limitations of the study, and sensitivity analysis has been provided for compliance with the standard method (Finkbeiner, 2014; ISO, 2006b).

2.1. Process involved and different scenarios developed

Two processes are involved in this study. They are composting of garden organics and pyrolysis of oversized screenings of garden organics composting to produce biochar. The analysis is based on data obtained from a composting facility in City of Greater Geelong, Victoria, Australia and flaming pyrolysis system provider in regional Victoria, Australia (specific location obscured at provider request). The data used were from the case studies of two industries, which were achieved through a combination of interviews and primary data collection by site visit. After multiple zoom meetings with the members from participating organisations, a face-to-face interview was also conducted. Factual and process-based information were obtained during the interview and surveys to reduce the bias that could have occurred in data collection process. The questionnaire included questions regarding understanding the process technically, economically, and socially, required inputs and outputs, and perception. To reduce the bias in the collected data, different approaches were used for survey. A focused grouped discussion including 4–5 staff or team members with different responsibilities were interviewed at both locations. The data thus obtained was validated by the researcher and experts at Deakin University by visiting both the composting centre and pyrolysis facility.

2.1.1. Composting process

The compost generated by the City of Greater Geelong at their composting site goes through various processes (Fig. 1). Municipal garden waste is formed into windrows with surface area of approximately 929.6 m^2 and allowed to degrade undisturbed for around one week. During this time, the breakdown of organics generates heat, and the temperature of the windrows can reach 80 °C. This stage, known as pasteurization, kills harmful pathogens. The windrows are held at >55 °C for three days, before the waste pile is turned mechanically. The holding and turning process is repeated three times, which may take two weeks in total. The third step in the process is maturation. After two weeks of pasteurization, the piles of waste are irrigated with 10000 L of water per row (320–350) tonnes of garden organics. The irrigation process is repeated if the moisture is not trapped, or the organics require more water. The piles are then allowed to mature for around six weeks. The product of this third maturation step is unscreened compost, which contains particles of various sizes. The compost is screened using a screening machine to sizes <20 mm and >20 mm. The compost of size >20 mm is called oversized screenings which is used as a feedstock in this study to prepare biochar. Emissions during the maturation phase were not accounted for in the system boundary.

2.1.2. Biochar production process

The OS were transported around 250 km north of Geelong, for pyrolysis. A flaming pyrolysis system was used for the conversion of OS to biochar (Fig. 2). Feedstock was added using a hopper and two augers longitudinally. Air was injected from one end of the chamber using a fan. The chamber was partially ignited, which gave rise to hot gases responsible for gasification in the chamber. In the rectangular chamber, one side had gas flow, and the other side had limited oxygen, where biochar is produced at the temperature of 550 °C. Biochar was collected at the other end of the chamber. The syn gas from the chamber was collected in the combustion chamber, cooled in the cooling tower, and passed to the atmosphere using a stack. Emissions from the liquid by-products or reduced and avoided emissions were not accounted for in the system boundary.

Biochar was characterised for physicochemical properties such as pH, EC, elemental analysis, proximate analysis, surface area, surface morphology, water holding capacity, cation exchange capacity and carbon stability. pH and EC were evaluated using an electrometric method, elemental analysis was conducted using an elemental analyser and proximate analysis was conducted using a thermogravimetric analyser. Surface area was measured using N_2 adsorption desorption isotherms obtained from BET surface area analyser. Scanning electron microscope was used for evaluation of surface morphology. The water-holding capacity of the biochar was calculated using the method by (Gray et al., 2014) with minor modifications. Cation exchange capacity was analyzed and presented as number of base cations (Na^+, Mg^{2+}, K^+ and Ca^{2+}). The stability of the carbon content of biochar thus produced was analyzed using several available methods some of which are $H:C_{org}$ ratio, recalcitrance index, ^{13}C NMR DP spectroscopy (Adhikari et al., 2023; Adhikari et al., 2023b). These experimental data provide relevance to the results obtained from the LCA.

2.1.3. Scenarios developed for the study

Three scenarios were developed by the authors for the comparison of potential environmental impacts. This was done with an aim to identify the alternative best practices with lower potential environmental impacts.

Scenario one - Current best practice (CBP): The CBP includes the

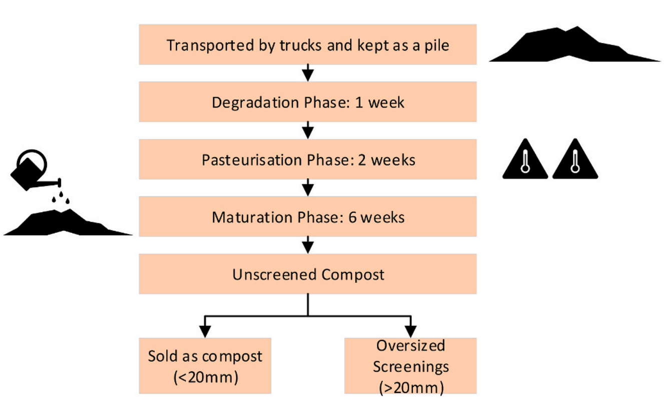

Fig. 1. Schematic diagram of composting and oversized screening at garden organics composting facility, COGG, Victoria, Australia.

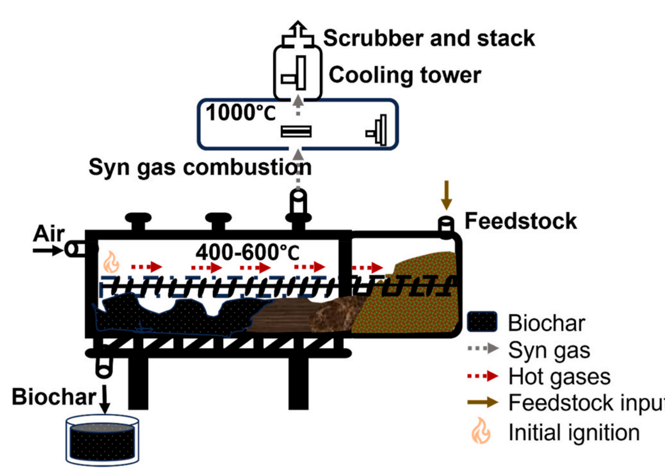

Fig. 2. Biochar production process using a flaming pyrolysis technology.

composting of municipal garden organic waste from the City of Greater Geelong (CoGG). The garden waste comprises mostly plant cuttings, trimmings, grass clippings, tree residue and storm-damaged trees, and garden clippings. This garden waste is delivered to the green waste facility at Moolap Station, Victoria, by refuse collection vehicles, screened for contaminants and ground before being transferred to the council composting site in Anakie, Victoria, for composting. Screening of oversized materials from marketable compost occurs after the composting process.

Scenario two - Alternative practice-one AP (I): This scenario includes converting the OS from the composting process to biochar using the flaming pyrolysis system from a private company in regional Victoria, approximately 240 km from the composting facility. Weekly, 150 tonnes of organics are discarded as OS, which if pyrolyzed, could be used as a valuable resource. Therefore, its large availability and lack of market make it a suitable feedstock for biochar production.

Scenario three - Alternative practice-two AP (II): Finally, the third scenario includes the process of conversion of garden organics directly to biochar with minimum transport on-site. The screening of large materials is conducted in the collection site, prior to the receival of garden organics in composting site, so composting processes are not included in the scenario. Under this scenario, different transportation distances were also explored to simulate pyrolysis facilities located closer and further from the source of the waste.

2.2. Life cycle assessment method

2.2.1. Goal identification, system boundary, and functional unit

The goal of this study was to evaluate the potential environmental impacts of CBP and two alternative scenarios (Fig. 3). The system boundary is cradle to gate. However, the study is comprehensive because further evaluation of environmental, social and economic benefits using alternative practices has also been discussed along with the results from LCA.

The functional unit applied in this research is the preparation of one tonne of biochar. The bulk density of the organic waste biomass was reported to be 300–500 kg/m^3 in the interview. It was assumed to be 363 kg/m^3 (López et al., 2010). This study analyzed the bulk density of recirculated yard trimmings, which is also referred to as the waste of the composting facility. During the lab conversion of these organic biomass to biochar we observed that the yield of biochar was 38%, which validates the information from interview of pyrolysis provider industry i.e., ~30%, highly depending on the moisture of the feedstock. This bulk density and yield (%) were used to identify the amount of organic waste and oversized screenings used to produce one tonne of biochar. From our calculation, 6.2 tonnes of organic waste produce 2.3 tonnes of OS of compost and this amount of OS of compost produces 1 tonne of biochar.

2.2.2. Life cycle inventory (LCI)

Data collection: Primary data was collected from interviews and focused discussions with the personnel providing biomass and producing biochar. Additional data available in databases such as Eco invent, peer-reviewed articles, statistical yearbooks or reports, and estimation was used when required. While the assumptions are made for certain aspects of inputs and outputs in the process, the references are cited and clearly stated. Primary data for biochar characterization was obtained from lab experiments for the use of biochar prepared and its benefits.

2.2.2.1. LCI for current best practice. The LCI for CBP included the inputs and outputs used during the composting process. 6.2 tonnes of

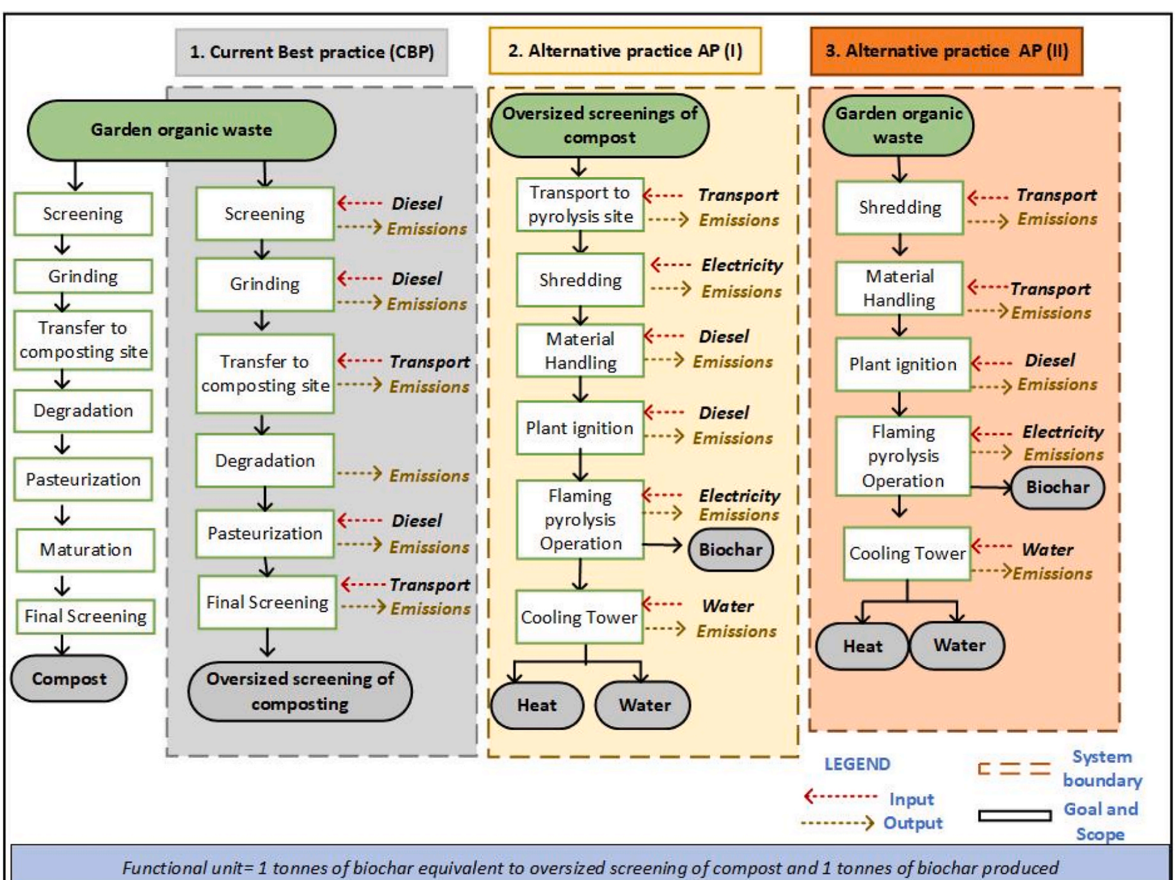

Fig. 3. Schematic diagram for evaluation of potential environmental impacts in different scenarios with distinct system boundary for three scenarios with input, output, and functional unit.

organic waste, 179.4l of diesel, 1012KWh electricity, and 65.7l of water along with 83.03 t × km transportation was used as input (Table 1). This input was used to produce 2.3 tonnes of OS compost as this amount of material was required to finally produce 1 tonne of biochar. The assumption for this input was based on 70% yield rate of compost (Abdul Rahman et al., 2020). After the interview with the officer at composting facility, it was noted that the half of the compost was marketable, and the other half was oversized screenings of composting. Thus, 6.2t of organic waste gives 2.3t of oversized screening of compost. The outputs regarding emissions were used from (Kumar et al., 2011). The emissions were normalized to the exposed surface area of the composting windrows for the CoGG composting site. It was assumed that the volume of the windrow was 1722 m^3, reflective of the measured dimensions of 120m long and 7m wide with 4.1m height (5.4m on the slope). The exposed surface area was 1350 m^2 which was finally normalized to the functional unit. The normalized emissions for LCI are provided in Appendix B, (Table B1). The internal transportation required as well as the inputs for the process such as screening of waste for contamination, grinding of waste, transfer of ground waste to trailer truck for transport to composting site, road transport to composting site, material handling in composting site for degradation process, turning piles for pasteurization, and maturation as well as water required for irrigation during maturation has been accounted for in the inventory.

2.2.2.2. LCI for alternative best practice AP (I). Two different alternative practices were evaluated. The first, AP (I), considers transportation of oversized screening (OS) after composting, for 240 km from the local council site to produce biochar at a regional processing facility in Victoria, Australia. Transport of 2.3 tonnes of OS compost for 240 km was included in the inventory as road transport (Table 2). Other material handling in the site for processes such as shredding, use of energy for startup of the equipment, electricity consumption by the flaming pyrolysis technology, and water required in the cooling tower is included as input for AP(I). The amount of biochar produced in the process, emissions, along with water and heat are included in the output for LCI. Emission of CO_2 during the pyrolysis process was estimated using the carbon content of the feedstock and resultant biochar. Detailed

Table 1
Life cycle inventory showing input and output flows for the current best practice (CBP) scenario of composting followed by screening of oversized materials. *(f)* means the functional unit for CBP.

Input flow for CBP	Unit	Amount	Data source
Diesel	l	179	Primary data
Electricity, production mix UCTE - UCTE	kWh	1012	Primary data
Freshwater	l	65.71	Primary data
Organic waste (garden waste)	t	6.13	Primary data
Transport, freight, lorry >32 metric ton, EURO5	t × km	83.03	Primary data
Output flow for CBP	**Unit**	**Amount**	**Data source**
Compost (oversized screenings)	t	*(f)* 2.3	Primary data
1,2-Benzenedicarboxylic acid, di-c8-10-alkyl esters	kg	0.64	Kumar et al. (2011)
Acetaldehyde	kg	1.93	Kumar et al. (2011)
Aldehydes, unspecified	kg	0.08	Kumar et al. (2011)
Alkenes, C4	kg	0.13	Kumar et al. (2011)
Ammonia	kg	0.05	Kumar et al. (2011)
Dimethyl disulfide	kg	0.01	Kumar et al. (2011)
Dioxins and furans, unspecified	kg	0.01	Kumar et al. (2011)
Hydrocarbons, aliphatic, alkanes, cyclic	kg	0.08	Kumar et al. (2011)
Ketones, unspecified	kg	0.05	Kumar et al. (2011)
Methane	kg	0.25	Kumar et al. (2011)
Methane, biogenic	kg	0.47	Kumar et al. (2011)
PAH, polycyclic aromatic hydrocarbons	kg	0.04	Kumar et al. (2011)
Wastewater	t	1.83	Kumar et al. (2011)
Volatile organic compound	kg	0.51	Kumar et al. (2011)

Table 2
Life cycle inventory showing input and output flows for the alternative best practice (I) scenario for composting followed by oversized screenings (OS) that are transported to an off-site facility to produce biochar. *(f)* means the functional unit for AP (I).

Input flow for AP (I)	Unit	Amount	Data source
Diesel	l	67.84	Primary data
Electricity, production mix UCTE - UCTE	kWh	828	Primary data
Freshwater	l	9200	Primary data
Oversized screening from composting	t	2.3	Primary data
Transport, freight, lorry >32 metric ton, EURO5	t × km	552	Primary data
Output flow for AP (I)	**Unit**	**Amount**	**Data source**
Biochar	t	*(f)* 1	Primary data
Heat	MJ	65320	Basu (2018)
Water for industrial use	l	9000	Primary data
1,2-Benzenedicarboxylic acid, di-c8-10-alkyl esters	kg	0.64	Kumar et al. (2011)
Acetaldehyde	kg	1.93	Kumar et al. (2011)
Aldehydes, unspecified	kg	0.08	Kumar et al. (2011)
Alkenes, C4	kg	0.13	Kumar et al. (2011)
Ammonia	kg	0.05	Kumar et al. (2011)
Dimethyl disulfide	kg	0.01	Kumar et al. (2011)
Dioxins and furans, unspecified	kg	0.01	Kumar et al. (2011)
Hydrocarbons, aliphatic, alkanes, cyclic	kg	0.08	Kumar et al. (2011)
Ketones, unspecified	kg	0.05	Kumar et al. (2011)
Methane	kg	0.25	Kumar et al. (2011)
Methane, biogenic	kg	0.47	Kumar et al. (2011)
PAH, polycyclic aromatic hydrocarbons	kg	0.04	Kumar et al. (2011)
Wastewater	t	1.83	Kumar et al. (2011)
Volatile organic compound	kg	0.51	Kumar et al. (2011)
Cadmium	mg	0.04	Tisserant et al. (2022)
Carbon dioxide	kg	1578	Based on C content
Chromium	mg	2.15	Tisserant et al. (2022)
Copper	mg	4.88	Tisserant et al. (2022)
Mercury	mg	0.01	Tisserant et al. (2022)
Nickel	mg	0.97	Tisserant et al. (2022)
Zinc	mg	19.3	Tisserant et al. (2022)
Particles (>PM10)	mg	29250	Tisserant et al. (2022)
Particles (PM0.2 - PM2.5)	mg	103250	Tisserant et al. (2022)
Particles (PM2.5 - PM10)	mg	62400	Tisserant et al. (2022)

description of the process is provided in Appendix B, section B2. Other emissions are taken from (Tisserant et al., 2022), and normalized to the functional unit of the study (1 tonne of biochar produced). Hot water was produced in the pyrolysis process. This was documented in terms of heat and water output. The heat from the flaming pyrolysis system was referenced (Basu, 2018) and water output was used according to the data from the interview. The other assumptions made were, water from the process is used as dust suppressant on-site and to quench the biochar produced. The gas emissions from the stack were fully combusted before release to the atmosphere.

2.2.2.3. LCI for alternative best practice AP (II). The third scenario, AP (II), included selected inventory data from AP(I) as listed in Table 3, however, composting inputs or outputs were not included, because screening of organic waste was assumed before the composting process. Tisserant et al. (2022) evaluated pyrolysis of forest residue and organic biomass in their study, therefore in this study both oversized screenings and garden organic waste are assumed to have similar emissions. Additionally, the transport distance in this scenario was zero as on-site biochar production was considered for this scenario.

2.2.3. Life cycle impact assessment (LCIA)

LCIA was conducted using Open LCA software. Primary data along with previous literature and Ecoinvent database was used for data acquisition. The LCIA method that was used in Open LCA software was ReCiPe 2016 Midpoint (H), linking the default providers and unit process (Huijbregts et al., 2020). The ReCiPe 2016 Midpoint (H) uses 18 impact indicators. They are Freshwater eutrophication (FEu), Human non-carcinogenic toxicity (HNCT), Ozone formation- Human Health (OFHH), Ionizing Radiation (IR), Stratospheric Ozone Depletion (SOD), Fine Particulate Matter Formation (FPMF), Human Carcinogenic Toxicity (HCT), Ozone Formation-Terrestrial Ecosystems (OFTE), Marine Eutrophication (MEu), Fossil Resource Scarcity (FRS), Freshwater Ecotoxicity (FE), Global Warming (GW), Land Use (LU), Water Consumption (WC), Terrestrial acidification (TA), Marine ecotoxicity (ME), Mineral Resource Scarcity (MRS), and Terrestrial Ecotoxicity (TE). This method converts the life cycle inventory input to above mentioned 18 life cycle impact indicators on a midpoint level by considering the environmental impact per unit stressor, called as characterisation factor. These characterisation factors at midpoint level identify the environmental impact and cause. For any impact category in the life cycle inventory, there is a connection of process with environmental flows. The midpoint level characterisation is strongly related to the environmental flows, and have varying impact indicators compared to the endpoint level where the characterisation is based on three categories such as human health, ecosystem quality and resource scarcity (Hauschild and Huijbregts, 2015). Therefore, in this study we used ReCiPe 2016 Midpoint (H) as a LCIA method.

Table 3
Life cycle inventory showing input and output flows for the alternative best practice (II) scenario of directly producing biochar from oversized screenings of organic wastes on-site, thus replacing composting for these materials *(f)* means the functional unit for AP (II).

Input flow for AP (II)	Unit	Amount	Data source
Diesel	l	115.68	Primary data
Electricity, production mix UCTE - UCTE	kWh	828	Primary data
Freshwater	l	9200	Primary data
Organic waste (garden waste)	t	2.3	Primary data
Output flow for AP(II)	**Unit**	**Amount**	**Data source**
Biochar	t	*(f)* 1	Primary data
Heat	MJ	65320	Basu (2018)
Water for industrial use	l	9000	Primary data
Cadmium	mg	0.04	Tisserant et al. (2022)
Carbon dioxide	kg	1578	Based on C content
Chromium	mg	2.15	Tisserant et al. (2022)
Copper	mg	4.88	Tisserant et al. (2022)
Mercury	mg	0.01	Tisserant et al. (2022)
Nickel	mg	0.97	Tisserant et al. (2022)
Zinc	mg	19.3	Tisserant et al. (2022)
Particles (>PM10)	mg	29250	Tisserant et al. (2022)
Particles (PM0.2 - PM2.5)	mg	103250	Tisserant et al. (2022)
Particles (PM2.5 - PM10)	mg	62400	Tisserant et al. (2022)

2.2.4. Life cycle interpretation

The results from LCI and LCIA were evaluated for the three scenarios. The sensitivity of data to different parameters with respect to the goal and scope of the study was evaluated. The results were then communicated effectively in the form of tables and graphs in relation to the large-scale environmental impacts.

2.3. Sensitivity analysis

The environmental impacts in 18 different impact categories were identified for 3 different scenarios. However, the fact that the environmental impacts change with the change in transport distance could not be overlooked. The sensitivity of transport distance was evaluated using two different approaches.

Firstly, the influence of transport distance for biochar production was evaluated for the scenario AP(II). For this purpose, tests were run starting from 100 km up to 1600 km, increasing every 300 km. However, due to the nature of the results, only impacts up to 900 km were evaluated in detail. Secondly, the magnitude of potential impacts that could be leveraged while using scenario AP(II), compared to scenario CBP was identified. This includes evaluating how far biochar production can be scheduled so that the process still holds the current environmental impacts but has additional soil and carbon sequestration benefits by use of biochar produced from AP(II).

2.3.1. Uncertainty analysis

In addition to optimisation of transport distance, uncertainty in the results due to uncertainties in different input and output parameters have been evaluated using a comprehensive Monte-Carlo analysis (10,000 simulations). A triangular distribution approach was used due to a small sample size of the results and assumptions made (Tisserant et al., 2022). For triangular distribution, maximum, minimum and mode values of each parameter was provided before simulation. The maximum and minimum values were assumed to be around 10–20% skewed from the median depending on the type of the parameter (Fawzy et al., 2022; Jungbluth et al., 2008). The detailed assumption for each parameter is provided in appendix A (Tables A1–A3).

2.4. Assumptions and limitations

Given the assumptions and limitations in methodology and data analysis, the conclusion from this study should be regarded as conservative and specific to waste management technique such as composting and pyrolysis. These limitations provide opportunities for future research. The specific assumptions and limitations of this study are as stated below.

- In the composting stage (CBP), only the emissions until pasteurization stage was considered for output in the LCI (Table B1). The emissions from literature were converted to 21 days (8 h per day) and used as input, therefore, this assumption must be considered when reporting the results.

- It was assumed that the output in CBP is specific to the oversized screening of composting and the marketable compost <20 mm, is not accounted for. Additionally, in AP(I) oversized screenings of compost is used as input and AP(II) uses the garden organics before composting as input.
- As the garden waste undergoes screening process, possible contaminants such as glass and metals are screened and assumed to be removed, therefore, the contaminants in the compost and biochar thus produced are not accounted for.
- Potential environmental challenges such as leachate collection and treatment, and odour and noise generated during composting were not accounted for in the LCA. In practice, leachate from the composting facility was collected in an artificial pond, where the water is biologically treated and continuously monitored by the composting facility.
- The benefits of soil application of compost and biochar have not been compared in this study; however, the soil carbon sequestration benefits of biochar have been discussed in principle, to provide context for this alternative product.
- It is also important to consider that the inventory only includes the collection of feedstocks from the collection centre to the production of biochar. Additional benefits of biochar post-use (e.g. from the application to soil as a soil-improver, with benefits such as the addition of carbon to the environment) are not captured by this analysis.
- Further, because the analysis is reflective of two specific industries and their associated inventories in Australia, it does not capture potential variations, such as seasonal changes, and weather effects in emission related to composting.
- Finally, as the physiochemical properties of biochar such as porosity, and carbon content play an important role in carbon sequestration (Xia et al., 2024), the results in this research correlate mostly with garden organics, and compost as a feedstock.

3. Results and discussion

3.1. Biochar characterization

Biochar prepared from the oversized screening of composting was alkaline in nature with pH of 10 ± 0.01 and EC of 3160 ± 57 μS/cm. It had a surface area of around 8 m^2/g and this biochar increased the water holding capacity by 45%v/v (Adhikari et al., 2023). The biochar had a high concentration of exchangeable calcium (480 ± 15 mmol/kg) and NH_4 exchangeable CEC of 1640 ± 140 mmol/kg. Carbon content of the biochar was 49.1 ± 0.8%, with a H:C molar ratio of 0.24 and recalcitrance value of 0.48, indicating highly stable biochar carbon (Adhikari et al., 2023b). Among the carbon present in the biochar, 90% of the biochar carbon was aromatic stable carbon. Detailed characterisation of this biochar can be found in previously published literature (Adhikari et al., 2023; Adhikari et al., 2023b), including reporting of the improvements this biochar brought to sandy soils after application at 2 wt %.

3.2. Environmental impacts in different scenarios

Our results show that with the change in scenarios from CBP, to AP(I) to AP(II), the overall impact of each process is reduced in each damage category (Table 4). The damage categories were grouped according to their inclination to each group. Firstly, fine particulate matter formation, global warming, stratospheric ozone depletion, and ozone formation-terrestrial ecosystems were all directly related to emissions to the atmosphere. Secondly, freshwater ecotoxicity, marine ecotoxicity, freshwater eutrophication, marine eutrophication, and water consumption were grouped as impacts to water resource. Thirdly, terrestrial acidification, land use and terrestrial ecotoxicity were used to evaluate impacts on land resource. Finally, the potential impacts on resources and human health, included mineral resource scarcity, fossil resource scarcity, ionizing radiation, human carcinogenic toxicity, ozone formation-human health, and human non-carcinogenic toxicity. Using these data, we compared how these three different scenarios CBP, AP(I) and AP(II) would change their impacts in each group (Table 4).

Table 4
Environmental impacts for three different scenarios grouped according to impact to varying damage categories. CBP: current best practice, AP(I): oversized screening of composting to biochar, and AP(II): organic waste to biochar.

Damage categories	Impact indicators	Unit	CBP	AP (I)	AP (II)
Emission to air	Fine particulate matter formation (FPMF)	kg $PM_{2.5}$ eq	0.08	0.19	0.05
	Global warming (GW)	kg CO_2 eq	125	232	56
	Stratospheric ozone depletion (SOD)	kg CFC_{11} eq	0.0002	0.0003	0.0001
	Ozone formation, Terrestrial ecosystems (OFTE)	kg NOx eq	0.3	0.9	0.1
Water	Freshwater ecotoxicity (FE)	kg 1,4-DCB	0.3	1.3	0.1
	Marine ecotoxicity (ME)	kg 1,4-DCB	0.9	3.0	0.4
	Freshwater eutrophication (FEu)	kg P eq	0.0009	0.0051	0.0004
	Marine eutrophication (MEu)	kg N eq	0.0005	0.0015	0.0007
	Water consumption (WC)	m^3	−1479	−3801	989
Land	Terrestrial acidification (TA)	kg SO_2 eq	0.2	0.5	0.1
	Land use (LU)	m^2a crop eq	12	19	3
	Terrestrial ecotoxicity (TE)	kg 1,4-DCB	230	1479	39
Resources and human health	Mineral resource scarcity (MRS)	kg Cu eq	1	1	0.4
	Fossil resource scarcity (FRS)	kg oil eq	35	116	63
	Ionizing radiation (IR)	KBq Co-60 eq	0.3	233	231
	Human carcinogenic toxicity (HCT)	kg 1,4-DCB	7	41	1
	Ozone formation, Human health (OFHH)	kg NOx eq	0.3	0.8	0.1
	Human non-carcinogenic toxicity (HNCT)	kg 1,4-DCB	11	51	5

The results revealed that scenario AP(II) provided significant opportunities for reduced and negative impacts to LU, MRS, HCT, HNCT. AP(II) provided least environmental impacts to all the impact indicators, except water consumption and ionizing radiation. The potential GHG emissions can be reduced up to 55% by on-site pyrolysis of oversized garden waste compared to composting of this fraction of organic waste. The percentage change in the potential environmental impacts outlined by the ReCiPe midpoint 2016 method of the three different scenarios was evaluated (Fig. 4). The baseline impact was assumed to be zero for all impact indicators, which is indicated by the red dotted line in the figure.

3.2.1. Emissions to air

The AP(II) was observed to be a more sustainable option than

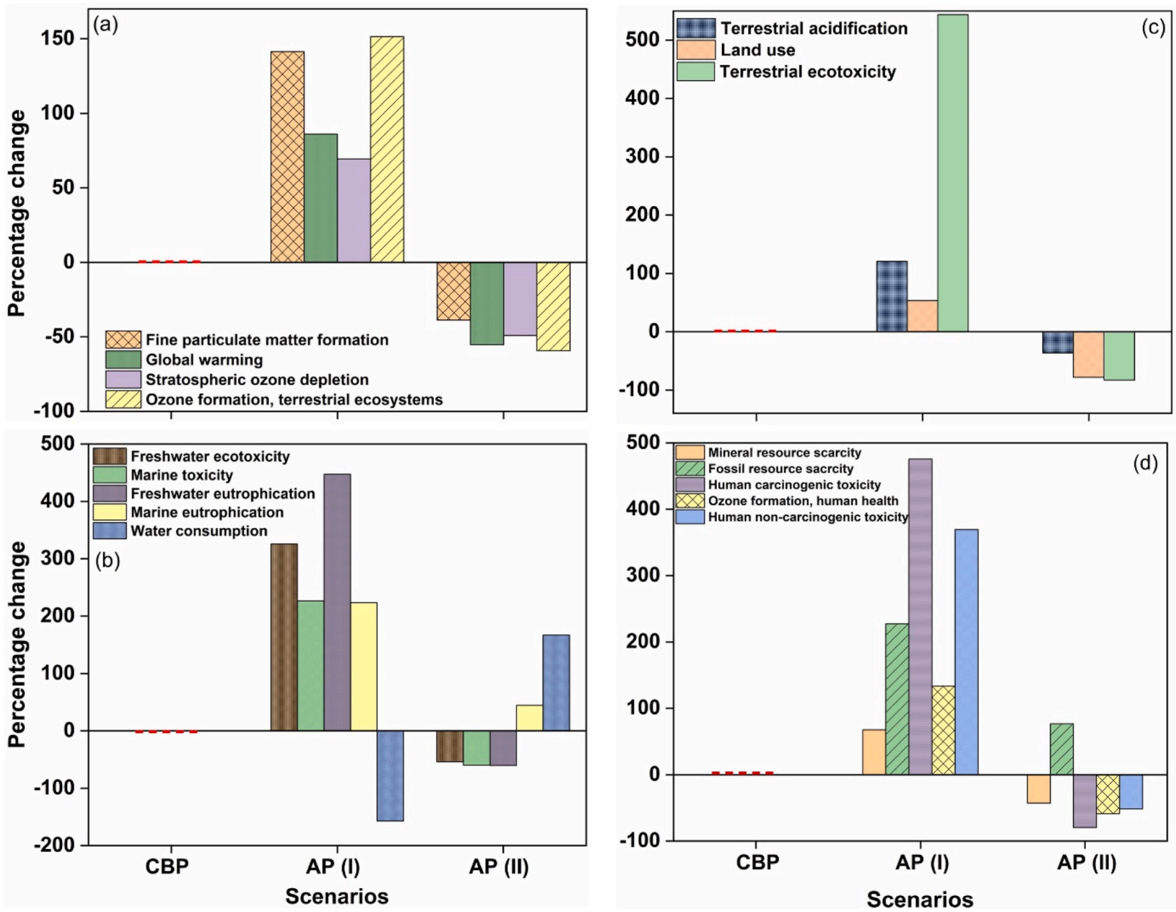

Fig. 4. Percentage change in impacts with change in each scenario shown by different impact indicators from LCIA in 4 different damage categories where (a) refers to impact to air and emissions, (b) refers to impact to water, (c) refers to impact to land and (d) refers to impacts to resources and human health. The percentage change is relative to the scenario current best practice (CBP).

composting, with lower emissions of the gases contributing to global warming such as PM $_{2.5}$, CO_2, CFC_{11} and NO_x. AP(II) has a lowest emission of around 56 kg CO_2 eq, with lowest PM formation 0.05 kg $PM_{2.5}$ eq and contributes to 0.0001 kg CFC_{11} eq for stratospheric ozone layer depletion (Table 4). Compared to the CBP, potential impacts relating to greenhouse gas emissions are reduced by AP(II) by up to 59% (Fig. 4a). In the CBP, the FPMF is influenced by the combustion of diesel and electricity. As AP(I) includes using the oversized screenings of composting, the additive effect of inputs from composting and its conversion to biochar increases the FPMF. However, FPMF is further decreased in scenario AP(II), based on the assumptions for AP(II). Additionally, AP(II) avoids the steps such as the windrow composting by converting the garden waste directly to biochar, contributing to lowest GHGs emission. The stratospheric ozone depletion (SOD) increased in scenario AP(I) by more than 69% due to the processes such as transport, use of diesel, electricity, and transport of the feedstock to prepare biochar. These impacts were reduced in scenario AP(II) by 49%. In addition, ozone formation affecting terrestrial ecosystems (OFTE) also showed a similar trend, where AP(I) increased the impact by around 151% compared to CBP, and AP(II) reduced this by 59%. Looking from a planetary boundary perspective, the impact indicators such as SOD and OFTE are still in the safe operating space (Richardson et al., 2023) compared to the preindustrial scenario. These indicators have not been increased and are not in the zones of increasing risk. However, the CO_2 concentration and radioactive forcing have already trespassed the limit, therefore, these need more consideration. Given the high magnitude of effects from the production of a functional unit of product, the substantial impact a process may have on emissions when scaled up can be envisaged.

Studies showed similar results regarding global warming (GW), where Colón et al. (2015) mentioned that windrow composting showed GW of around 150 kg CO_2 eq/Mg of organic municipal solid waste. Similar results were provided by our study including uncertainties, where CBP provided around 125 kg CO_2 eq per tonne of compost produced from garden organics (Figure C1). The lower result for global warming without uncertainty n parameters for CBP may be due to the degree of uncertainty and our focus on oversized screenings rather than the entire compost produced. There was a similar trend of results after the uncertainty analysis using Monte-Carlo simulations (Figure C1). With the uncertainty analysis, AP(II) was observed to be the more sustainable practice among the three studied waste management practices regarding the emission to air.

3.2.2. *Water*

The result from this study reveals that AP techniques are more effective in reducing eutrophication and toxicity to water resource compared to the CBP. AP(I) significantly reduced the potential impacts on water by 150–450% compared to CBP. Water consumption was reduced by 157% (−1479 to −3801 m^3) in AP(I). However, AP(II) shows some water consumption during the process and needs improvement, but other impacts to water such as freshwater and marine eutrophication and ecotoxicity are significantly reduced by up to 60% compared to CBP (Table 4). Production of 1 kg of compost (<20 mm) from garden organics (CBP), contributes to around 0.9 g of phosphorus in the freshwater bodies, whereas production of biochar with these garden organics, AP(II), contributes to only ~0.4 g of phosphorus. Eutrophication in the heavily polluted rivers was identified by presence of 0.000001 kg/L of phosphorus in Australian rivers (ANZECC, 1992). Compared to CBP and AP(II), AP(I) provides opportunity to reduce the risk of eutrophication by reduction of P deposit due to the process. The impacts observed in this study are generally low, however it should be considered that these impacts are for the functional unit of 1 kg of compost, or biochar produced using the system. Therefore, scaling up the waste management techniques in a real case will amplify the effect to water resource. A bigger picture view of planetary boundaries according to the earth system process of freshwater change including blue water (water required for river regulation and aquatic system integrity) (Porkka et al., 2023) and green water (water required for hydrological regulation of terrestrial ecosystems, climate, and biogeochemical processes) (Wang-Erlandsson et al., 2022), identified these system processes as beyond the limit compared to the pre-industrial era. A small project like the scenario developed by the authors in this article have shown that the water ecosystem can be significantly impacted by anthropogenic activities, which may contribute to a synergistic effect when combined with atmosphere and terrestrial ecosystem.

The impact values followed a similar trend when uncertainties were added to the system (Figure C2). Overall AP(II) was deemed to be the best for impact to the water resource where it reduced the freshwater ecotoxicity, marine ecotoxicity, and freshwater eutrophication. However, it slightly increased marine eutrophication (0.0005 kg N eq in CBP to 0.0007 kg N eq in AP(II)). Similarly, water consumption was found to be negative only for CBP and AP(I), and some water consumption was observed in the system, providing some opportunities for improvement for AP(II).

3.2.3. Land

The potential impacts on land as a resource from CBP, AP(I) and AP (II) were assessed by examining the extent of terrestrial acidification (TA), land use (LU), and terrestrial ecotoxicity (TE). Among the three scenarios, AP(II) is likely to be a sustainable and cleaner waste management technique due to its lowest terrestrial acidification potential (0.14 kg SO_2 eq) and ecotoxicity impacts (39 kg 1,4-DCB eq) along with the moderate impact on land use (3 m^2a crop eq). The CBP and AP(I) scenario has a significant overall impact on land, which may be reduced by adapting the alternative practice AP(II) by more than 70% (Fig. 4c). Even though AP(I) shows a net positive impact on land use, there is room for improvement for this technique regarding impacts such as TA, LU and TE.

The magnitude of potential environmental impacts reduced significantly by using a Monte-Carlo sensitivity analysis. However, the pattern of potential impacts using different techniques was like the data without variations. The results showed that AP(II) is the most sustainable practice for waste management (Figure C3). When projects like the CBP in large scale adapt to the proposed technique AP(II), the impacts to the environment could be potentially reduced and the levels of the earth system processes could be controlled. Inputs such as transport, and combustion of diesel, electricity used during the process production of agricultural machinery and shed influenced these environmental impacts. Even if the water consumption was observed to be low in AP(II), the leachate produced from the composting site led to other potential environmental impacts such as increased terrestrial acidification and ecotoxicity compared to CBP and AP(II) that would require active mitigation strategies. The impacts to forest cover represents the earth system process called land system change, which is measured as the area of forested land as a percentage of original forest cover. Deforestation, land use, and unexpected fire hazard are reducing the forest cover and have reduced the percentage of global forest cover from 100% in pre-industrial time to 60% in current scenario (FAO, 2020; Richardson et al., 2023).

3.2.4. Resources and human health

Among the three scenarios studied, AP(II) is observed to be the most sustainable in terms of mineral resource scarcity. CBP and AP(I) provide impact equivalent to extraction of 0.7 kg and 1.2 kg of copper respectively, whereas AP (II) indicates the extraction of 0.4 kg equivalent of copper, which is less than both CBP and AP(I) (Table 4). AP(II) reduced the mineral resource scarcity, and carcinogenic toxicity to humans and the environment compared to the CBP (Fig. 4d), however it increased the fossil resource scarcity to some extent compared to CBP. The formation of ozone impacting human health was increased by AP(I) but decreased up to 59% by AP(II). Electricity and diesel inputs and the use of machinery for material handling processes increased the ionizing radiation impact in AP(I) and AP(II).

The uncertainty analysis also showed a similar pattern, where it was observed that AP(I) showed higher potential impacts relating to mineral resource, fossil resource, ozone formation, and human health (Figure C4). From the analysis, AP(II) was observed to be a sustainable waste management technique for resource management. However, AP (II) has a room for improvement in managing the fossil resource, which can be balanced by soil application of biochar (Joseph et al., 2021).

3.3. Change in impacts with transport distance

The influence of transport distance has been explored in different ways in the LCA analysis in previous literature. However, the conclusions have not always been consistent. The sensitivity of the potential environmental impacts was evaluated with different transport distances. Our results in section 3.2 showed that AP(II) (biochar production at the organics collection centre) has the lowest environmental impact compared to CBP and AP(I). Thus, this analysis was conducted for case AP(II), where the same inputs and outputs were used, only changing the transport distance in Open LCA using the ReCiPe midpoint H, 2016 impact analysis method. The impacts of transportation distances from 100 to 1500 km was evaluated and detailed data is provided in Table C1. The impacts surpassed the impacts from the CBP after 900 km, so for this analysis only up to 900 km is presented (Fig. 5). Among 18 environmental impact indicators studied; 10 indicators showed a significant increase in the potential impact with transport distance. Five of the impact indicators did show a small increase but not as significant as the previous ten indicators and three showed no influence of transport distance (Table C1).

Results show that transport distance plays a significant role for potential environmental impacts on AP(II). Overall, there was an increase in impacts relating to emissions; global warming and ozone formation, and terrestrial ecosystems with an increase in transport distance (Fig. 5a). A significant increase was observed for global warming with an increase from 56 kg CO_2 eq to 249 kg CO_2 eq for 900 km increase in transport distance. However, the fine particulate matter formation did not show any significant change. A similar trend was observed for impact indicators relating to water: Impacts on freshwater and marine ecotoxicity along with freshwater eutrophication significantly increased with increase in transport distance, whereas the marine eutrophication did not show significant increase with transport distance (Fig. 5b). Freshwater ecotoxicity was increased from 0.14 kg 1,4-DCB at in-situ production to 3.16 kg 1,4-DCB at 900 km. Similar pattern was observed for marine ecotoxicity and freshwater eutrophication. Among the indicators showing an impact on land, the terrestrial acidification did not change as significantly as land use and terrestrial ecotoxicity (Fig. 5c). Finally, other environmental impacts such as toxicity to humans (carcinogenic and non-carcinogenic) and fossil resource scarcity increased with an increase in transport distance. In contrast, mineral resource scarcity, ozone formation: human health, and ionizing radiation did not change with an increase in transport distance (Fig. 5d). Therefore, evaluation and quantification of optimum transport distance are required to make any process carbon negative and improve the possible negative environmental impacts of the certain process (Cartier and Lembk, 2021; Dutta and Raghavan, 2014; Hammod, 2009; Owsianiak et al., 2021; Peters et al., 2015).

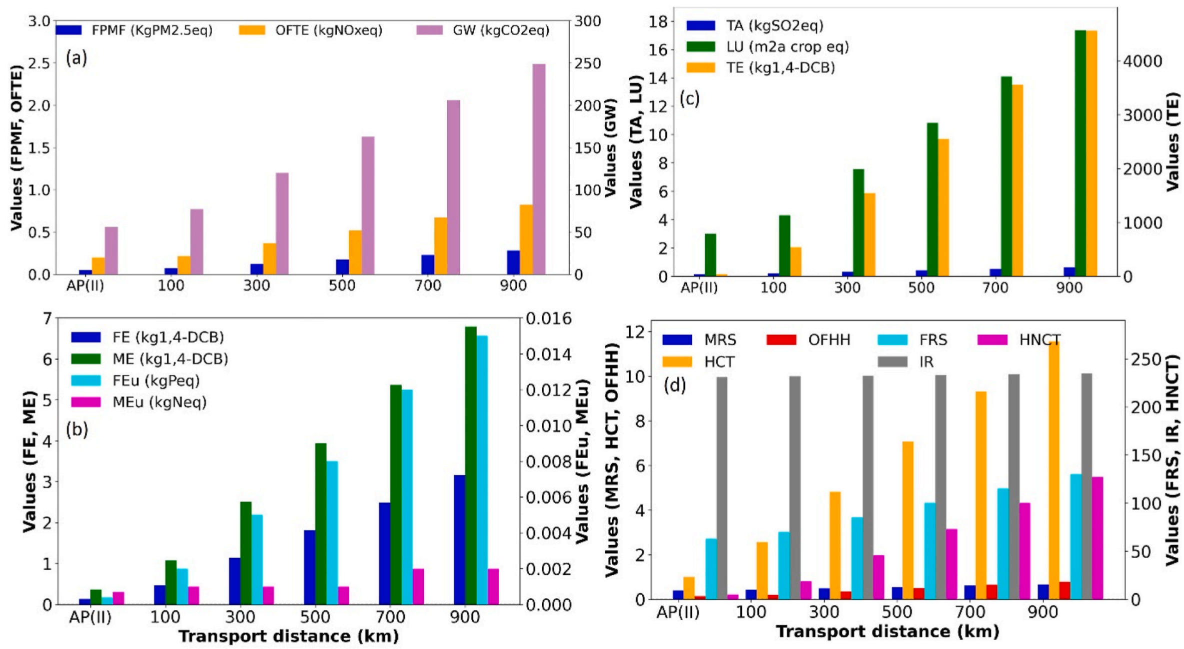

Fig. 5. Changes in impact with change in transport distance for alternative practice AP (II). (a) shows impacts indicators related to emissions such as fine particulate matter formation (FPMF), Ozone formation, terrestrial ecosystems (OFTE), and global warming (GW). (b) shows impacts on water resources such as (freshwater ecotoxicity) FE, marine ecotoxicity (ME), freshwater eutrophication (FEu), and marine eutrophication (MEu). (c) shows impacts on land, terrestrial acidification (TA), land use (LU), and terrestrial ecotoxicity (TE). and (d) shows impacts on others, mineral resource scarcity (MRS), ionizing radiation (IR), human carcinogenic toxicity (HCT), ozone formation, human health (OFHH), fossil resource scarcity (FRS), and Hunam non-carcinogenic toxicity (HNCT).

3.4. Evaluation of maximum transport distance

The results revealed that scenario AP(II) provided the least environmental impacts compared to CBP and AP(I). To evaluate the transport distance effect for different impact indicators, we ran the same analysis with 9 transport distances as mentioned in section 2.3. After that the values were compared to the impacts for CBP to compare the amount of impact that could be leveraged when adopting this new alternative practice, AP(II). The maximum transport distance for emission of FPMF, GW and OFTE by adopting AP(II), instead of CBP and having a similar emission would be 300 km. This indicates that the pyrolysis plant can be as far as 300 km from the organics collection facility and would still provide similar emission to the CBP. However, the maximum transport distance would be different if water resource impacts were prioritised, where transport of 300 km may cause significant impacts to water resource, compared to the current practice of composting of garden organics. To have a similar impact on land use, the transport can be as far as 500 km. Transportation of up to 500 km can also have potential impacts such as freshwater eutrophication and carcinogenic toxicity to humans (Table C1).

In summary, although conversion of garden organics to biochar on-site with minimum transport distance provides least environmental impacts, our analysis shows that it is still possible to include transport over practical distances and incur similar environmental impacts as CBP. It is possible that on-site biochar preparation may not be feasible in the actual scenarios as discussed above due to trade-offs including economies of scale, co-production of heat and bio-fuels and treatments needed for emissions. This research for the first time shows that transport distance during pyrolysis can have negligible impacts on one impact indicator but may have significant impact on another indicator. Thus, optimisation of transport distance is recommended to have a balance between benefits and trade-offs of the pyrolysis process. This analysis also shows that an alternative practice can be adapted and proposed with other co-benefits. Therefore, these findings are of great importance and need to be considered when scaling up the waste management industries and technologies.

3.5. Holistic benefits of biochar soil application

3.5.1. Biochar soil carbon sequestration

Biochar lifts the soil organic carbon ceiling by up to 25% within 8 years, while decreasing the soil organic carbon mineralisation by 18% when applied at the rate of 10 Mg per hectare of land (Joseph et al., 2021; Weng et al., 2022). Agricultural use accounts for approximately 427 million hectares of land in Australia (ABARES, 2023). Compost and biochar are two types of soil enhancers used by agricultural businesses in Australia. In 2016–17, compost was used for 209,072 ha of agricultural land as a soil enhancer, and biochar was used in 4319 ha of agricultural land (ABS, 2016–17). Comprehensive benefits of using biochar in agriculture as a soil enhancer can be estimated using this scenario at a national scale. Assuming that biochar is a stable form of carbon (Adhikari et al., 2023b) and 90% of compost (including oversized screenings) would be decomposed every 3 years (Joseph et al., 2021), or longer if oversized screenings are used., application of biochar would show significant benefits regarding soil carbon sequestration and CO_2 emission reduction as well as the other benefits to soil and crop yield (Patel and Panwar, 2023) compared to composting of garden organics. For example, approximately 1878 million Mg of biochar carbon could be sequestered across all agricultural land if 4270 million Mg of biochar is deployed at 10 Mg per hectare over 8 years. 90% of carbon present in biochar prepared from the AP(II) scenario is stable carbon according to the NMR study results (Adhikari et al., 2023a). In addition, this application could increase the soil organic carbon by 25% and reduce the soil carbon mineralisation. This will initially reduce the volume of garden waste going to landfill, composting or other management methods. It will also improve the soil productivity, soil water holding capacity and overall soil physical, chemical, and microbial environment. Therefore, conversion of garden organic waste to biochar with optimized conditions act as a soil improver as well as a climate saver.

3.5.2. Authors' perspective

The Life Cycle Impact Assessment (LCIA) in LCA analyses specific products/processes locally, while planetary boundaries gauge global environmental impacts based on Holocene conditions (Richardson et al., 2023). LCIA guides local decision-making, while planetary boundaries offer a broader perspective, evaluating nine critical factors. Integrating these methods highlight the need for holistic environmental assessment (Lade et al., 2020). For one of the factors, biosphere integrity (Richardson et al., 2023), net primary production (NPP) and human appropriation of the biosphere's net primary production (HANPP) are key measures. Anthropogenic activities strain resources to meet the needs of 8 billion people. Biochar is one of the effective, and proven method for carbon dioxide removal and is listed as a negative emissions technology (Guo et al., 2022; Jeswani et al., 2022; Werner et al., 2022). Our research suggests that biochar can enhance soil carbon, mitigating impacts like excessive land use and carbon sink depletion (Adhikari et al., 2023). Scaling this process could address human needs while safeguarding NPP. However, large scale production and application of biochar are not yet prevalent in the Australian region. To use this technology effectively and have maximum carbon sequestration from biochar production and use in a large scale, policy support, public awareness, market development and commercialization is required (Pourhashem et al., 2019; Zilberman et al., 2023). In addition, the research conducted in a case basis should be interpreted in the global level to provide a holistic view of environmental impacts provided by the product or process. Finally, this approach will help develop opportunities for products and practices with less emissions as well as aid large scale carbon sequestration for more than 100 years.

3.5.3. Social and economic impacts

Findings of this study hold significant social and economic implications. Application of biochar will provide benefits such as improved soil health, enhanced crop yields in underdeveloped areas and degraded soils, along with generation and offset of energy during production (Kumar Mishra et al., 2023). Additionally, soil carbon sequestration contributes to climate change mitigation, contributing to a sustainable environment. Socially, deployment of biochar will increase jobs opportunities and investment for farmers, engineers, and associated workers in the agricultural and biochar production sectors, enhancing community resilience and economic stability. Economically, the use of biochar acts as a slow-release fertilizer, reducing cost for traditional fertilizers, opportunity of market development for biochar and providing quantifiable economic benefits such as potential revenue from carbon credits. In this study, the maximum travel distance for each impact category has been identified, which can be implemented in composting facilities in other parts of the world. This can reduce the cost of the pyrolysis process with a confidence of impacts to different damage categories discussed in section 3.2. However, the barriers for commercialization of biochar technology such as high initial costs for technology establishment, lack of policy and government support, lack of standardised regulations and limited public acceptance and awareness effect large scale adoption of this technology. Enhanced awareness and support from policymakers, industry stakeholders, researchers and communities could aid in overcoming these barriers.

The use of biochar technology to improve waste management for cleaner and sustainable production that reduces waste, valorises waste, and reduces emissions can provide long term social and economic and environmental benefits. Future studies should not only focus on technical and quantifiable benefits but also the social and economic benefits from use of biochar technology at a large scale.

4. Conclusion

CBP performs moderately across all impact categories, thus indicating opportunities for optimisation. AP(I) shows significant increase in potential impacts; therefore, it cannot be recommended as the most sustainable alternative technology, mainly because it is an additional step, rather than replacing the composting process. AP(II), which directly produces biochar by screening organic waste before composting emerges as the most sustainable waste management technique with the lowest impacts on emissions, water, land, resources and human health, along with a moderate impact on marine eutrophication, fossil resource and water consumption. Therefore, it can be concluded that large scale implementation of AP(II) provides opportunities for wider environmental benefits such as waste management, improve land use efficiency and is best aligned with the UN Sustainable Development Goals. In addition, our novel findings show that while on-site production of biochar with minimal transport has the lowest environmental impacts, it is still possible to incorporate transportation of materials and reduce the environmental impacts as compared to a scenario that considers composting only. Transport of feedstock up to 500 km away from the waste collection on-site will still produce less emissions and impact to environment compared to the current scenario. In addition to being a sustainable waste management technique, large scale soil application of biochar acts as genuine transfer of carbon to the soil, avoiding emission from waste decomposition, along with removing and reduction emission from negative priming of soil carbon, and soil carbon sequestration by addition of long-term stable carbon to the soil. The potential environmental benefits and impacts of various strategies for managing garden organic waste through composting and/or biochar production were evaluated, to support effective valorisation of waste. Production of one tonne of biochar directly from garden waste without composting and using on-site pyrolysis would release approximately 56 kg CO_2 eq of GHGs. In that one tonne of biochar produced, around 45% is a highly stable aromatic carbon, which when applied in soil, can sequester this carbon for >100 years. GHG from pyrolyzing garden waste can be offset through soil application, creating a win-win scenario.

However, given the acknowledged limitations and assumptions made during this study, to truly evaluate various organic waste management strategies, future research should include cradle-to-grave LCAs with a detailed life cycle inventory, including local conditions, to thoroughly support decision-making.

CRediT authorship contribution statement

Sirjana Adhikari: Writing – review & editing, Writing – original draft, Resources, Methodology, Investigation, Formal analysis, Conceptualization. **M.A. Parvez Mahmud:** Writing – review & editing, Validation, Supervision, Resources, Conceptualization. **Ellen Moon:** Writing – review & editing, Validation, Methodology, Investigation, Conceptualization. **Wendy Timms:** Writing – review & editing, Visualization, Validation, Supervision, Resources, Project administration, Methodology, Funding acquisition, Conceptualization.

Declaration of competing interest

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

Data availability

Data will be made available on request.

Acknowledgements

Authors would like to thank Deakin University for providing DUPRS scholarship to Sirjana for her PhD. This project has been reviewed by the faculty of Science and Built Environment Human Ethics Advisory Group, Deakin University and recognises that it complies with the *National Statement on Ethical Conduct in Human Research (2007); Updated 2018).*

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.

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