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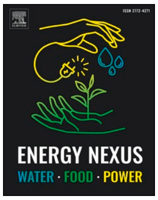

Environmental, economic, and social trade-offs in biochar and biosurfactant-based soil remediation: A critical review based on mass flow analysis

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ABSTRACT

Biochar and biosurfactants are emerging bio-based materials for remediating contaminated soils. While their pollutant removal mechanisms are well studied, broader environmental, economic, and social implications remain underexplored. Existing studies often rely on a 1 kg functional unit, limiting direct comparisons. This critical review evaluates and quantifies the multidimensional sustainability trade-offs of using biochar and biosurfactants to remediate one hectare of contaminated land, based on real-world applications. Use of common functional unit (1 hectare of land treatment) enables direct, meaningful comparison.

Material flow analysis reveals biochar's superior energy efficiency (net output for the grid ~290 GJ/ha) and economic returns, despite higher production emissions (100 kg CH_4, 55 kg N_2O, 38 kg PM_{10}, and 1.7 kg PAHs/ha). In contrast, biosurfactants emit negligible direct pollutants but demand significantly more energy (2340 GJ/ha). Both materials offer social benefits, such as enhanced rural livelihoods and health, yet face challenges like land use conflicts and patent barriers. Policy measures are proposed to mitigate these issues. Finally, the synergistic use of biochar and biosurfactants is highlighted as a promising avenue for future research in sustainable soil remediation.

1. Introduction

Soil remediation has recently gained significant attention due to the shrinking availability of arable land and increasing urbanization. Ensuring future food security therefore relies, in part, on restoring contaminated soils. Petroleum hydrocarbon-polluted sites are particularly challenging to remediate. Traditional remediation methods (incineration, thermal desorption, in-situ chemical oxidation, vitrification etc.) often involve higher energy consumption or synthetic chemicals, which may leave toxic residues. Consequently, there is growing interest in bio-based materials for soil treatment, such as biochar and biosurfactants.

Biochar is a special organic material that is made by pyrolyzing biomass in the absence of oxygen. During the pyrolyzing process, its moisture content and some of the organic matter escape as syngas, particularly containing methane, oxides of nitrogen, water vapor, etc. It is identified in the literature that the composition of biochar determines its physical and chemical characteristics [1]. Further, the composition mainly depends on the feed materials. In lignocellulosic feedstock materials, the composition can be successfully predicted with linear models showing a strong coefficient of correlation (adj $R^2 = 0.98$) [2]. Moreover, the carbon concentration determines reactivity of biochar,

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depending on the persistent free radical intensity (PFR) (correlation of 0.80) [2]. Based on these chemical and physical characteristics, biochar has a wide variety of sustainable applications in different industries. For instant, environmental, energy, agriculture, medicine, cosmetics, food, catalyst, and construction sectors (Fig. 1) [3]. In addition, biochar shows better stability in natural environments, especially in soils, aiding carbon sequestration for centuries.

In contrast, biosurfactants are bio-organic materials produced through the fermentation of biomass (or agro-industrial waste). Like all surfactants, biosurfactants are amphipathic materials, containing both hydrophilic and hydrophobic parts, which makes them versatile in dissolving in polar or non-polar liquids. For the same reasons, biosurfactants can form preferential partitions at non-dissolving liquid-liquid interfaces (such as water and oil) with varying degrees of polarity and hydrogen bonding [4]. Moreover, the chemical structure of biosurfactants facilitates the formation of micelles, bilayers, or vesicles at the interface when mixed. Depending on the energy charge in the hydrophilic part of the biosurfactant, they can be categorized as non-ionic, cationic, anionic, or amphoteric [4]. Unlike biochar, the applicability of biosurfactants varies—from detergent production to emulsification, foaming, and dispersing functions. However, due to their biodegradable nature and other properties, biosurfactants are also widely utilized in various applications. Some of the applications include environmental, energy, agriculture, medicine, cosmetics, food, textile, nanotechnology, and cleaning sectors (Fig. 2) [5,6].

Contaminated site remediation, especially the remediation of contaminated soil, is one common application for both biochar and biosurfactants coming under environmental applications. In the case of biochar, remediation of soils can occur through adsorption, chemical precipitation, or cation exchange [2]. Key factors aiding these mechanisms include high surface area (10–500 m^2/g), high cation exchange capacity (10–50 cmol/kg), and neutral to slightly alkaline pH (6–10) [2,7,8]. Other than the feed materials, the pyrolyzing temperature, vapor residence time and other chemical and physical alterations can also have a significant effect on the properties mentioned above [3,7]. Biosurfactants, on the other hand, can be used to remediate contaminated soil by reducing surface tension, allowing pollutants to dissolve more readily in the liquid fraction [9]. Moreover, biosurfactants can increase the microbial population in soils, aiding the biological degradation of pollutants like hydrocarbons. Additionally, microbial activity can reduce the mobility of metal ions in soil [10,11]. It has been observed that biosurfactants can reduce the surface tension of pollutants from 72 mN/m to 30 mN/m in some cases [10]. Some studies report a synergistic approach, where biosurfactants and biochar are used together for site remediation, utilizing the adsorption capacity of biochar and the solubilization effect of biosurfactants. Remediated soil shows an increase in total polycyclic hydrocarbon degradation by 28–30 % when 5 % biochar and 1000 ppm biosurfactants are used together [10]. However, it has also been identified that the high adsorption capacity of biochar can reduce the bioavailability of both pollutants and surfactants, thereby reducing overall performance [10]. Hence, synergistic approaches require further research in this area.

Though the soil remediation applications of both biochar and biosurfactants are well-documented in the literature, their environmental, economic, and social effects are often studied separately. This reduce the comparability of the two approaches. As a result, it is difficult to determine which method is more beneficial for a particular soil remediation case under different aspects (environmental, economic, and social). To address this gap, this review presents a performance evaluation of the environmental, economic, and social benefits and impacts of biochar- and biosurfactant-based soil remediation. To achieve this objective, this study will critically examine the existing literature on contaminated soil remediation using biochar and biosurfactants. Based on the critical review, the manuscript will then present an energy and emission flow analysis focusing on the treatment of 1 hectare of contaminated land using the two materials. The economic impact will be calculated based on the material requirements for treating 1 hectare of land. Finally, social impacts and benefits will also be evaluated based on data from the literature. The novelty can be seen in the comparison between the two materials based on the application (treatment of 1 hectare of contaminated land) and evaluation of total sustainability focusing on environmental, economic and social impacts. This manuscript aims to help understand how the two different soil remediation methods perform across the environmental, economic, and social dimensions.

2. Environmental implications of different stages of biochar and biosurfactants' life cycle

Production of biochar or conversion of waste biomass into biochar demonstrates several advantages. Mainly, the production mitigates the carbon dioxide emissions from biomass by keeping carbon in a much

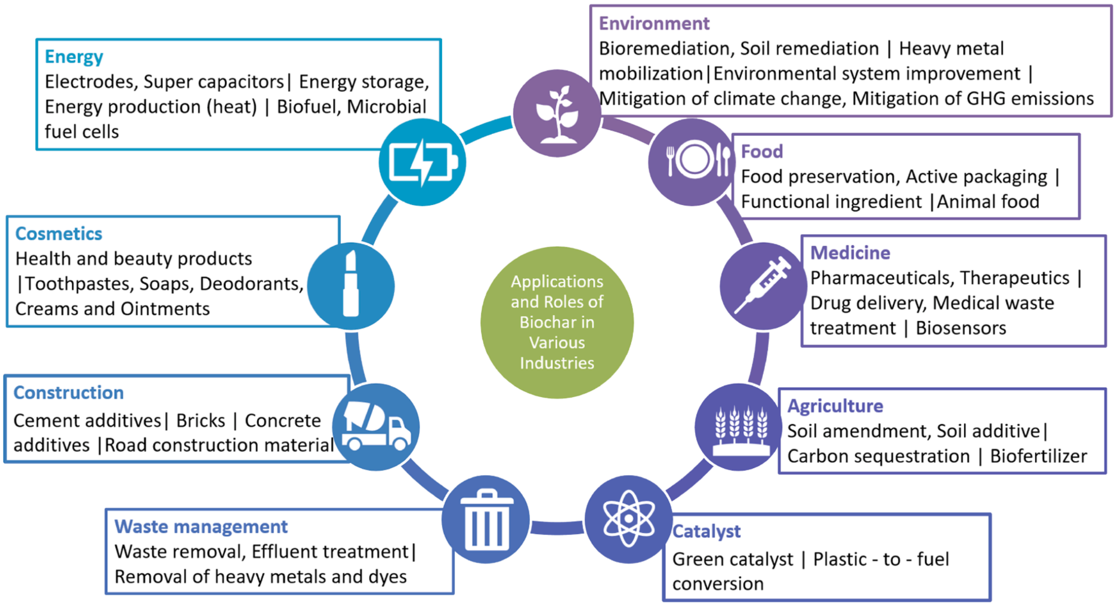

Fig. 1. Current commercial applications of industry produced biochar. Based on: [3].

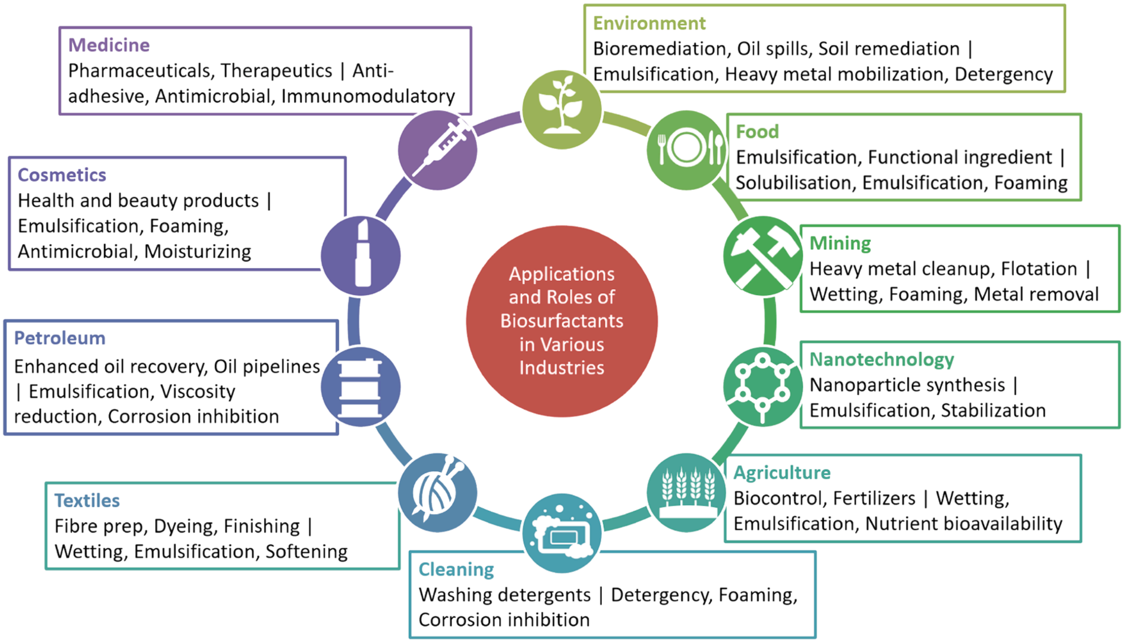

Fig. 2. Current commercial applications of industry produced biosurfactants. Based on: [5,6].

stable form. This procedure can sequester carbon over centuries, and there is a potential that it can be several millennia as well due to the longer half-life (∼5730 years) of carbon [12,13]. When applied in soil, biochar will continue to benefit the soil structure by improving soil fertility, pH levels, microbial activity and water retention [14–16]. Similarly, Bio-surfactant production from biomass also shares number of advantages and is often used as a product to treat petroleum hydrocarbon contaminated soils [17]. Despite the repeatedly mentioned advantages of both biochar and biosurfactants, their environmental impacts need to be carefully analyzed. These impacts occur at various stages of their life cycle, including production, transportation, application, and post-use. Each stage involves the use of energy, materials, and water, which can contribute to environmental burdens. Additionally, different types of outputs or emissions may result during application and after their use. A detailed assessment of these factors is essential to fully understand the overall sustainability of using biochar and biosurfactants (Fig. 3). The following subchapters (2.1 to 2.4) will be dedicated to the comparison and evaluation of the different life cycle phases of biochar and biosurfactant.

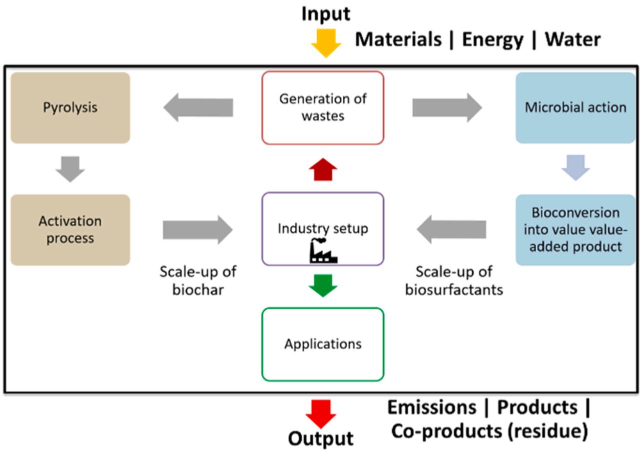

Fig. 3. Primary processes involved in biochar and biosurfactant production to application (transportation aspects are involved in the industry setup).

2.1. Production phase

In the biochar production phase, pretreatment of the selected biomass is usually carried out to sort, dry and grind the materials. This process is useful for the uniform decomposition of the biomaterials, thus should be done using electrical shredders. The power consumption of industrial shredders can vary from 103 kW to 420 kW, depending on the capacity [18]. However, energy consumption for pretreatment depends on many other factors such as dimensions required (finer the best to increase the surface area), time of shredding, type of biomass and quantity of production. When focusing on the activities taking place, biosurfactant production requires extensive pretreatment only when agro-industrial waste is used for the biosurfactant production. Moreover, most other waste types (dairy waste, municipal solid waste, glycerol waste, industrial waste) (even agro-industrial waste in some cases) used in the biosurfactant production is already in a shredded or semi/liquid phase [19]. In this case, it is safe to assume that energy requirements in the pretreatment stage are relatively lower in the biosurfactant production. Similarly, the economic impact of the pretreatment stage of the biomass (to produce biochar) must be higher than the biosurfactant production. This is due to the increase energy consumption, requirements of mechanical instruments with high-capacity installation or fossil fuel as the energy source. However, the feed material requirement for biochar production is seemingly lower than that of biosurfactants. For instance, Harsono et al. [20] mentioned in one of the case studies examined, it required 4800 tons of feed material (palm oil empty fruit bunches) per year to produce 960 tons of biochar [20]. Another study reveals that, when bagasse and paddy straw were used as the feed material, the expected biochar yield is 24–28 %, while for cocopeat it is around 46 % [21]. Mukherjee et al. [22] shows that biochar yield changes with the pyrolyzing temperature. Accordingly, authors mentioned a range of biochar yield (to produce 1 kg of biochar; 1.92–3.57 kg feed materials) that matches with the range of temperatures (300–600 °C). Here, the highest requirement of biomass is at the highest temperature and the lowest requirement at the lowest temperature [22]. Accordingly, it is possible to approximate that the feed material requirement for the biochar production is 2–5 kg per 1 kg of biochar yield.

In the case of biosurfactant, it is challenging to determine the feed material requirement in kilograms as the references give the yield per liter of feed materials. As an example, Karmainski et al. [23] explains, the yield of biosurfactants gathered from various manuscripts. Accordingly, the given yield for the corresponding biosurfactant (glycolipid titer) is 6.7 g/L, where feed material contains 1050 g/L of acetate concentration. Correspondently, the yield is reported to be 0.13 g biosurfactant/ g of acetate. Hence, to produce 1 kg of biosurfactants, 8 kg of acetate is needed [23]. Additionally, Qamar & Pacifico [24] explained rhamnolipids production from sugarcane bagasse has up to 9.1 g/L yield. In contrast, for sophorolipids from food waste has up to 115.2 g/L, and other biosurfactants have 2–5.5 g/L yield when feedstock such as banana stalks or orange peels were used [24]. Assuming that, substrate concentration to be around 20–250 g/L (based on the reported literature), the feed material requirement can be calculated to be from 2.2–10 kg per 1 kg of biosurfactant production. Further literature analyzed given in Table 1.

After pretreatment, the biomass will be dried in an oven to reduce the moisture level. This can be done even using sunlight, which lowers the energy intensity of the process. Though alternative sources used, these energy requirements are negligible when it comes to the pyrolyzing of biomass. Researchers showed that the pyrolyzing temperature can vary from 350 °C to 800 °C depending on the pyrolyzing method [3,14]. Accordingly, microwave-based pyrolyzing reactors require a lower temperature (350 °C). Nevertheless, it is identified that using microwave-based pyrolyzing limits the amount (around 1 ton/h) of the feed biomass due to the microwave penetration limitations [14]. Many scholars argue that, increase in pyrolyzing temperature enhances the quality of the biochar produced. As a matter of fact, many producers have used higher temperatures (600–700 °C) [51]. Moreover, biochar produced from high temperatures can be useful in acidic soils to improve their structure, fertility and sequestering carbon [52]. However, drawbacks can also be seen in using higher temperatures for biochar production. Firstly, the low yield of biochar; at 300 °C, the yield is identified to be 50–60 %, but at 500 °C it decreases to 31–46 % [48]. Secondly, higher temperatures negatively corelate to the hydroxyl and aromatic functional groups availability in biochar reducing their adsorption capacities. It is studied that; the FTIR bandwidth strength corresponds to hydroxyl and aromatic groups (3200–3400 cm^{-1} and 1580–1600 or 3050–3000 cm^{-1} respectively) significantly reduces as the temperature reaches beyond 600 °C. Moreover, bandwidth strengths increase when the temperature is lower than 500 °C [53]. Thirdly, higher energy use occurs when higher temperatures are accompanied. Upon the heating rates and pyrolyzing time, the energy requirement for the production stage of Biochar is very high. Considering the facts, Gahane et al. [25] assessed the total energy consumption for biomass-based biochar production can vary between 1.44 MJ to 2.16 MJ per 1 kg of Biochar production [25]. Similarly, Munoz et al. [26] assessed the total energy usage to produce and apply 1 kg of biochar in the soil is to be around 1.9 MJ to 4.4 MJ [26]. Hence, Values estimated by Gahane et al. is appropriately denoted to the production stage. Considering the energy densities of commonly used energy sources for biochar production, this can be calculated to be 0.047 L of diesel or 0.5 kWh of electricity requirement (M. Rashid [49,54]). However, recirculation of syngas can reduce the energy requirement of the process significantly. More practically, energy generation through syngas circulation can be introduced into the grid as compensation of the energy used to produce biochar. In some cases, it is reported that energy production can be up to 6.7 MJ per 1 kg of biochar production through syngas [26]. Hence, the overall production of biochar is energy positive.

Higher emissions produced by the manufacturing process are another environmental concern. Undoubtedly, biochar can sequester carbon for a long time. Quantitatively, this carbon sequestering can be up to 2.68 kg CO_2 equivalent per 1 kg of biochar production [31]. This can be 43–69 % of the total carbon availability from the starting raw materials [49]. Accordingly, the rest of the carbon and other products can be released along with syngas and through leaching (after adding to the soil), yielding hazardous effects to the environment. It is often mentioned that these contaminants highly depend on the feedstock and the conditions which are used for the pyrolysis [51]. Though emissions of CO_2 (in the use phase) are not considered contributing to the climate change effects as the origin is biogenic, emissions of other gases (i.e. CH_4) should be taken into consideration. Further, the potential greenhouse effect of those is higher than CO_2. A study carried out by Mohammadi et al. [37] demonstrates that emissions of CH_4 from 1 kg of biochar production can range between 10^{-5} kg to 2×10^{-3} kg, depending on the biochar production system and pyrolysis temperature. Additionally, production can result in N_2O and PM_{10} particle emissions of 10^{-3} kg and 7×10^{-4} kg per kg of biochar [37]. Similar values were reported by other authors as well [49]. Other than these, there are risks of polycyclic aromatic hydrocarbons (PAH) and dioxin emissions during the biochar production if parameters were not controlled appropriately. In some cases, it is reported that these emissions can be around 0.1–34 mg/kg of biochar, especially at low temperature pyrolyzed biochar [48].

Compared to biochar production, bio-surfactant production has a higher energy intensity, being bioreactor operation to be the most energy-intensive stage. Bippus et al. [55] reports that the bioreactor operation is responsible for 58 % of the total climate change impacts coming from the production stage [55]. Specifically, an aeration system which supplies oxygen to the bacterial systems, mechanical stirring to keep the particles in suspension and homogeneous, and temperature regulations for the cooling system and disinfection processes require

Table 1
Comparison of various production parameters in biochar and biosurfactant manufacturing with corresponding references.

Parameter	Unit	Biochar		Biosurfactant	
		Value ranges	**References**	**Value ranges**	**Notes & References**
Energy Consumption	MJ/kg	1.44 – 4.4	[25,26]	1173– 11,340	[27,28]
Energy Production	MJ/kg	1.00 – 6.70	[26]	Not Applicable	Inherent process characteristic [29]
Carbon Sequestering	Gt CO_2 eq/year	0.2 – 6.6	[30,31]	Not Applicable	No sequestration during production [32]
Crop Residue Sequestration	Pg C/year	0.36 – 0.72	[33]	Not Applicable	Feedstock-dependent, not sequestration [34,35]
CH_4 Emissions	kg/kg	0.002 – 0.00001	[36,37]	Not Applicable	Negligible amount released due to fermentation byproduct [38]
N_2O Emissions Reduction	tons/ha	20–50	[39–42]	Not Applicable	Use-phase benefit only [43]
N_2O Emissions	kg/kg	0.001	[37]	0.0005 - 0.002	Aerobic process emission [44]
PM_{10} Emissions	g/kg	0.7	[37,45,46]	0.09	[27]
PAH Emissions	kg	0.10 – 0.034	[47,48]	Not Detected	[28]
CO_2 Abatement	kg CO_2 eq.	1.47 ± 0.70	[31]	Not Applicable	–
Diesel Usage	L diesel/kg biochar	0.047	[49]	Not Applicable	–
GHG Emission Reduction	kg CO_2 eq./MT dry matter	2700	(Gudmundur [50])	Not Applicable	–

higher energy. Any significant change or disturbance in one of these factors can end up being the microorganisms to be killed and stop the production line. Moreover, authors often mentioned that the substrate used for the fermentation can also contribute to making the process energy-intensive due to the pretreatment stages involved [56]. Secondly, purification of produced biosurfactants can also require energy in centrifuging stages and distillation processes depending on the process used. Further, the distillation process can further increase the energy intensities than the mechanical or chemical (solvent extraction-like) separation methods. This is due to the continuous temperature regulation process required. Another important factor to be considered is the reaction time for bio-surfactant production. Frequent arguments show that the reaction time varies from 113 to 437 h depending on the quality of the bio-surfactant required. Here, longer hours of reaction often result in high-quality biosurfactant yield (especially manosylerythritol Lipids (MEL) types) [57]. It is imperative to think that by reducing the reaction time, the total energy demand can be reduced. Hence, better optimization of reaction parameters is often required for an effective system. Improving the quality of the substrate used (with lower impurity levels), it can lower the energy demand for the purification steps significantly (from 21 % to 12 %) [58]. Nevertheless, bio-surfactants are made from a biological system, so as the production is highly sensitive to external conditions, hence they should be treated carefully [57]. Schonhoff et al. [56] argued that Rhamnolipids (RL) based bio-surfactant production is environmentally impacting than MEL type bio-surfactant production. The main reason for this finding is that RL produces microorganisms (*Pseudomonas aeruginosa*) that are opportunistic pathogenic and obligatory aerobics. Hence, higher safety concerns and energy requirements are necessary for the maintenance of the biosystem. Also, RL production for specific volume of feedstock is lower than that of MEL. However, the results show that the substrate is not the major reason for the huge difference [56]. Quantitatively, RL production requires 9563–11,340 MJ/kg of biosurfactant production while MEL production requires only 1173–1429 MJ/kg of energy [28,58,59].

Gaseous emissions derived directly from bio-surfactant production are mainly CO_2. Since the substrates are biobased, the process emissions can be carbon neutral. Other volatile emissions are not reported [59]. However, pathogenic emissions that can be harmful to the ecosystem and human health can escape from the system if not properly managed. Moreover, cytotoxicity is often reported with RL-type biosurfactants (*Pseudomonas* and *Marinobacter* strains). Specially, when the concentration exceeds 0.5 mg/ml (often at $\leq$0.25 mg/ml), causing significant cell death for skin and liver cells of the human body. In contrast, industrial surfactants have significantly lower levels of cytotoxicity (<0.002 mg/ml) [60,61]. Additionally, some biosurfactants can increase the ultraviolet-B radiation (UVB) sensitivity of the skin by enhancing the chances of UVB-induced damage to the skin, although biosurfactants often produce a protective layer against oxidative DNA damage in the skin [61]. Nevertheless, these ecological hazards can be properly managed by concentration optimization to reduce cytotoxicity. Further, using safer bacterial or fungal strains (i.e. *Marniobacter*) or utilizing biosurfactant-producing microorganisms with phytoremediation techniques can reduce these risks [62]. Importantly, bio-surfactants are often advantageous than industrial surfactants. Mainly, due to the fact that industrial surfactants make non-biodegradable residuals and cancer-making intermediate products throughout the advanced oxidation stages [63,64]. Table 1 shows the list of literature aided for the emissions and energy requirements for biochar and biosurfactant along with the corresponding values.

2.2. Transportation phase

Unlike the production stage, the transport of biochar has always resulted in an environmental impact. This is due to the emissions related to fossil fuel usage, which cannot be compensated for. It can be understood that higher mass or higher distances in transport can increase the overall impact of the transportation stage. Moreover, the mode of transportation can also be a significant factor to consider, although road-based transportation is predominant [31,65,66]. Hence, it is reasonable to assume that the environmental impacts of locally produced biochar are less than those of transported biochar in a life cycle perspective. Moreover, efficient transportation techniques such as bulk density reduction can often result in improved transportation efficiency. For instance, literature shows that biochar density higher than 700 kg/m^3 can reduce the overall CO_2 emissions by 1–5 % [67,68]. As per the literature, transporting biochar and other products for 3 to 25 km can result in 1.6 to 8.6 kg CO_2/ton of feedstock emissions respectively [37,69,70]. However, changes in distances and material content make it difficult to have a clear idea of emissions related to the transportation of biochar only [37]. Desjardins et al. [71] studied the carbon footprint of total transportation of biochar for 522 km (collection of feedstock materials: 100 km and biochar transportation to destination: 422 km), has given an average emission of 6.55 kg CO_2 eq/kg of biochar [71–73]. Additionally, Massana et al. have revealed that 3 % (4.5 kg CO_2 eq. for 10 km) of CO_2 emissions from the entire lifecycle of biochar are due to the transportation [74]. This can be equivalent to 1.55×10^{-2} kg CO_2 eq./kg of biochar to be transported for a 1 km distance.

Additional environmental burdens from biochar transportation are also reported. It is estimated that biochar transportation is responsible for 21 % under human toxicity impact category, 20 % under freshwater eutrophication and 6 % under fossil fuel depletion compared to overall impacts given by the biochar production, transportation and application [26]. These impacts can arise mainly from fossil fuel usage. However, other sources are also identified. As examples, particle emission through wind erosion during transportation and heavy metal leaching due to temporary storage facilities can be given [48]. These emissions can be spread through micro or nano-level particles, contaminating soil or water sources or even humans and animals directly. This can occur largely when bulk biochar transportation. Hence, additional prevention measures are needed to limit and prevent carbon dust dispersion and contamination. It is estimated that during transportation and application, approximately 1–3 % of biochar is lost to the environment ([75]; Y [76]). Another prevention mechanism is to decentralize the biochar production minimizing long-term transportation, which can significantly reduce the impacts arising from the biochar transportation [77].

Biosurfactant transportation has a lower impact on the environment compared to its fossil fuel-based counterpart. This is mainly due to the elimination of transportation of fossil fuels, which always involves long distances. Another fact is that bio-surfactant production is largely decentralized and established where an abundance of organic waste (i.e. molasses or distillery products) can be found [78–80]. Accordingly, minimum transportation is required specially when raw material requirement considered. However, application sites can be away from the production facilities, which can elevate the impact related to transportation [17,81,82]. It is estimated that biosurfactants made from food waste can drop the environmental impacts by 10–25 % compared to the petrochemical-based ones. Further, 8 % reduction from this due to the transportation cuts [79].

Transporting liquid bio-surfactants poses a higher environmental risk than transporting powder products due to the possibility of spillage. However, biosurfactants are 100 % biodegradable, hence the threat to the environment is minimal. Nevertheless, threats to human safety or the safety of the organisms are still increased due to the possible availability of pathogenic microorganisms [83,84]. Moreover, if the powdered form is preferred, further transportation-related efficiency can be achieved by increasing the bulk density. Quantified environmental impact for biosurfactant transportation is not to be found in the literature. In addition, the impacts can be varied largely by the region, transportation medium and scale. Assuming truck-road-based transportation and the feedstock of food waste, emissions for 1 km transportation of biosurfactant can be estimated to be 10^{-1} to 3×10^{-4} kg CO_2 eq./kg of biosurfactant [79,83,84]. This was estimated through the contribution of the transportation sector in biosurfactant production and total emissions related to biosurfactant production reported in various literature. The estimated values are lower compared to the biochar-related transportation. This must be expected due to the short distances involved in the raw material acquisition and the lower yield of biosurfactant compared to the biochar [79,86]. These transport related data for both biochar and biosurfactant are summarized and compared in Table 2.

2.3. Use and post-use phase

Many authors cite the advantages of using biochar in soil conditioning due to various reasons. Mainly, it is reported that biochar can increase the fertility of the soil by enhancing its structure, adding more organic matter and reducing the mobility of many contaminants (heavy metals and organic contaminants) [51,66,92]. Further, authors demonstrated that it could reduce soil erosion and enhance soil biodiversity and microbial growth [65,66,93]. Another vital point reported is the emission reductions due to the biochar application. Accordingly, biochar can reduce 54–70 % of nitrous oxide and approximately 80 % of methane emissions due to the lowering of other soil fertilizer use [31, 52]. This has been quantified to be around 0.05 –0.07 kg of nitrogen-based fertilizer abatement per 1 kg of biochar application [37]. Specifically, this abatement is responsible for 0.05 kg of methane, 6×10^{-5} kg of di-nitrogen oxide and 2×10^{-4} kg of ammonia equivalent per kg of biochar application [49]. Ultimately, this can be equal to 1.36 kg CO_2 equivalent per kg of biochar (Chris [94]). Munoz et al. [26] also reported that application of 1 kg of biochar in soil can reduce the CO_2 emission by 2.6 kg on average, which is higher than the value reported by the previous manuscripts [26]. However, it is safe to report that, on average the CO_2 emission saving can be 1.3 to 2.6 kg CO_2 equivalent per 1 kg of biochar under optimistic assumptions.

Negative environmental implications were also reported from biochar-soil application. Primarily, researchers reported alterations of soil pH due to the biochar applications. High pH levels effects micronutrients (zinc, manganese and iron) of the soil, reducing their availability to plants [48]. The same authors also reported that higher alkalinity of the soil reduces soil fertility by 50 %. Though the soil pH alterations may depend on the soil conditions (e.g. buffer capacity) where the biochar is applied, major changes can occur when higher pyrolyzing temperatures (>500 °C) were used. This is mainly due to the availability of carbonates, hydroxides or oxides of magnesium, potassium or calcium in the biochar [95,96]. Moreover, it is often identified that these substances are abundant in the products if the feedstock used is poultry industry-based [97]. However, according to Xiang et al. [48], biochar made from poultry litter or with high pyrolyzing temperature in acidic soils can be beneficial in elevating the pH of the acidic soil. On the other hand, biochar may absorb an average of 65 % of plant available nutrients (nitrogen, potassium and phosphorus) due to its high adsorption capacity of the same. This may reduce the effective nutrient availability of the soil, advantageous in the long term but disadvantageous in the short-term crop production. Moreover, authors have also reported that the high bioavailability of arsenic in the soil that conditioned by the biochar. This is reported to be a 101.6 % increase in arsenic if straw-based feed materials have been used in biochar production [48]. Further, Wang et al. [98] reported the toxicity increase related to polyaromatic hydrocarbons (PAH) and environmentally persistent free radicals (EPFRs) due to the biochar addition. It is identified that biochar applied soils can have 200- 1000 μg of PAHs in 1 kg of soil, which is significantly high enough to affect the plant germination and microbial community (J [48,99]). Additionally, EPFRs species, such as reactive oxygen species, can reduce plant growth by damaging cell structure. Moreover, it is also reported that EPFRs species can enhance the microbial mortality rate by 50 %. Other than this, volatile organic carbon (VOC) can also be detected among the gaseous emissions in the long term in the biochar added soils. However, in monitored or certified biochar VOC can range from 0.9–1.2 mg/kg of biochar. In non-controlled systems this can be higher (1.4–3.8 mg/kg) [75]. These use and post-use related data for biochar are summarized in Table 3.

Examining the biosurfactants usage in soil remediation applications, it is understood that it has been frequently used in remediating oil (petroleum hydrocarbon) spills or metal-contaminated soil. Scholars argue that it is an environmentally friendly approach as the biosurfactants show minimal toxicity for aquatic and soil ecosystems [81, 109]. A few researchers discuss that biosurfactants detach the hydrocarbons from the soil particles, enhancing the microbial decomposition of the same, aiding immediate bioremediation [82,110]. For an instant, it can remove 15–30 % more oil from contaminated sites than the conventional methods and 90–100 % of contaminated metals (lead, chromium and zinc). Moreover, as the used biosurfactants degrade naturally within 1–3 weeks, it reduces pollution in the long run [81]. Also, it is reported that due to the antimicrobial properties of some biosurfactants, the remediated soil can be used for agricultural purposes without using pesticides in it, which can be an economic benefit as well [110]. However, biosurfactants enhance the diversity of micro-organisms in the soil or the water when applied. Primarily this is because, increase of the solubility of hydrocarbons in the soil increase the nutrient availability [111]. This can be affected negatively if not monitored as the excess microbial community can produce toxic by-products or can cause clogging's of reservoirs.

Apparently, quantification of the impacts of biosurfactant usage is difficult as it can be vary significantly based on the nature of the contamination, soil type or structure, and biosurfactant to be used. Nevertheless, some implications can be identified in the recent

Table 3
Various use and post/use phase emission parameters of biochar with corresponding references.

Parameter	Units	Value ranges	References
CO_2 Abatement	t CO_2 eq per hectar	0.05–0.07	[37, 100–106]
PAH Emissions	kg	0.0002- 0.001	(D [48,98])
VOC Emissions Controlled	kg	0.90-0.0000012	[75]
VOC Emission uncontrolled	kg	1.40-0.0000038	[75]
Inorganic nitrogen fertilizer (Urea) abatement	kg	3.35	[107]
Inorganic phosphorus fertilizer (P_2O_5) abatement	kg	0.06	[25,73]
Inorganic potassium fertilizer (K_2O) abatement	kg	0.03	[26,108]
Compost treatment minimization	kg	15.415	[25,37]

Table 2
Comparison of various emission parameters of biochar and biosurfactant transportation with corresponding references.

Parameter	Unit	Biochar		Biosurfactant	
		Value ranges	**References**	**Value ranges**	**References**
CO_2 Emissions	kg CO_2 eq.	0.0155–0.45	[71,74]	0.1- 0.0003	Calculated [79,83,85]
% loss of biochar during transport/application	%	1–3	(Aller, 2016b; [75,87]; Z [88])	Not Applicable	Not Applicable
GHG Emission	kg CO_2 eq.	0.01–0.04	(Aller, 2016c; [37,89])	Not Quantified	[90]
Transportation cost	dollars per mile	0.10- 0.50	[91]	0.15–0.75	Calculated [79,83,85]

literature. Mulligan [43] shows that in-situ remediation of soil using biosurfactant avoids the soil transportation and disposal off the site, avoiding up to 8–12 tCO_2 eq. per hectare of land [43]. However, this avoidance is due to the non-presence of transportation, hence not a strong evidence accounts for the application phase. In contrast, Pacwa-Płociniczak [112] reports the case of hydrocarbon reduction in the contaminated soils by 35–68 % using biosurfactants. It is shown that the application avoided 1.5–3.2 tCO_2 eq. per hectare emissions by eliminating long-term leaching without adding any other impacts to the soil system [112]. This can be converted to 0.5 – 3.2 kg CO_2 eq./kg of biosurfactant, considering the dose of biosurfactant per hectare can range between 1–3 tons/ha (Özlem [11]). A few manuscripts report that the toxicity eliminations and biodegradability benefits of biosurfactants when applied in soil remediations. Accordingly, biosurfactants can eliminate 0.22–6.11 kg CO_2 eq. of impacts per kg of biosurfactants that otherwise occurred for the aquatic environment if synthetic surfactants were used. Similarly, the biodegradability of biosurfactants can eliminate 0.9–1.7 kg of CO_2 eq. impacts that would occur through synthetic surfactant accumulation [43,113]. These use and post-use related data for biosurfactants are summarized in Table 4.

Table 4
Various use and post/use phase emission parameters of biosurfactants with corresponding references.

Parameter	Units	Value ranges	References
CO_2 Abatement	tCO_2-eq/ha	8.00–12.00	[43]
Metal removal % (Pb, Cr, Zn)	%	90–100	[113]
Avoided leaching reduction	tCO_2-eq/ha	1.5–3.2	[112]
toxicity avoidance	Kg CO_2-eq/ha	0.22–6.11	[113]
biodegradability	kg CO_2-eq	0.9–1.7	[43]

2.4. Mass flow analysis based on application dosages

As per the analysis made so far environmental effects of the materials (biochar or biosurfactants) addition in soils are highly sensitive to the mass of biochar or biosurfactants that is being produced, transported or added to the soil. However, the amount of the production or the transportation depend on the application dosage of the considered material. Hence, it is important to understand the relevant biochar and biosurfactant soil application dosages in order to evaluate the overall effect of the two products under one functional unit. Further explaining, it is not correct to compare 1 kg of biochar with 1 kg of biosurfactant as the actual application dosage may vary. For instant, what 1 kg of biochar does to contaminated soil maybe done by much less mass of biosurfactant. Hence, it is vital to understand the actual application dosage and back calculate the mass required from each material.

In the case of biochar, it is often recommended to use smaller dosages in contaminated or agricultural sites at the beginning and to monitor the effect of biochar over the time. It is recommended to change the dosages after this monitoring of the soil conditions if required. Often, this smaller dosage refers to 1–3 tons of biochar per hectare of land. However, these dosages are highly dependent on various factors such as soil type, economic nature and the environmental concerns and also nature of the land (contaminated or agricultural). Moreover, different feedstocks used in biochar production can also affect the dosage determination (Jonathan [114,115]).

In most cases of agricultural lands, the biochar dosage is set to be in between 2.5–25 tons per hectare. It is reported that these limits balance the economic aspects and the soil conditioning efficiencies effectively (Jonathan [114,116]). For instant, Yang et al. [116] reported that sandy loam soils in semiarid regions required 15 tons/ha of biochar application to improve the nutrient availability and fertilizer efficiency [116]. In contrast, for tobacco cultivation low rate (900 kg/ha) of biochar has been applied for optimum growth, while higher rates negatively affected the plant health and the soil nutrient availability [115]. Alternatively, in contamination site remediation, the dosage can be determined based on the contaminated pollutant, biochar characterization and the soil pH levels. For metal-contaminated sites, often 5–10 % of biochar by weight is recommended for effective remediation. It is studied that, this dosage immobilizes the cadmium, copper and zinc in acidic soil types [97,117]. In another study, it is mentioned that chromium and arsenic can be immobilized in the contaminated site in the long term by adding lower rates (0.3 % by weight) of biochar through chemical adsorption and redox reactions [118]. Nevertheless, organic pollutant-contaminated soil required lower rates of biochar applications. As an example, Gao et al. [119] reported that 3–5 % of biochar by weight is efficient in removing 90 % of atrazine pesticides from the contaminated site. It is worth to note that the researchers used iron-enhanced modified biochar for the treatment mechanism [119]. Another study carried out by Xiang et al. [120] mentioned that, only 1–2 % of wood-derived biochar by weight is enough to treat the contaminated site. Specifically, for hydrocarbon-contaminated soils treatment using pollutant-degrading bacteria in the void spaces available in the applied biochar [120]. Moreover, alkaline biochar is mostly required in lower rates (5 % by weight) due to the effective pH raising ability [117,97]. Given the ranges suggested by various articles, the safe range of biochar application can be identified as 1–10 % by weight. This can be calculated as 10–100 tons of biochar requirement per hectare of land.

In the case of biosurfactants, the dosages are highly dependent on the nature and the content of the contamination, the properties or the structure of the soil and the properties of the biosurfactants to be used. For instance, Zenginyurek [11] explained that higher biosurfactant dosages (>1000 ppm of rhamnolipids) are required for soils contaminated with hydrophobic pesticides. This is mainly due to the elevated levels of bioavailability needed for the treatment of such soil systems (Özlem [11]). Bustamente et al. [113] reported that highly porous soil systems often require 20–30 % higher dosage of biosurfactants than the lower porosity soil systems. Authors have shown that this is due to the lower sorption's reported by lower porosity soils. Moreover, authors also reported that the critical micelle concentration (CMC) value of the biosurfactant is often useful for the dose optimization. Mainly lower CMC values denote that the biosurfactant is effective at lower concentrations and vice versa. In addition, above the CMC values (>45 mg/L for surfactin), the biosurfactant can solubilize the hydrophobic contaminants, increasing the availability for the degradation [113]. Other than these, general factors like cost of the products, toxicity limits and mixing strategies can also affect the effective dose of the biosurfactant to be used in the contamination sites [121].

Considering the specific cases of biosurfactant applications, 1.6–1000 ppm (mg/kg of soil) is often reported when rhamnolipids were utilized. For instance, 13 % extra removal was observed when rhamnolipids were applied to remove trifluralin (a hydrophobic soil pollutant) with high dosages. In the case of crude oil removal, efficiency around 80 % was reported when similar dosages of rhamnolipids were used [113,121]. In acidic soils, endosulfan contaminants removal was experimented with limited degradation results at 0.98–195 ppm of sophorolipids dosage (Özlem [11]). Moreover, hydrocarbon removal from the contaminated soils were achieved with promising results when dosage concentrations exceeded CMC limits (45–50 mg/L) of the sophorolipids biosurfactants [113]. Alternatively, around a 60 % reduction of octane in the contaminated site was observed when *Lactobacillus* biosurfactants were used. However, the dosage was not clearly mentioned in the study [121]. These values can be converted to "tons/ha" assuming the soil bulk density of 1.8 kg/m^3 and the average contaminated depth to be 0.2 m. The calculation shows, 1–3 tons/ha will be the optimum dosage range to be used in contaminated sites [121].

Table 5 summarises the material flow, emissions and energy requirements under different phases of the biochar and biosurfactant life cycle, calculated to treat 1 hectare of contaminated land. Energy and emissions per 1 kg of materials were multiplied by the dosage identified in the Section 2.4 to calculate the energy and emissions per 1 hectare of

Table 5
Calculated values for main parameters under different life cycle stages of biochar and biosurfactant, focusing on the functional unit of 1 hectare of land contaminated soil remediation.

Life cycle phase	Parameter	Unit	Biochar performance (±Standard Deviation)	Biosurfactant performance (±Standard Deviation)
Production phase	Feed material requirement	ton/ha	260 ± 139	16 ± 8
	CH_4	ton/ha	100 ± 57	NA
	N_2O	ton/ha	55 ± 26	NA
	PM10	ton/ha	39 ± 18	NA
	PAH	ton/ha	1.7 ± 1	NA
	CO_2 abatement	ton/ha	1.5 ± 0.7	NA
	Energy consumption	GJ/ha	79.2 ± 37.4	2340 ± 677.23
	Energy Production	GJ/ha	368.25 ± 173.9	NA
Transport	CO_2	ton/ha	12.8 ± 6.1	0.15 ± 0.08
Use and Post use	CO_2 abatement	ton/ha	136.5 ± 71.3	15.5 ± 2.33
	PAH	ton/ha	0.3	NA
	VOC	ton/ha	0.19	NA

*All the values mentioned here are calculated based on Table 1–4 as the methodology explained in Section 2.4.
**Standard deviation calculated based on uniform distribution.
***NA stands for not applicable.

land. The standard deviation of each average value was calculated assuming a uniform distribution of values. Normal distribution was not possible to use as the ranges denote minimum and maximum values only. Premathilake et al. [122] discusses the standard deviation calculation using maximum and minimum values using uniform distribution, which is adapted here in this article [122,123].

Fig. 4 shows the mass flow diagram built using Stan 2.7 software based on the findings in the Table 4 in comparative form. It is possible to understand that the biochar shows a higher environmental degradation than biosurfactant to treat one hectare of contaminated land, considering the emissions. This is mainly due to the low biosurfactant material requirements to treat the land even it requires higher energy and consumption per kg of material than biochar. Moreover, the number of emissions (CH_4, N_2O, PM_{10}, VOC, PAH) made by biochar pre and post use is also a crucial factor. However, the energy consumption of biosurfactant production is extremely higher than the biochar, which can be a negative point in biosurfactant usage. It is important to note that, biochar produces a net energy out, while biosurfactant required net energy input. Accordingly, biochar has a negative energy requirement, while biosurfactant has a positive requirement. Moreover, Energy requirement in biosurfactant is 10 times more than what biochar produces. Since the energy generation has the highest impact in the environment, impacts of biosurfactant is too high compared to that of biochar. This makes biochar-based treatment environmentally logical than using biosurfactant for the same reason.

3. Economic implications of biochar and biosurfactants considering different life stages

Economic impacts of biochar are highly sensitive to the selling price. Other than that, it can also be dependent on the types of feedstocks used, pyrolysis technology and the applications. Evidence in the literature suggests that on average, the selling price of 1 kg of biochar can vary from 0.44 $ to 0.98 $, with the cost of the feedstock materials in the range of $ 0.12–0.15/ kg of biochar [14,52]. In some cases, using agricultural waste (including digestates) as the feedstock and utilizing cheaper pyrolysis technologies often resulted in a broader selling price range from $ 0.01 to $3.5 per kg of biochar [26,93]. In contrast, modified or activated biochar with higher quality elevates the price per kg than the average values. This is mainly due to the pre- and post-processing needs, processing or monitoring for quality standards (microbial availability) and proper packaging requirements [48,77]. Surplus energy generation is also elevating the economic benefits of biochar production. Many scholars report that some systems are able to supply 68.7 % of their energy requirements and extra heat supplies for external grids [31].

The addition of biochar increases the profitability of the agricultural lands through the increase in crop yield. Further, it reduces the overall cost of irrigation as biochar helps soil to retain more water while enriching the soil fertility through the increase of soil biodiversity [31, 65,66,93]. Kuppusamy et al. [52] reported that crop yield can be increased up to 7–320 % depending on the application range of 2.5–50 t/ha [52]. Hence, users can increase the economic benefits through the increased yield and reduction of the operational costs per kg of biochar used. In addition to that, economic benefits can be expected through carbon sequestration, which is related to carbon removal certificates. It is identified that an economic benefit of $110 can be compensated for the avoidance of 1 ton of CO_2 equivalent [31]. This benefit can be converted into $ 0.143 - 0.286 /kg of biochar used with the values identified in Section 2.

Transportation can also be an important factor that increases the economic efficiency of biochar. Best practice would be local production of biochar eliminating transportation which could otherwise elevate the costs specially incurred if longer distances are involved. Moreover, lower bulk density (250–350 kg/m^3) can elevate the costs as it increases the frequency of transportation [14,31,66,93]. Ultimately, the investments for industrial quality biochar production plants can be higher. However, recent findings show a positive net present value (NPV) and returns of investments up to 8 % in biochar production facilities where raw materials are abundant or fertilizer supplies are disrupted [14,77]. Considering all these factors, the economic benefits of biochar can be calculated to be $ 0.433 – 1.156 per kg of biochar. It is important to note that the facility installation costs, and other operational costs should be taken into consideration when developing the profit margins. Here it is shown only the potential economic benefits arising from 1 kg of biochar application which calculated through limited data availability.

Considering the biosurfactants' life stages, producing the same has a considerable cost compared to its synthetic counterpart lowering its overall economic benefits. It is identified that 30–50 % of the cost is associated with the raw material consumption, which can be reduced to 10–30 % if agro-industrial waste is used [17,19]. A major part of the cost of production is accounted for fermentation and purification, where higher energy intensity and controlling is used. It is estimated that 60–80 % is often involved with these two production stages [109,110, 124]. Schonhoff et al. [59] reported that large-scale rhamnolipids-based biosurfactant production can cost $ 5–10 /kg while mannosylerythritol lipids cost $ 3–6 /kg [28]. However, Akbari et al. [17] estimated that the cost of biosurfactant production generally results in the range of $ 2–5 /kg, which is costlier than the synthetic surfactants ($ 1–4 /kg) [17]. In contrast, the selling prices of the biosurfactants depend on the raw materials, method, and the scale of production. Quality products with traditional production methods have the highest selling price recorded at around $ 34/kg [125]. However, optimization (46 % glucose syrup or corn steep liquor as the substrate) and large-scale production (over 8000 MT/year) can reduce the cost of production drastically, having the selling price per kg as low as $ 6.2 ([126]; Neil [127]). Common biosurfactants such as rhamnolipids and sophorolipids in commercial production lines with advanced strain improvements and process optimizations report much realistic selling prices in the range of $ 2.5 –

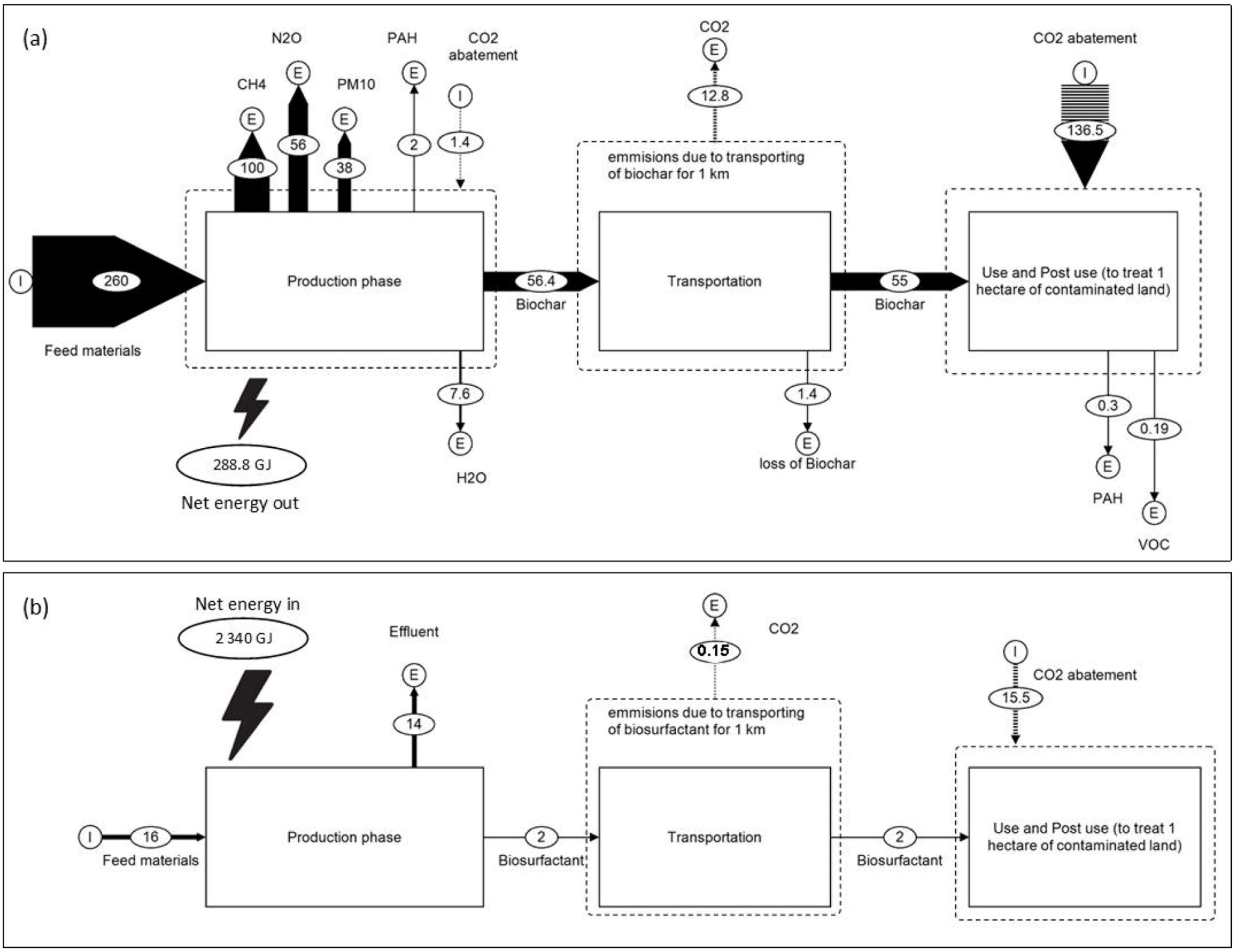

Fig. 4. Material and emissions flow with energy consumptions to treat 1 hectare of contaminated land under different life cycle stages using (a) biochar or (b) biosurfactant (*assuming 0.15 m depth contamination in the land with density of 1.8 g/cm^3. **Thickness of the arrows represent the size of the flow in each case. ***All the values are provided in tons unless it is mentioned. Based on values in Table 4).

5/kg which can compete with their synthetic counterparts (Simon [128]).

Transportation significantly cut down the cost of biosurfactant use. Mainly, articles encourage the use of in-situ biosurfactant production to reduce the cost of longer transportation [81,82,110]. Some researchers show that $ 0.5–2 /kg savings can be achieved using in situ biosurfactants compared to synthetic ones [82,124]. Moreover, it is identified that the long-term impacts of biosurfactants on the soils were nil, suggesting that 20–30 % savings in long-term chemical usage [17,124]. Furthermore, remediated soil systems can be successfully utilized for agricultural uses, further enhancing the economic benefits of biosurfactant usage [81,82,85]. These economic savings can vary from $ 1–4 /kg of biosurfactant, as literature suggests. Considering the cost, the selling prices and the potential savings, the overall economic benefit of using biosurfactant can be calculated to be varied from $ −1.5 to 7 /kg of biosurfactant. This shows that there is a possibility that the overall benefits can be negative given a cost instead of a benefit in the worst case. However, the values used here are taken from multiple literature and calculated based on assumptions. Hence, it can be deviated from the true values.

Fig. 5 shows the comparison of average costs and overall benefits to treat 1 hectare of contaminated land by using biochar or biosurfactants. For the calculations, biochar and biosurfactant consumption for the treatment of 1 hectare of land was considered as identified in Section 2. Later, costs and incomes identified for 1 kg of materials were multiplied by the mass of materials needed for the treatment of 1 hectare of land. The figure also denotes the standard deviation for each category, calculated assuming the uniform distribution. As per the figure, using biochar is seemingly beneficial than using of biosurfactant for the soil remediation in contaminated sites.

4. Social insight on biochar and biosurfactant applications

Literature that discusses about social implications specifically by site remediation application of biochar or biosurfactants, is severely lacking. However, a few articles managed to point out several social implications of biochar and biosurfactants-related applications, which have been discussed below.

Mohammed et al. [129] show multiple social impacts resulting from biochar in the sub-Saharan African context. In the article, it is reported that food security can be increased substantially using biochar as a soil amendment for agricultural land preparation. It is observed that a 15–25 % crop yield enhances the life of small farm holders in the area. Furthermore, the adoption of biochar-related small-scale industries has emerged as a powerful catalyst for advancing gender equity. By utilizing accessible, decentralized technologies such as dual-chamber pyrolysis reactors—designed to operate efficiently in field or household settings with minimal technical expertise—these initiatives break down

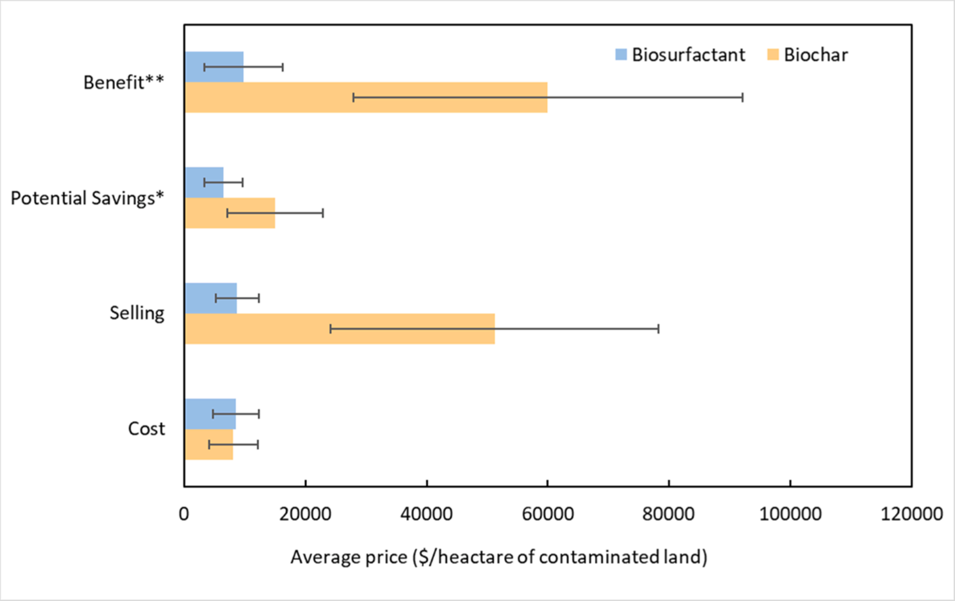

Fig. 5. Average costs and benefits of using biochar or biosurfactant to treat 1 hectare of contaminated land. (*potential savings denote the possible income from carbon sequestering, additional crop production or yield, and avoided fertilizer use. **benefit denotes the net economic income, which is calculated through the difference between the costs and incomes. Based on the values calculated throughout the Section 3).

traditional barriers to participation [15]. Also, it is observed 40 % increase in women's participation in briquette production in Northern Ghana, where income was increased, reducing the poverty gap. Moreover, the use of biochar briquettes in the area replaces the costly and polluting charcoal-based energy sources, reducing the cost of energy by 20–35 % [129]. The simplicity of biochar production systems, which require low upfront investment and basic operational know-how, creates opportunities for women to engage in sustainable entrepreneurship, skill development, and decision-making roles within their communities. This shift not only empowers women economically but also fosters inclusive growth by redistributing productive resources and amplifying their voices in climate-smart solutions [15]. However, on the social cost side, a pilot project undertaken in Côte d'Ivoire shows that industry-scale biochar production plants and plantation arrangements can displace 10–15 % of communities from the land-scarce regions [129, 130]. Further, literature shows that there is a reportingly high increase (5–10 %) of respiratory diseases among populations living close to biochar production sites due to the dust emissions [130]. Similar problems can arise when transporting low bulk density biochar due to particle emissions. It is reported that the final price of biochar can be increased by 25–40 % if the bulk density is lower, reducing the affordability of low-income populations (Benjamin D [131]). Moreover, many non-governmental organizations (NGOs) argue that the use of biochar in the lands poses risks of soil commodification, which harms indigenous land rights in the long term [130].

In the case of biosurfactants, similar social acceptance can be seen. For an instant, rural sugar processing areas of India and Brazil witnessed a 20 % increase in the employment rate when the industries started to produce biosurfactants using sugar industry wastes (molasses). However, 70 % of the jobs were seasonal [5,132]. In another case, despite the existing training gaps, biorefineries in the European Union, increased the employment for skilled labourers (fermentation technicians) by 12–18 % [28]. Moreover, health benefits were also observed. Communities close to detergent factories that used biosurfactants instead of synthetic surfactants reduced the respiratory-related disease by 8–12 %. In addition, the use of biosurfactants reduced the water pollution-related diseases by 12–18 % in rural areas in Brazil, enhancing the quality of life of the nearby populations [5]. In transporting of biosurfactant, it is studied that the toxicity-related impacts that occur through spillage can be reduced by 30 %, implying a safe working environment for workers [6]. Further, transporting related costs can be reduced drastically, and the employment rate can be increased substantially if the biosurfactants are produced locally as per some researchers [28]. On the negative side, the main concern was the manufacturing cost of biosurfactants, which is up to 50 times than its synthetic counterparts. This seriously limits the market penetration (less than 5%) of biosurfactants in low-income sub-Saharan Africa [5]. Moreover, having many patents for Europe and North American countries in biosurfactant production increases the imported product prices of biosurfactants in the lower-income countries up to 15–20 % [28].

5. Conclusions

The use of biochar and biosurfactants in contaminated land remediation presents distinct trade-offs across environmental, economic, and social dimensions. The manuscript analyzed per hectare of land application of biochar and biosurfactant rather than comparing directly 1 to 1 kg of material production. Biochar exhibits higher production emissions per hectare—approximately 100 tons of CH_4, 55 tons of N_2O, 38 tons of PM_{10}, and 1.7 tons of PAHs—while biosurfactant production generates negligible direct emissions. However, biosurfactants demand significantly more energy (∼2340 GJ) compared to biochar (∼80 GJ), which can even contribute surplus energy to the grid. Transport-related emissions are also higher for biochar due to its greater material volume needed to treat 1 hectare of land. During application, biochar continues to emit pollutants (0.015-ton PAHs and 1.99-ton VOCs per hectare), whereas biosurfactants remain emission-free. Considering the material flow, using of biosurfactant is beneficial to the environment. However, energy consumption and production play a vital role for the decision making. When powered by renewable energy, biosurfactants offer superior environmental performance; otherwise, biochar is the more energy-efficient and sustainable option though emissions are present.

Economically, biochar currently delivers stronger returns (≈ $60,000 per ha) compared with biosurfactants (≈ $9750 per ha), while biosurfactants provide added value through enhanced biodegradation rather than mere sequestration. Socially, biochar can improve food security and energy access, whereas biosurfactants support rural employment and safer working conditions but remain limited by high costs and patents. Policy measures—including incentives for low-energy biosurfactant production, R&D investments, minimum quality standards and fair-pricing regulations for biochar, and patent easing and technology transfer for biosurfactants—will be essential to enhance economic feasibility and ensure equitable sustainability outcomes.

Finally, the synergistic use of biochar and biosurfactants remains underexplored. Given their complementary mechanisms, future research should focus on quantifying the combined environmental, economic, and social impacts of integrated application strategies.

CRediT authorship contribution statement

Dilshan Sandaruwan Premathilake: Conceptualization, Formal analysis, Investigation, Methodology, Visualization, Writing – original draft, Writing – review & editing. **Arumadura Aruna Manoj Prasad Silva:** Conceptualization, Formal analysis, Visualization, Writing – review & editing. **W.A.M.A.N. Illankoon:** Conceptualization, Validation, Writing – review & editing. **Asia Rosatelli:** Validation, Visualization. **Fabrizio Beltrametti:** Project administration, Resources. **Adriana Bava:** Funding acquisition, Project administration. **Francesco Battaglia:** Funding acquisition, Project administration, Resources. **Marco Mendola:** Funding acquisition, Project administration, Resources. **Filippo Passaro:** Project administration, Resources. **Alessandro Abbà:** Supervision, Validation, Writing – review & editing. **Andrea Franzetti:** Resources, Supervision, Writing – review & editing. **Mentore Vaccari:** Project administration, Resources, Supervision, Writing – review & editing.

Declaration of competing interest

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

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Data availability

Data is already given in the manuscript

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