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Comparative sustainability assessments for integrated cassava starch wastes biorefineries

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A B S T R A C T

Sustainable development in cassava starch industries is hampered by high cost and environmental burdens associated with the business-as-usual (BAU) waste management strategies. In BAU, starch wastewater & bagasse wastes are anaerobically digested to produce biogas for starch drying with the digestate getting disposed into watercourses while the cassava stalks are burnt. Converting the wastes into high-value bio-products in an integrated cassava wastes biorefinery (CWB) could enhance the economic exploitation while reducing the environmental burdens of the wastes. Five simulated CWBs and the BAUhave been assessed and compared using simulations in SimaPro and a percentage sustainability index (PSI) estimation tool to identify product integration schemes that support the development of sustainable CWBs. The CWB scenarios included (I) combined heat & power, with (II) hexose-bioethanol, (III) pentose & hexose-bioethanol, (IV) pentose-bioethanol + glucose syrup, and (V) pentose-bioethanol + succinic acid. The environmental impacts generally increased with the number of product integrations within the biorefinery gate boundaries. However, accounting for avoided emissions from the corresponding fossil-products, the CWBs show higher emission savings than the BAU. The PSIs for the CWBs show that scenarios (I)-(II) favour the economic dimension over the environment dimension and vice versa for scenarios (III)-(V) and the BAU. Based on the substantial net power (~148–363 kW h/1 t feedstock) and fossil emission reduction potentials, implementation of green power tariffs could enhance the economic dimension for near-term applications of the CWBs. Thus, the CWBs should be explored for their potential to enhance sustainable industrial developments in cassava starch industries.

1. Introduction

Growing demands for cassava (*Manihot esculentum*) starch in industrial applications (e.g. pharmaceuticals, food) resulted in expanded cassava cultivation (~292 million t/a) and starch processing (FAOSTAT, 2019). However, high costs and environmental burdens associated with waste management and energy sources in cassava starch industries (CSI) hamper sustainable industrial developments (Hansupalak et al., 2016; Kleih et al., 2013; Zhang et al., 2016). The cassava starch facilities (CSF) generate large amounts of cassava starch wastewater (CWW) and cassava bagasse waste (CB), with respective generation capacities at 12–20 m^3 and 1.4 t (35–40% moisture) per t starch produced (Chavalparit and Ongwandee, 2009). Crop harvesting also generates woody cassava stalks (CS) estimated at 51% of the cassava roots by mass, which are mainly designated as wastes with only 10–20% used as planting materials (Zhu et al., 2015). In the current wastes management scheme [business-as-usual (BAU)], the CB and CWW are anaerobically digested to generate biogas for producing starch drying hot air (SDHA), followed by disposal of the digestate into watercourses (Tran et al., 2015). The CS are openly burnt in the farms (Zhu et al., 2015). Electricity for the CSF operations and waste treatment (~90–260 kW h/t starch) is fossil-based (Sriroth et al., 2000; Tran et al., 2015). Consequently, high water & land pollution, carbon footprints, and waste treatment & energy costs pose limitations to sustainable developments in the CSIs (Hansupalak et al., 2016; Kleih et al., 2013; Zhang et al., 2016).

Substitution of fossil-based energies & products with biomass-based alternatives is accelerating sustainable low carbon economies based on the promising economic and environmental benefits revealed in previous reports (E4tech et al., 2015; IEA, 2013). Use of the edible cassava roots as feedstock for bioenergy production [first

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Acronyms/Abbreviations

1G	First generation biorefineries
2G	Second generation biorefineries
AD	anaerobic digestion
BAU	business-as-usual
C5	pentose sugars
C6	hexose sugars
CB	cassava bagasse
CE	circular economy
CHP	combined heat and power
CS	cassava stalks
CSF	cassava starch facilities
CSI	cassava starch industries
CSL:	corn steep liquor
CWB	cassava starch wastes biorefineries
CWW	cassava starch wastewater
DAP	diammonium phosphate
DM	dry mass
EH	enzymatic hydrolysis
eLCA	environmental life cycle assessments EtOH: ethanol
FEP	freshwater eutrophication potential
FETP	freshwater ecotoxicity potential
FRSP	fossil resource scarcity potential
FU	functional unit (a biorefinery that converts 1-ton combined feedstock: comprising (mass basis) 45.2% CWW, 0.9% CB, 53.9% CS)
GHG	greenhouse gases
GS	glucose syrup
GWP	global warming potential
HTP	human toxicity potential
LCC	life cycle costing
LCSA	life cycle sustainability assessments
PSI	percentage sustainability index
SA	succinic acid
SDHA	starch drying hot air
sLCA	social life cycle assessments
TAP	terrestrial acidification potential
TBL:	Triple bottom line (economic, environmental, social)
TETP	terrestrial ecotoxicity potential
TS	total solids

generation (1G) bioenergy] has been advocated because of high yields of starch (Hanif et al., 2017; Okudoh et al., 2014). Various studies have shown environmental benefits of the cassava-based 1G bioethanol vs. the fossil alternatives (Hanif et al., 2017; Leng et al., 2008). However, because of the prevalent food uses for the cassava root starch (Howeler et al., 2013), the use of the cassava residues as feedstock for second generation (2G) bioenergy is preferable for sustainable co-production of food (starch) and bioenergy, especially for regions with declining arable lands (Kim and Dale, 2004). Laboratory and pilot demonstrations have shown possibilities for converting the CWW, CB & CS to high-value bioproducts such as bioethanol, succinic acid, glucose syrup and combined heat and power (CHP) (Knight, 2011; Li et al., 2017; Zhang et al., 2016). Thus, there is potential for the development of industrial-scale multi-product biorefineries for integration into cassava starch processing, potentially enhancing economic exploitations of the wastes and industrial developments in the CSIs. Sustainability of industrial developments calls for potential maximum transdisciplinary value extraction from the applied resource's entire life cycle, including economic, environmental and social values (Geissdoerfer et al., 2018). Hence, based on the various aspects of the sustainability concept, the idea that biorefineries are fundamentally sustainable, due to the renewability and environmental savings (CO_2 sequestration) potentials of the biomass feedstock, is subject for debate (Hofer and Bigorra, 2008; Pfau et al., 2014).

The sustainability concept, therefore, promotes developments having three-dimensional fundamental stability (3D)- economic, environmental and social, termed Triple Bottom Line (TBL) sustainability (Agrawal and Singh, 2019; Parada et al., 2017). However, separate and different assessment methodologies as well as indicators where the concept is limited to one dimension (1D) (e.g. economic, social) or two dimensions (2D) (e.g. socio-environmental, socio-economic) exist (Parada et al., 2017). The 1D & 2D assessments are too limited to inform sustainability decisions as the performance of each dimension is essential to various stakeholder priorities such as investors (economic), employees (economic & social) and government/policy makers (social & environmental) (Moncada et al., 2016; Parada et al., 2017). Globally, several sustainable development policies are shifting towards the 3D criteria. An example is the proposed framework for transitioning from Millennium Development Goals (MDGs) to Sustainable Developments Goals (SDGs) in the '2030 Agenda for Sustainable Development' (OECD, 2016). Consequently, studies have emphasized the need to incorporate 3D sustainability in biorefinery designs and implementations, attentive to related impacts such as food security, environmental pollution, energy security, and socio-economic impacts (Moncada et al., 2016; Parada et al., 2017). Specifically, incorporating sustainability evaluations in biorefinery designs could facilitate identification of hotspots for improvements, and the selection of sustainable product integration schemes from possible options (Moncada et al., 2016; Parada et al., 2017).

Discrepancies in sustainability indicators dominate current sustainability discussions, attributed to the lack of standardized methodologies (Ciroth et al., 2011; Moncada et al., 2016; Parada et al., 2017). For example, the reliance of the social aspect on opinions of diverse stakeholders (e.g. investors, policy makers) with different priorities introduce subjectivity in the outcome (Falcone and Imbert, 2018; Ren et al., 2018). Nevertheless, some methodologies under development allow for adaptations for context-specific objectives and have proven useful for biorefinery implementation decision support. Life Cycle Sustainability Assessment (LCSA) is one such example, which has been advocated for decision-making towards more sustainable products or processes (Ciroth et al., 2011). The LCSA methodology supports the evaluation of environmental, social and economic impacts of the considered process or product along the entire value chain or under equal boundary specifications for purposes of comparing projects (Ciroth et al., 2011; Finkbeiner et al., 2010; Kloepffer, 2008). LCSA has been applied in waste biorefinery designs for some industries, such as the sugar mill industry (Gnansounou et al., 2017; Nieder-Heitmann et al., 2019). Nieder-Heitmann et al. (2019) for instance, applied LCSA to rank the sustainability of sugarcane bagasse & trash based biorefineries producing bioenergy only, bioenergy integrated with succinic acid, itaconic acid, or polyhydroxybutyrate & succinic acid. Conversely, little has been done for the biorefineries based on wastes from CSIs, thus, hampering their adoption.

Potential economic benefits from integrated cassava starch wastes biorefineries (CWB) over BAU's must not be pursued at the expense of higher environmental burdens and socio-economic detriments (Honnery et al., 2013). Thus, environmental burden mitigation and sustainability enhancement strategies must be considered in process designs and CWB products selection. The need for the 3D approach for the CWB calls for preliminary performance assessments to identify possible sustainable CWB scenarios, as well as identification of hotspots in the CWBs for prospective improvements. Therefore, in this study, the environmental burdens and the sustainability of five innovative CWB concepts and the BAU have been assessed and compared, to provide preliminary decision support frameworks for product integration schemes that can support development of sustainable CWBs. The CWB schemes incorporate innovative circular economy (CE) strategies, i.e. revitalization of products or resources after their end-of-life or functional life for reuse as raw materials rather than treated as waste (Agrawal and Singh, 2019), as possible TBL sustainability enhancement schemes in the CSIs. The innovative CE strategies involve total recovery & conversion of field wastes (CS) & process wastes (CWW + CB) into alternate products [bioethanol, glucose syrup, succinic acid, CHP] potentially supporting synergistic enhancements in economic, environmental and total in-house (CSF& CWB) energy provisions in CSIs. The incorporated CE strategies, therefore, promote sustainability measures regarding prudent and extended usage of the CSI's resources (Agrawal and Singh, 2019). The comparative LCSA was done using a percentage sustainability index (PSI) tool, custom-built for two perspectives of decision makers: (i) mutual investor-environmentalist perspective and (ii) investor perspective. The findings contribute to sustainable CWB process schemes that will advance investment decisions and applications in CSIs.

2. Description of the conceptualized cassava starch wastes biorefineries

The CWW and CB feedstock capacities for the studied scenarios were specified based on generation capacities for typical 200 t starch/d CSF, while the CS feedstock was projected based on feasibility demonstrations for the CWBs in a previous study (Padi and Chimphango, 2020a). The CWW, CB, and CS feedstock were, thus, specified at 377.83 t/h (Colin et al., 2007), 7.29 t DM/h (Chavalparit and Ongwandee, 2009), and 450.89 t/h (Padi and Chimphango, 2020a), respectively. The baseline conventional wastes management scheme to be compared with the CWB scenarios is presented in section 2.1. The process descriptions for the proposed CWB scenarios (Fig. 1), detailed in sections 2.2-2.6, were adapted from the previous study (Padi and Chimphango, 2020a) as summarised in Table 1.

2.1. Conventional management scheme for the cassava starch wastes (Business-As-Usual (BAU) scenario)

The BAU scenario (Fig. 2a) describes the prevailing approaches to handling the cassava starch wastes (summarised in Table 1). The process involves AD of the CWW (377.83 t/h) + CB (7.29 t DM/h) wastes to generate biogas for SDHA, followed by disposal of the effluent (digestate) into watercourses (Hansupalak et al., 2016; Tran et al., 2015). On the other hand, the CS (450.89 t/h) disposal simply involves gathering and open burning in the wild (Zhu et al., 2015). Thus, in the eLCA simulations, environmental loadings of the CWW + CB digestate are designated as emissions to water, while gaseous and solid emissions from the SDHA system (Fig. 2a) are designated as air emissions and landfill disposal respectively. For the open burning of the CS, complete combustion was assumed, where all gaseous emissions are designated as emissions to air, and the solid particulates (such as ash) are allocated to land emissions (Fig. 2a).

2.2. Combined heat and power (CHP) production (Scenario I)

Scenario (I) (Fig. 2b) comprises AD of the CWW (377.83 t/h) + CB (7.29 t DM/h) to produce biogas, followed by treatment of the AD effluent for recovering useable water and dried sludge. The biogas and dried sludge (40% w/w moisture) complement the CS (450.89 t/h) fuel to generate combined heat [SDHA, 170 °C, 3257 kJ/kg starch (Chapuis et al., 2017)] and power using a steam boiler (60 atm, 454 °C) & condensing turbo-generator system (Fig. 2b). The AD effluent is aerobically digested, followed by reverse osmosis (RO) treatment to generate treated water (Humbird et al., 2011) for recycling as process water. The brine effluent from the RO is evaporated to 50% w/w salts using a multiple effect evaporator, for subsequent incineration in the biofuel combustor (Humbird et al., 2011).

2.3. Production of hexose based bioethanol integrated with CHP (Scenario II)

In Scenario (II) (Fig. 3), the CWW (377.83 t/h) + CB (7.29 t DM/h) (38.4 kg/m^3 TS) is dewatered to 30% TS (w/w) and heated to 50 °C (Virunanon et al., 2013) via direct steam injection (13 atm, 268 °C), followed by addition of a commercial enzyme cocktail [Liquozyme® SC DS (0.2% w/w), Spirizyme® Fuel (0.066% w/w), and Novozyme® NS 50012 (0.4% w/w)] for the 24 h batch hydrolysis (Virunanon et al., 2013). The derived hexoses (C6 sugars) from hydrolysis are fermented to ethanol using *Zymomonas mobilis* (Z. mobilis) for 36 h (Humbird et al., 2011). In addition to CSL (corn steep liquor) and diammonium phosphate (DAP) nutrient requirements, at the projected demands (Appendix, Table A1), about 10% (w/w) of the sugars is used for Z. mobilis seed production while the rest is fermented to ethanol (Humbird et al., 2011). Temperature in the hydrolysis (50 °C) (Virunanon et al., 2013) and fermentation (32 °C) (Humbird et al., 2011) units is controlled using cooling water and chilled water respectively (Humbird et al., 2011). Bioethanol recovery entails two distillation columns (beer column & rectifier) and a molecular sieve adsorption unit (MSA) (Humbird et al., 2011). The beer column separates 40% ethanol and vents (mainly CO_2) from the fermentation broth as a side-draw and an overhead respectively. Ethanol entrained in the overhead is recovered via a water scrubber and returned to the beer column (Humbird et al., 2011). The rectifier dehydrates the 40% ethanol side-draw from the beer column into a 92.5% ethanol overhead and a 0.05% ethanol bottoms (Humbird et al., 2011). The rectification product (92.5% ethanol) is further dehydrated to a 99.5% ethanol product using the MSA (Humbird et al., 2011), while the separated solids in the beer column bottoms are recovered using pressure filtration for onward drying to 75% w/w solids for fuel applications in the CHP. With the exception of a Condensing Extraction Steam Turbine (CEST) that ensures steam extraction (13 atm) for the hydrolysis and distillation operations, the CHP follows is the same as that in section 2.2. All wastewater generated in the process is treated in the same way as that for Scenario II, and the resultant biogas and dried solids are integrated with the CS (450.89 t/h) as fuel for the CHP.

2.4. Co-conversion of pentose & hexose to bioethanol integrated with CHP (Scenario III)

Scenario (III) (Fig. 4) models conversions of pentose (C5 sugars) and hexose (C6 sugars) derived from integrated CB (7.29 t DM/h) and 10% CS (~45 t/h) to bioethanol, coupled with conversion of 90% of the CS (~405.8 t/h) to CHP. The bioethanol process commences with hammer milling of 10% of CS for blending with the CB + CWW. The mixture (23% w/w solids) is first dewatered to 49% solids then

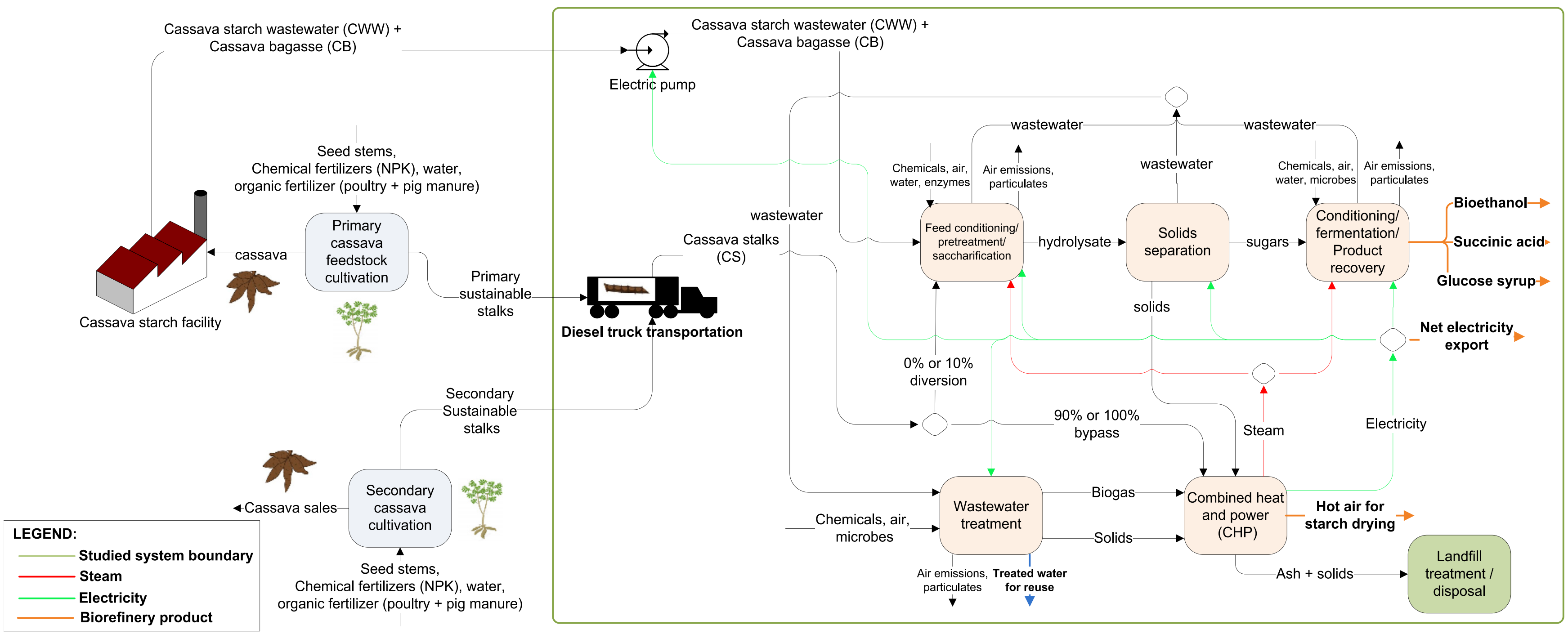


Fig. 1. Simplified diagram showing the system definitions and boundaries of integrated cassava waste biorefineries.

Table 1
Summary of the cassava wastes biorefinery scenarios.

Biorefinery scenarios	Feedstock inputs/h	Process description	Products recovered/h	Reference(s)
Business-as-usual (BAU)	377.83 t cassava starch wastewater (CWW) + 7.29 t cassava bagasse (CB) + 450.89 t cassava stalks (CS)	Anaerobic digestion of CWW + CB for biogas for hot air production & open burning of CS; Process electricity (360 kW) sourced from coal-based grid power	185 t starch drying hot air- SDHA (170 °C); 1.34 t surplus biogas	(Padi and Chimphango, 2020a, 2020b)
(I)	377.83 t CWW + 7.29 t CB + 450.89 t CS	CWW + CB biogas plus CS converted to combined heat and power (CHP); energy self-sufficient process	185 t SDHA; 303.07 MW electricity	Padi and Chimphango (2020a)
(II)	377.83 t CWW + 7.29 t CB + 450.89 t CS	CWW + CB for producing bioethanol and 100% CS by-passed to CHP; Enzymatic hydrolysis (EH) pre-treatment of CB; energy self-sufficient process	185 t SDHA + 289.2 MW electricity + 1.478 t bioethanol	Padi and Chimphango (2020a)
(III) 377.83 t CWW + 7.29 t CB + 450.89 t CS	CS + CB + CWW for bioethanol with 90% CS by-passed for CHP production; dilute acid + EH pre-treatment of CB + CS; energy self-sufficient process	185 t SDHA + 123.39 MW electricity + 8.955 t bioethanol	Padi and Chimphango (2020a)	
(IV) 377.83 t CWW + 7.29 t CB + 450.89 t CS	CS + CB + CWW for co-production of GS, bioethanol and CHP with 90% CS by-passed to CHP production; dilute acid + EH pre-treatment of CB + CS; energy self-sufficient process	185 t SDHA + 166.47 MW electricity + 5.722 t bioethanol + 9.287 t glucose syrup	Padi and Chimphango (2020a)	
(V) 377.83 t CWW + 7.29 t CB + 450.89 t CS	CS + CB + CWW for co-production of SA, bioethanol and CHP with 90% CS by-passed for CHP production; dilute acid + EH pre-treatment of CB + CS; energy self-sufficient process	185 t SDHA + 163.58 MW electricity + 5.722 t bioethanol + 6.908 t succinic acid	Padi and Chimphango (2020a)	

preheated to 100 °C by the addition of hot water (rectifier bottoms) and steam (13 atm, 268 °C), under control to achieve 30% w/w solids (Humbird et al., 2011). The mixture is then pre-treated with 1% H_2SO_4 and heated with steam (13 atm, 268 °C) to 170 °C for ~0.3 h (Humbird et al., 2011; Martín et al., 2017). The heated slurry is detoxified using a flash tank (130 °C) which separates 99% w/w of the inhibiting furfural and 10% w/w of the hydroxymethyl furfural (HMF) to the vapour. The detoxified product is conditioned to 27% w/w solids and pH of 5 through dosing with ammonia solution (Humbird et al., 2011). The solids are then separated from the sugar liquor and conditioned to 35% w/w solids mixture using water, followed by enzymatic hydrolysis via an in-house produced cellulase enzyme. The separated sugar liquor is divided into 92% and 8% for the ethanol fermentation and on-site enzyme production, respectively. Cellulase production via growth of the fungi *Trichoderma reesei* in a submerged aerobic environment (Humbird et al., 2011), is presumed for the in-house cellulase process. The ethanol fermentation & recovery, wastewater treatment, and CHP processes follow similar descriptions as those for scenario (II) (section 2.3).

2.5. Integrated pentose-based bioethanol, glucose syrup, and CHP production (Scenario IV)

Scenario (IV) is similar to scenario (III) in section 2.4, except for the diversion of the C6 sugars from the enzymatic hydrolysis for glucose syrup (GS) production (Fig. 5). The GS process begins with centrifuging the C6 sugar hydrolysate, thereby separating the insoluble solids including ash and fibre (Hobbs, 2009), which are washed to recover glucose losses (Knight, 2011). The hydrolysate is then purified via adsorption with granular activated carbon, which removes colour and odour inducing impurities such as HMF (Hobbs, 2009), followed by steam concentration to the 70% glucose syrup product using steam at 9 atm and 232 °C in a multiple-effect evaporator (Hobbs, 2009; Humbird et al., 2011).

2.6. Integrated pentose-based bioethanol, succinic acid, and CHP production (Scenario V)

Scenario (V) (Fig. 5) is similar to scenario (IV) (section 2.5), except that the detoxified but non-concentrated GS is further converted to succinic acid (SA) (Fig. 5). The non-concentrated glucose syrup (150.74 g/L sugars) is fermented to SA based on experimental results (SA yields of 0.82 w/w DM CB) for *E. coli* SA fermentations (Sawisit et al., 2015). The *E. coli* seed growth/fermentation nutrients are presumed similar to *Z. mobilis* in ethanol fermentation (Liu et al., 2008). Caustic dosing (10 mol/L) controls the fermenter pH (Liu et al., 2008), while CO_2 for the *E. coli* growth/fermentation is supplied using that from the ethanol distillation beer column (see section 2.3) (Lynd et al., 2005). The fermenter broth is centrifuged and acidified to a pH of 2.2 through H_2SO_4 dosing, which favours separation of succinate from sodium succinate salts (Klein et al., 2017). The succinate is then concentrated via evaporation (101 °C and 1 atm) (Vlysidis et al., 2011), and further purified using selective adsorption (ZSM-5 zeolite), evaporative concentration to saturation (90 °C, 0.7 atm) (Klein et al., 2017), crystallization at 4 °C (Vlysidis et al., 2011), and air drying to obtain the SA (98.1% purity) (Klein et al., 2017).

3. Methodology

The sustainability of the conceptual CWBs is evaluated based on the principles of LCSA, as defined in Eq. (1). Thus, the requisite environmental Life Cycle Assessments (eLCA), Life Cycle Costing (LCC), and social Life Cycle Assessments (sLCA) are evaluated based on the principles of Life Cycle Assessments (LCA), Techno-economic Assessments (TEA), and related socio-economic impacts respectively (detailed in section 3.3). The eLCA follows the standards defined by the ISO14040 and ISO14044 (ISO, 2006), involving: (i) definition of goal & scope for the study, which includes delineating the system boundary and functional unit (FU) to facilitate comparison of scenarios, (ii) life cycle inventory (LCI), (iii) life cycle impact assessments (LCIA), (iv) results interpretation. The TEA was based on Aspen Plus® process simulations for the CWBs, reported in previous studies (Padi and Chimphango, 2020a, 2020b).

$$LCSA = eLCA + LCC + sLCA \quad (1)$$

Where: eLCA-environmental life cycle assessment; LCC- life cycle costing; sLCA-social life cycle assessment. The additions (+) are figurative and involve methodological valuations (detailed in

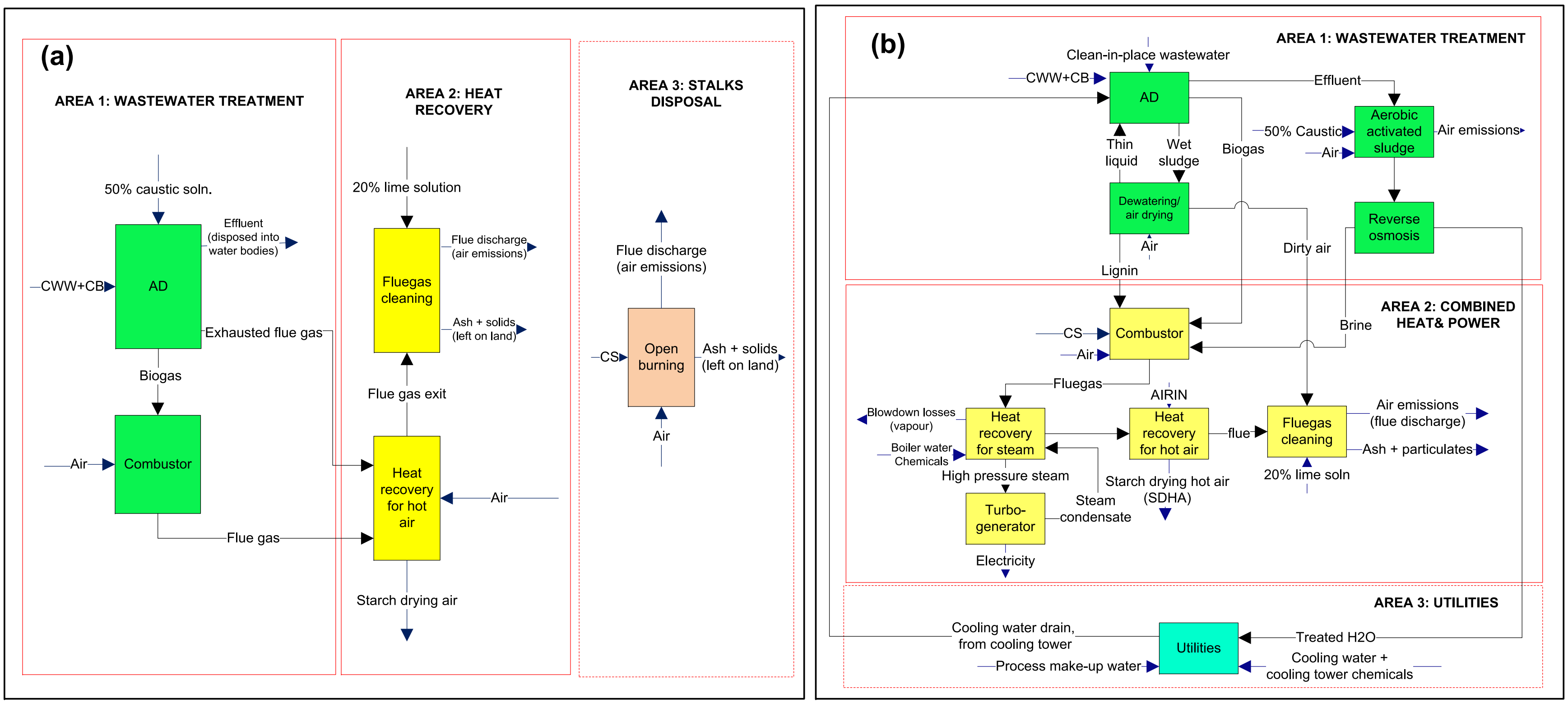


Fig. 2. Schematic diagram of the cassava waste conversion/disposal systems. (a) Anaerobic digestion (AD) of cassava starch wastewater (CWW) and bagasse (CB) for biogas based starch drying hot air generation, plus open burning (disposal) of cassava stalks- CS (field wastes) [BAU scenario]. (b) Production of combined heat and power (CHP) from CS integrated with biogas from CWW + CB [scenario (I)] [Adapted from (Padi and Chimphango, 2020a)].

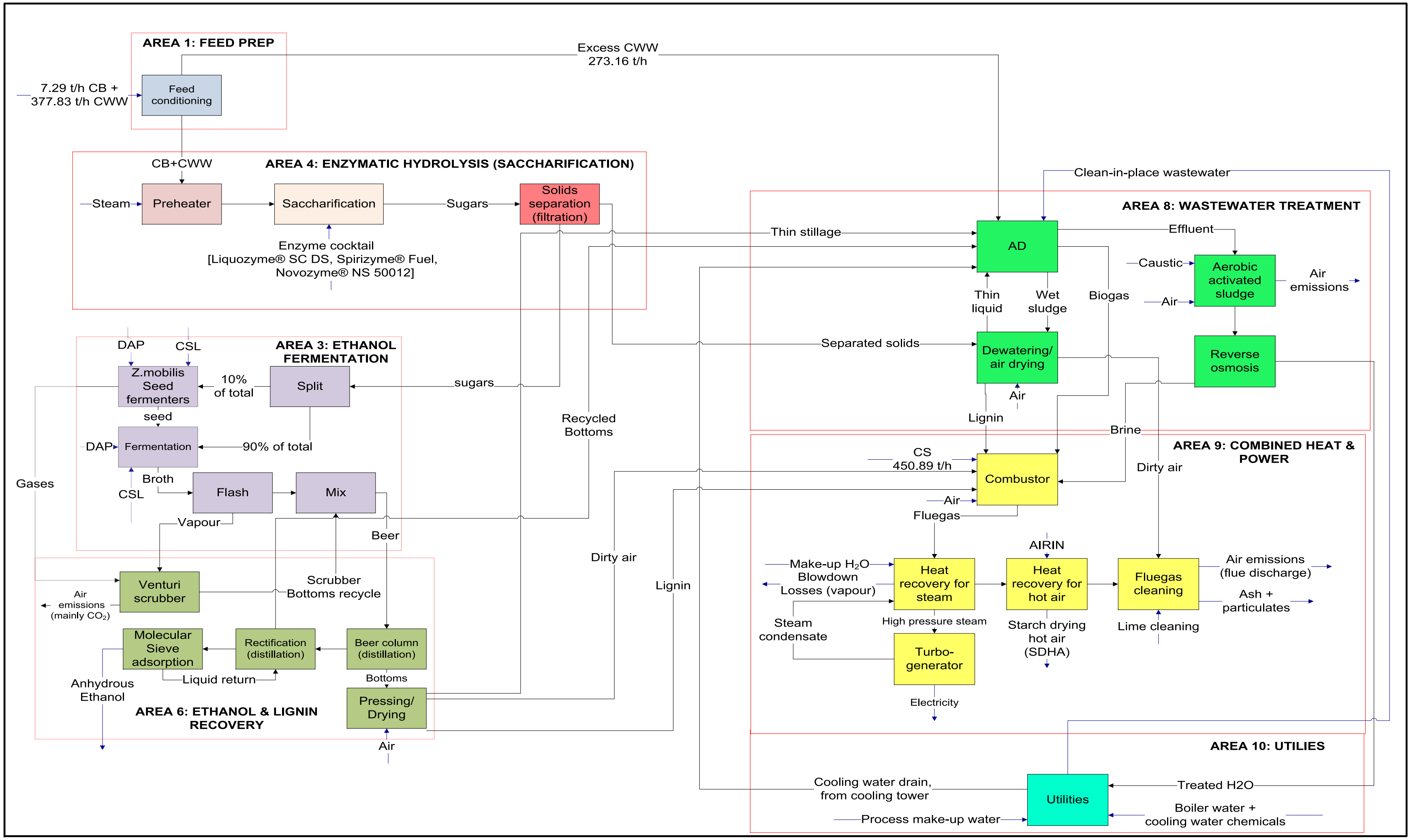


Fig. 3. Schematic layout of the cassava starch wastes biorefinery co-producing bioethanol from integrated cassava starch wastewater and bagasse, and combined heat and power (CHP) from cassava stalks [scenario (II)] (Adapted from (Padi and Chimphango, 2020a)). In the diagram, DAP = diammonium phosphate, CSL = corn steep liquor, AD = anaerobic digestion, CWW = cassava starch wastewater, CB = cassava bagasse, CS = cassava stalks.

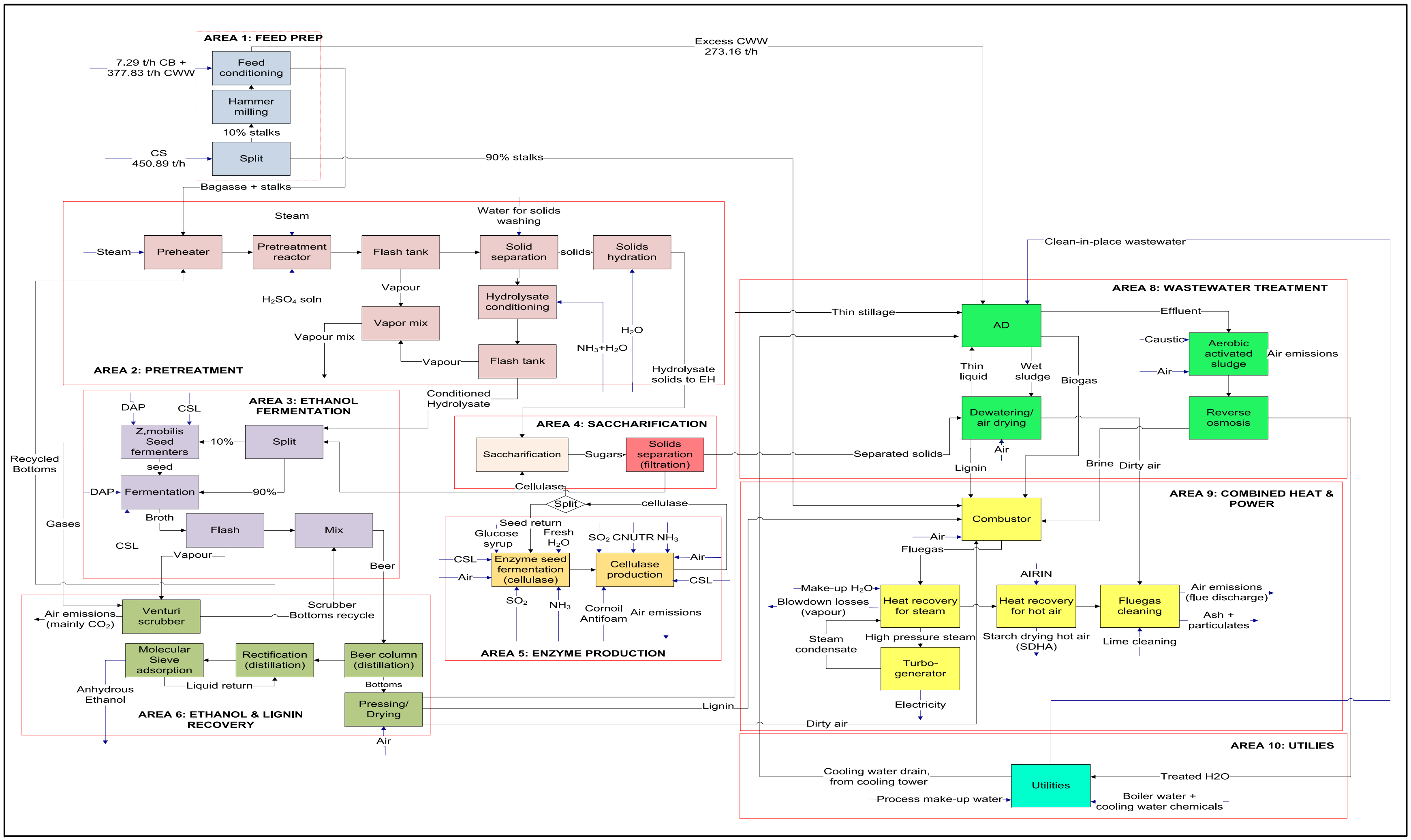


Fig. 4. Schematic diagram of the cassava starch waste biorefinery co-producing bioethanol from integrated CWW + CB & 10% CS, and combined heat and power (CHP) from 90% CS [scenario (III)] (Adapted from (Padi and Chimphango, 2020a)). In the diagram, AD = anaerobic digestion, CB = cassava bagasse, CNUTR = Cellulase nutrient mix, CS = cassava stalks, CSL = corn steep liquor, CWW = cassava starch wastewater, DAP = diammonium phosphate, EH = enzymatic hydrolysis.

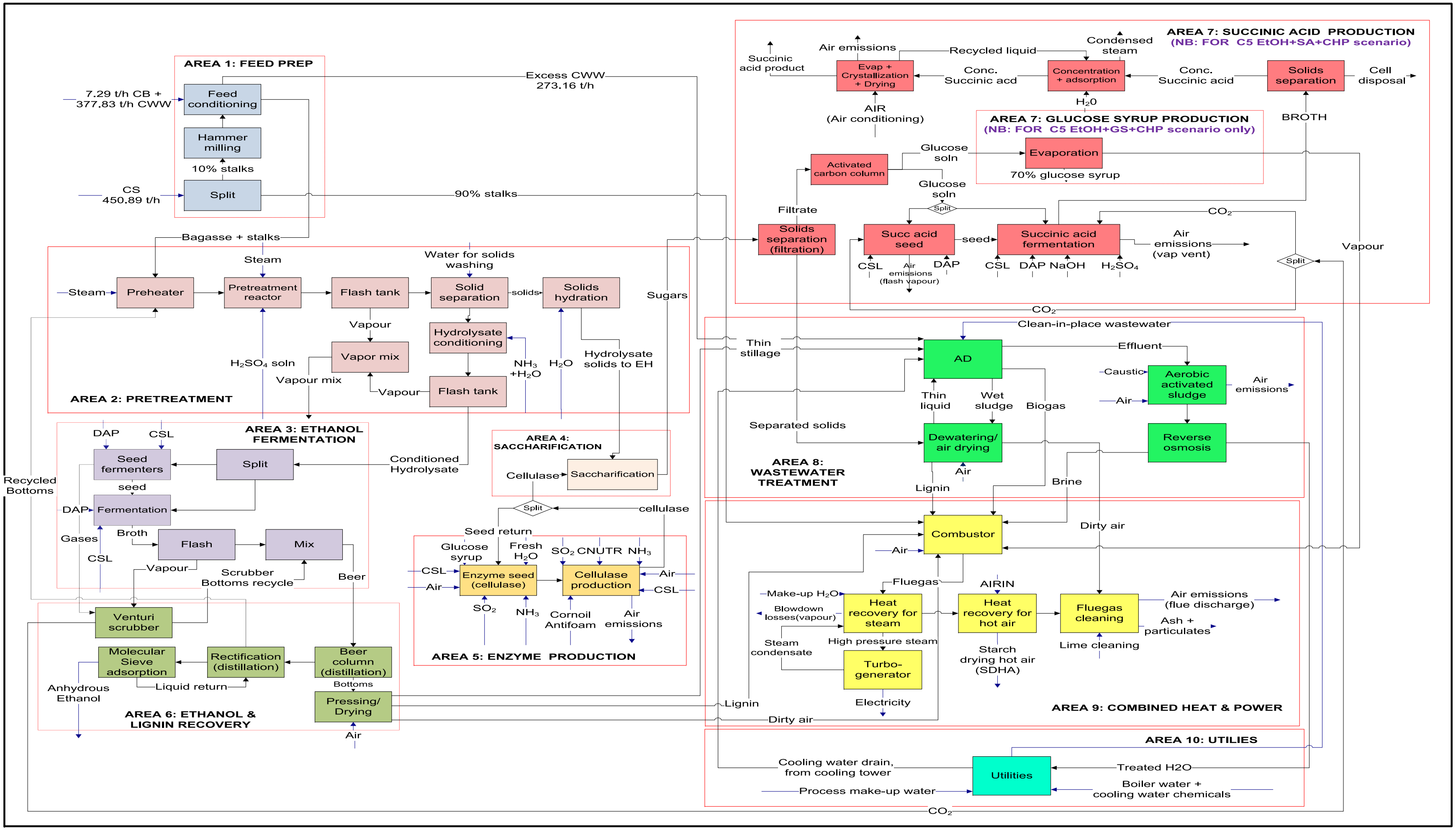


Fig. 5. Schematic layout of the cassava starch waste biorefinery co-producing combined heat and power (CHP) from 90% CS and bioethanol plus glucose syrup [scenario (IV)] or succinic acid [scenario (V)] from integrated CWW + CB & 10% CS (Adapted from (Padi and Chimphango, 2020a)). In the diagram, AD = anaerobic digestion, CB = cassava bagasse, CNUTR = Cellulase nutrient mix, CS = cassava stalks, CSL = corn steep liquor, CWW = cassava starch wastewater, DAP = diammonium phosphate, EH = enzymatic hydrolysis.

section 3.3.2).

3.1. Goal and scope of the study

The main aim of the study is to evaluate and compare the environmental impacts and the sustainability of conceptual CWBs (section 2) for purposes of identifying product integration schemes that can support sustainable developments of CWBs. The sustainability of the CWBs was assessed by means of LCSA (Eq. (1)). The study aims to contribute to knowledge on sustainability of CWBs for stakeholder (CSIs, investors, policy makers) deliberations regarding implementation decisions. The presumed geographical setting for the biorefinery is South Africa. The functional unit (FU) is specified as a biorefinery converting 1-ton combined feedstock, comprising (w/w) 45.2% CWW, 0.9% CB, and 53.9% CS (Appendix, Table A1). The system boundary is delineated as feedstock transportation, plus biorefinery gate-to-gate, plus landfill treatment of generated ash (Fig. 1). The feedstock transportation highlights related environmental burdens incurred by the CWBs, which is absent in the prevailing BAU scenario (section 2.1). The scope definition further assumes the following CWB design:

(i) The CWB is an annex to the host 200 t starch/d CSF, thus, the CWW + CB feedstock is pumped to the CWB (Fig. 1).
(ii) The CS feedstock is transported from the farms to the CWB by means of diesel powered trucks (Fig. 1).
(iii) Feedstock cultivation is not considered due to the equal feedstock capacities in the CWB and BAU scenarios, thus, of minimal consequence to the comparative sustainability assessments.
(iv) Construction and decommissioning of the biorefinery infrastructure are excluded in the eLCA, due to the negligible environmental contributions to the biorefinery products, attributed to the relatively long lifespans of such infrastructure (Falano et al., 2014).

3.2. Life cycle inventory data and assumptions in assessing the environmental impacts

The LCI background data is obtained from related literature and/or Ecoinvent v.3.5 database (Ecoinvent, 2018), detailed as follows:

(i) Data on quantities of raw materials/products, utilities, energy, and emissions for the biorefineries, summarised in the Appendix (Table A1), is obtained from Aspen Plus® simulated mass and energy balances in previous works (Padi and Chimphango, 2020a, 2020b).
(ii) In the BAU scenario, where the ash from stalk burning is left untreated on the land (section 2.1), ash composition reports for thick CS by Veiga et al. (2016) is adopted.
(iii) In the CWB scenarios, relative to the landfill treatment of generated ash (Fig. 1), due to similarity of wood ash compositions to CS ash (Veiga et al., 2016), Ecoinvent v.3.5 LCI data for wood ash landfill treatment is assumed (Ecoinvent, 2018).
(iv) Concerning the CS feedstock, 20% w/w of the generated stalks is used for planting and social uses such as heating fuel (Ozoegwu et al., 2017; Zhu et al., 2015), thus, only 80% is recoverable for conversion in the CWB. In addition, the CS production rate is based on yield reports of CS-to-cassava root ratio of 0.51 (Zhu et al., 2015). The CS transportation distance, from farms to the CWB, is estimated relative to reports of 48 km (Hansupalak et al., 2016) radius for the primary sustainable CS (0.8×0.51 x 842 t cassava/d; 343.54 t CS/d) associated with the 842 t cassava/d feedstock for the 200 t/d CSF (Fig. 1). Consequently, the total CS (~10,821 t/d) transportation distance is proportionally estimated at ~270 km radius. Hence, Ecoinvent v.3.5 LCI data (Ecoinvent, 2018) for diesel truck for short haul distance (<322 km) have been considered.
(v) For CWW + CB transportation from the CSF to the annex biorefineries, pumping to 2.47 atm with a pump power of 32.77 kW (Aspen Plus® prediction), supplied using the generated bioelectricity (Fig. 1) in the CWBs or coal based grid power in the BAU (Padi and Chimphango, 2020a, 2020b), is presumed.
(vi) Coal based grid power is assumed for supplying the total electricity demands for the AD biogas SDHA process in the BAU scenario (Padi and Chimphango, 2020b), which was predicted at 360 kW through Aspen Plus® simulations (Padi and Chimphango, 2020b).

In eLCA for multi-product systems such as biorefineries, standardized methodologies are required to assess the environmental impacts of the wide-ranging products, where system expansion or partitioning are well established methods (European Commission, 2015; ISO, 2006). System expansion involves redefining the FU to include functions of all co-products, or allocations of avoided impacts from products assumed to be substituted by the co-products to the selected main product. Conversely, the partitioning method considers allocation of burdens to all products, based on physical (mass, volume, or energy content) or economic (production cost, market value) attributes. Partitioning by economic allocation using total revenues (detailed in Appendix, Table A2), which is an essential attribute to the study's interest of biorefinery sustainability (Gnansounou et al., 2015; Pereira et al., 2015), is considered in the present study.

3.3. Sustainability assessments of the cassava wastes biorefineries

3.3.1. Sustainability metrics

In view of the concerns on water & land pollution, and the high carbon footprints associated with current CSF waste management schemes (section 1), the environmental metrics deemed relevant for consideration include global warming potential (GWP), freshwater eutrophication potential (FEP), freshwater ecotoxicity potential (FETP), terrestrial acidification potential (TAP), terrestrial ecotoxicity potential (TETP), and fossil resource scarcity potential (FRSP).

The proposed methodology for the LCSA involves evaluation of the environmental metric as the conventional environmental life cycle assessment (eLCA), the costs metric (called Life Cycle Costing-LCC) as costs implications for each stage of the life cycle, and the social metric (called Social Life Cycle Assessment-sLCA) as the socio-economic impacts such as job creation among others (Ciroth et al., 2011), as summarised in Eq. (1). The referred metrics consist of sub-metrics that could be classified into two categories: 'hard' and 'soft' criteria (Ren et al., 2016). The hard criteria refer to quantifiable factors that can be evaluated and expressed in crisp values (e.g. capital costs), whereas the soft criteria are qualitative or subjective factors evaluated based on knowledge or experience of the decision-maker or stakeholder (e.g. social acceptability of a product) (Ren et al., 2016). In the present study, because of the lack of empirical stakeholder experiences due to the hypothetical status of the CWBs and unexplored in-depth stakeholder engagements, the proposed preliminary sustainability framework was confined to the hard criteria summarised in Fig. 6.

The eLCA was achieved using the related ReCiPe 2016 midpoint (H) v1.03 method (Silalertruksa et al., 2017) via simulations in

Table 2
Summary of the life cycle sustainability assessment (LCSA) metrics and illustration of the sustainability index calculations for the biorefineries.

	Targeted value	BAU Scenario	CHP, scenario (I)	C6EtOH + CHP, scenario (II)	C5−C6EtOH + CHP, scenario (III)	C5EtOH + GS + CHP, scenario (IV)	C5EtOH + SA + CHP, scenario (V)	
Life cycle costing (LCC) [a]								
Total capital investment, TCI (million US$)	Minimum	51.87	545.11	594.96	921.918	951.204	1007.61	
Total production cost, TPC (million US$/a)	Minimum	8.87	285.28	289.84	280.75	290.81	331.16	
Net Present Value, NPV (million US$)	Maximum	92.71	438.53	331.03	−1001.95	−388.53	3.21	
Environmental life cycle (eLCA) [b]								
Net GWP (kg CO_2)[c]	Minimum (negative values preferable)	1.3	−379	−356	−146	−211	−182	
Net water scarcity (NWS) (m^3)[c]	Minimum (negative values preferable)	−0.0042	2.06	1.42	2.65	2.07	4.49	
Terrestrial acidification (kg SO_2)	Minimum	0.55	0.32	0.31	0.39	0.39	3.60	
Freshwater eutrophication (kg P eq)	Minimum	0.90	0.02	0.02	0.02	0.02	0.06	
Terrestrial ecotoxicity (kg 1,4-DCB)	Minimum	6.63	19.05	19.10	27.92	28.40	68.90	
Freshwater ecotoxicity (kg 1,4-DCB)	Minimum	0.19	70.42	70.04	64.74	64.76	122.25	
Fossil resource scarcity (kg oil eq)	Minimum	0.57	7.22	7.27	8.89	9.20	22.39	
Social life cycle (sLCA)								
Job creation (number of jobs created)[d]	Maximum	23	32	46	60	65	69	
Energy security (net electricity export, kW)	Maximum	0.00	362.52	345.93	147.59	199.12	195.67	
Human toxicity potential (kg 1,4-DB eq)	Minimum	9.54	19.74	20.73	19.93	20.19	46.30	
Percentage sustainability index (PSI) for the 'Case A' sustainability perspective								**Weighting factors (%)**
Total capital investment, TCI		4.00	0.38	0.35	0.23	0.22	0.21	4
Total production cost, TPC		4.00	0.12	0.12	0.13	0.12	0.11	4
Net Present Value, NPV		6.77	32.00	24.16	−73.11	−28.35	0.23	32
Total LCC sustainability index (%)		**14.77**	**32.50**	**24.63**	**−72.76**	**−28.01**	**0.55**	**40.00**
Net GWP		−0.03	10.00	9.39	3.85	5.57	4.80	10.00
Net water scarcity (NWS)		10.00	−0.020	−0.029	−0.016	−0.020	−0.009	10.00
Terrestrial acidification		2.29	3.87	4.00	3.23	3.20	0.35	4.00
Freshwater eutrophication		0.11	3.93	3.96	4.00	3.92	1.52	4.00
Terrestrial ecotoxicity		4.00	1.39	1.39	0.95	0.93	0.38	4.00
Freshwater ecotoxicity		4.00	0.0106	0.0107	0.0115	0.0115	0.0061	4.00
Fossil resource scarcity		4.00	0.31	0.31	0.25	0.25	0.10	4.00
Total eLCA sustainability index (%)		**24.36**	**19.50**	**19.03**	**12.29**	**13.86**	**7.15**	**40.00**
Job creation (number of jobs created)		1.33	1.86	2.67	3.48	3.77	4.00	4
Energy security (net electricity export)		0.00	6.00	5.73	2.44	3.30	3.24	6
Human toxicity potential		10.00	4.83	4.60	4.79	4.72	2.06	10
Total sLCA sustainability index (%)		**11.34**	**12.69**	**12.99**	**10.71**	**11.79**	**9.30**	**20.00**
Total sustainability index (%)		**50.47**	**64.70**	**56.65**	**−49.77**	**−2.36**	**17.00**	**100.00**

[a] Values adopted from previous works- BAU values from (Padi and Chimphango, 2020b) & scenarios (I)-(V)'s from (Padi and Chimphango, 2020a).
[b] Simulation results in the present study for 1-functional unit (processing of 1-ton collective feedstock, comprising (w/w) 45.2% CWW + 0.9% CB + 53.9% CS).
[c] Net GWP = total biorefinery GWP minus total GWP for the equivalent fossil-based products (processes) & NWS = total biorefinery water scarcity minus total water scarcity for the equivalent fossil-based products (processes) (see section 3.3.1 & Appendix, Table A3). Therefore, negative Net GWP & NWS results imply environmental savings by the CWB products vs. corresponding fossil-products.
[d] Skilled + unskilled labour projections for the biorefineries based on the previous studies (Padi and Chimphango, 2020a, 2020b). 'Case A' sustainability perspective (mutual investor-environmentalist viewpoint, see section 3.3.2). BAU = business-as-usual, C5EtOH = pentose based bioethanol, C5−C6EtOH = pentose + hexose based bioethanol, C6EtOH = hexose based bioethanol, CB = cassava bagasse, CWW, cassava starch wastewater, CS, cassava stalks, CHP = combined heat and power, GS = glucose syrup, GWP, global warming potential, SA = succinic acid.

SimaPro 9.0.0.49 software (PRé Consultants, 2019). Characterization results were chosen for the referred environmental impact categories to enable various stakeholders to subject the findings to contextually relevant factors. Furthermore, to facilitate the incorporation of holistic environmental benefits of the CWBs vs. corresponding fossil-based processes in the LCSA, in addition to the FEP,

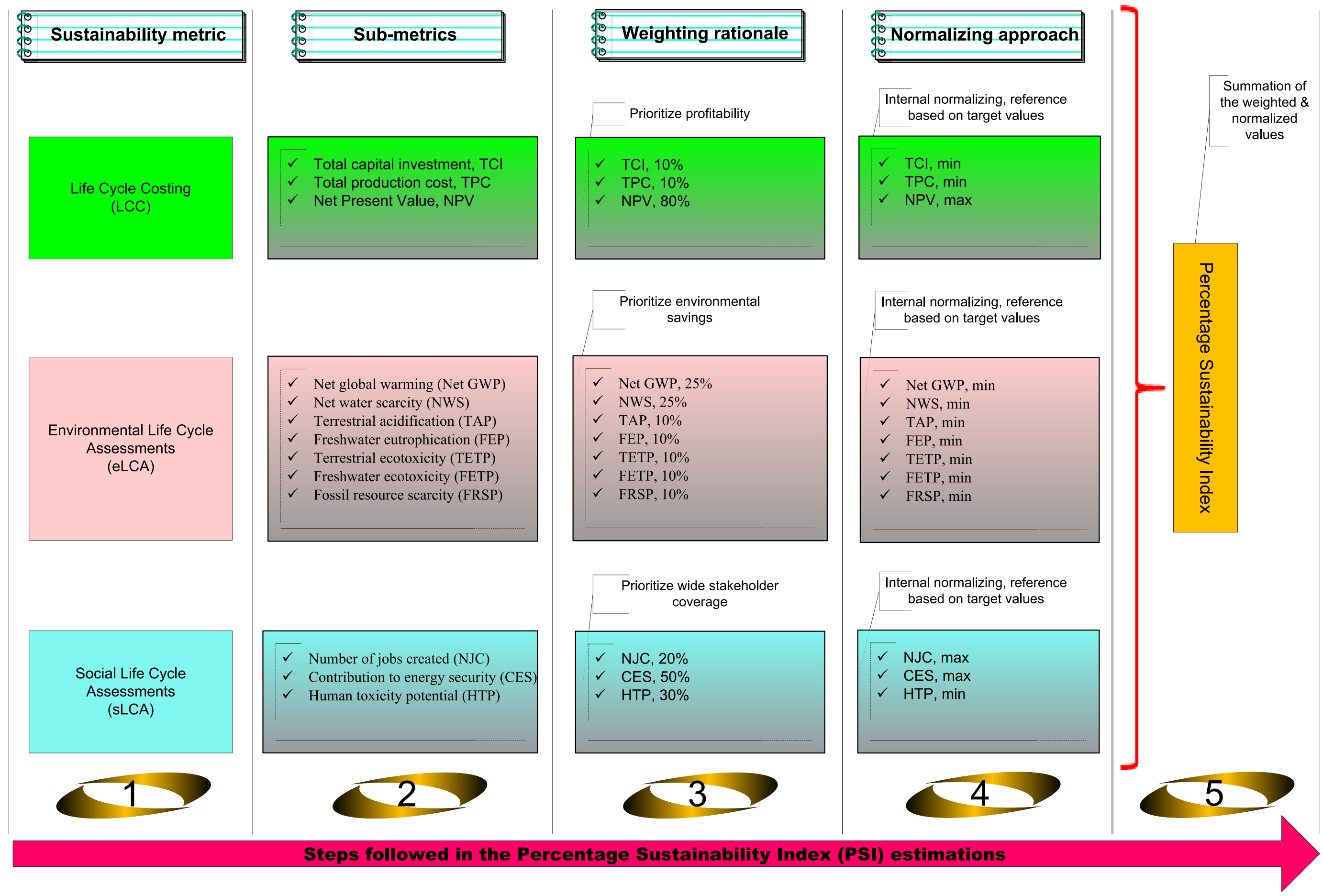


Fig. 6. The percentage sustainability index (PSI) framework applied in this study. Net GWP = total biorefinery GWP minus total GWP for the equivalent fossil-based products (processes) & NWS = total biorefinery water scarcity minus total water scarcity for the equivalent fossil-based products/processes (see section 3.3.1& Appendix, Table A3).

FETP, TAP, TETP and FRSP, net global warming potential (Net GWP) and net water scarcity (NWS) indicators have been included in the eLCA (Fig. 6). The Net GWP refers to the total biorefinery GWP minus the total GWP for equivalent fossil-based products (processes), detailed in the Appendix (Table A3). The NWS is similar to the Net GWP, except that the metric of interest is the water resource scarcity (Appendix, Table A3). The respective Net GWP and NWS were evaluated based on the single issue GWP method of IPCC 2013 (GWP_{100a}) and the water footprint method of Hoekstra et al. (Ecoinvent, 2018; Hoekstra et al., 2012).

In relation to the LCC, the Net Present Value (NPV) profitability indicator, total production costs (TPC), and total capital investments (TCI) have been considered (Nieder-Heitmann et al., 2019). The NPVs were estimated relative to year 2018 economic context for South Africa. The estimations involved projection of the TPC, TCI, and total revenues from product sales, which were based on simulated mass and energy balances in Aspen Plus®, detailed in previous works (Padi and Chimphango, 2020a, 2020b). The referred estimates, together with assumed operational and economic factors, including debt financing (debt-to-equity ratio of 1.5; 8% interest rate & 10-year recovery), operating period of 8410 h/a & plant life of 30 years, real term discount rate of 9.7% & inflation rate of 5.7%, were used to project discount cash flows, which were used to evaluate the NPVs as elaborated in the previous works (Padi and Chimphango, 2020a, 2020b).

With respect to the sLCA, the job creation metric has been considered (Gnansounou et al., 2017; Nieder-Heitmann et al., 2019), and was estimated as the skilled + unskilled labour projections for the biorefinery facility, based on the previous studies (Padi and Chimphango, 2020a, 2020b). High costs and unreliable supplies of energy have been cited as constraints to industrial developments of cassava in most of the deprived cultivation regions, such as Thailand (Tran et al., 2015), Ghana (Kleih et al., 2013) and Nigeria (Nang'ayo et al., 2005). Energy security benefits from the CWBs could, therefore, enhance industrial expansions and related socio-economic developments, thus, considered in the social criteria (Gnansounou et al., 2017; Ren et al., 2016). The prospective contribution of the CWBs to energy security was estimated as the net surplus electricity after meeting the in-house requirements (see Fig. 1 & Appendix, Table A1). Due to the human health risks concerns for biorefineries (Nanda et al., 2015), human toxicity potential (HTP) was also included in the sLCA (Ren et al., 2018), and was evaluated using the SimaPro models for the CWBs and the methodology of CML-IA baseline v3.05 (Nieder-Heitmann et al., 2019).

3.3.2. Sustainability indicator estimations

For purposes of comparing the TBL sustainability of alternate projects, several approaches to integrating the procedure or results for the various metrics of the LCSA (eLCA, LCC, sLCA) into a sustainability index, including weighting and normalization of the data, have been proposed (Finkbeiner et al., 2010; Keller et al., 2015). In the conventional weighting approach, the weights are based on the importance of the metrics and the priorities of the stakeholders (e.g. investors, policy makers, employees) (Ren et al., 2018), resulting in potential drawbacks of introducing uncertainties in the outcomes. As a result, the integration of the metrics into a single sustainability index has been recommended as an optional step in LCSA that could be tailored for context-specific purposes in tune with the project's objectives (Ciroth et al., 2011). Therefore, the reliability of the sustainability index depends on the estimated weights, with the objective method (e.g. entropy weighting) and subjective method (e.g. analytic hierarchy process-AHP) of estimations identified as useful with regards to accounting for importance/effects of each metric and preference of the decision-makers respectively (Moradi-Aliabadi and Huang, 2016). In effect, reliable weight estimates will require direct stakeholder participation and inputs via group discussions or surveys (Falcone and Imbert, 2018; Ren et al., 2018).

In relation to the proposed CWBs, because of the unexplored stakeholder engagements, a tailored approach was developed for the integration of the LCSA metrics into a percentage sustainability index (PSI) (summarised in Fig. 6), which was based on reports of high environmental burdens & waste treatment costs in CSFs (Hansupalak et al., 2016; Pingmuanglek et al., 2017; Sánchez et al., 2017), as well as potential high investment costs constraint to the uptake of the CWBs.[1] Hence, two perspectives of investment decision-making have been considered in the PSI weightings: (i) Mutual investor-environmentalist viewpoint: high and equal LCC & eLCA (40:40%) with low sLCA (20%) [**Case A baseline**], (ii) Investor viewpoint: high LCC (50%) with equal eLCA & sLCA (25:25%) [**Case B baseline**]. In addition, for both Cases A & B, sensitivity analysis was performed to analyze the responses of the sustainability (PSI) to changes in the total weights of the LCC and eLCA, which involve ±5% adjustments of the LCC or eLCA weights for ranges from 0% to their summed weight (i.e. A- 80%, B- 75%), while keeping the sLCA's constant. The sub-weightings prioritized motivations of profitability [allocation of 80% of LCC to NPV & 10% each to TCI and TPC] and environmental savings [allocation of 50% of eLCA to the environmental savings (Net GWP & NWS) & 50% to the biorefinery gate impacts], as illustrated for the Case A in Table 2. For instance, it is likely that irrespective of the capital (TCI) and production cost (TPC) demands, profitable investment returns (positive NPV) could motivate uptake of the CWBs. Relevant to the sLCA, the need to consider the impacts on the socio-economic wellbeing of all actors along the value chain (i.e. from raw material producers to products consumers) has been emphasized (Falcone and Imbert, 2018). The related limitations in the sLCA metrics were factored in their sub-weightings (allocations of 20% of sLCA to job creation, 30% to energy security & 50% to HTP). For example, the job creation estimate reflects employment in only the CWB facilities (section 3.3.1), whereas the HTP and the energy security may reflect impacts on broader actors (e.g. HTP includes impacts from raw materials & end-products; the surplus electricity exports could facilitate external industrial expansions). The weighted metrics were internally normalized among the CWB scenarios. Thus, for parameters with minimum targets (e.g. human toxicity potential) and maximum targets (e.g. NPV), the weighted metrics were normalized against the lowest and highest values respectively (see Table 2). The internally normalized results for each CWB scenario are then added to obtain the CWB's PSI (see Table 2).

[1] Observations from field visits and personal discussions with managements of Ayensu Starch Company Limited (cassava starch facility) and Caltech Industries (cassava ethanol facility) in the Central region (Awutu-Bawjiase) and Volta region (Hodzo) of Ghana, respectively.

3.3.3. Sensitivity analysis of the sub-metric weighting impacts on the sustainability projections

To establish the impacts of the sub-weightings on the PSIs, a related sensitivity analysis was performed using the 'Case A' stakeholder perspective (section 3.3.2) as case study. The analysis involved comparing the PSI for the 'Case A' baseline weighting scenario (presented in Table 2) to PSI values corresponding to various scenarios of adjusted sub-weights for each sub-metric: (i) Scenario 1 (Sc. 1)- Equal sub-weights for each category of the LCSA metrics (i.e. 13.33% each for TCI, TPC & NPV; 5.714% each for Net GWP, NWS, TAP, FEP, TETP, FETP & FRSP; 6.66% each for job creation, energy security & human toxicity potentials) (see Appendix, Table A.4), (ii) Scenarios 2–14 (Sc. 2–14)- For each category (i.e. LCC, eLCA, or sLCA), 35% of the total category weight is assigned to a sub-metric (dominant metric) while the 65% is equally split among the other sub-metrics [i.e. $\hat{W}_j = 35\%\ T$, ($j = 1, 2, 3 \ldots$); $\hat{W}_k = 65\%\ T/(n\text{-}1)$, ($k = 1, 2, 3 \ldots$); where $j \neq k$, $\hat{W}_{j \text{ or } k}$ = weight of the sub-metric 'j' or 'k', T = total category weights, n = total number of sub-metrics in the category]. This weighting process is repeated in a successive manner for the subsequent sub-metrics in the category (subsequent scenarios), while maintaining equal sub-weightings for the other categories (detailed in Appendix, Table A4).

4. Results and discussions

4.1. Environmental impact potentials of the biorefineries

4.1.1. Global warming potential (GWP) and fossil resource scarcity potential (FRSP)

In the simulated CWBs, transportation of CS from farms to the CWBs contribute substantially (23–68%) to the GWP (Fig. 7a), thus, a possible hotspot for mitigation deliberations. The estimated CS transportation distance of 270 km radius was based on avg. CS-cassava root yield ratio of 0.51 (Zhu et al., 2015) (section 3.2). Reports of higher CS-to-cassava yields, up-to 0.85 (Zhu et al., 2015), implies existing possibilities for cultivation of higher CS cultivars, which could potentially reduce the transportation distance, thus an avenue to substantial reductions in the GWPs of the biorefineries. Transportation of CS from farms to CWBs, using diesel powered trucks, contributed substantially to the GWP profiles, at respective contributions of ~68, 52, 50, 49, and 23% for the scenarios (I)–(V) (Fig. 7a), hence, a prospective avenue for GWP reductions in the CWBs.

With respect to the BAU scenario, the GWP of 4.50 kg CO_2 eq/FU (Fig. 7a) is mainly due to the non-renewable coal-based grid power presumed for supplying the AD biogas-SDHA process power (360 kW) (Padi and Chimphango, 2020b) and the FGD lime, with contributions of 9.4% and 88.7% (respectively) of the GWP (Fig. 7a). This assertion is further supported by the comparable avg. GWP reports of 0.84 kg CO_2 eq/kWh for pulverised coal power systems (without carbon capture and storage) (Widder et al., 2011), relative to the GWP of ~0.45 kg CO_2 eq/kWh coal power consumed for the BAU scenario [Calculated as: (0.094 × 4.5 kg $CO_{2\ eq}$/t feedstock) x (385.12 t feedstock/h) x (1/360 kW)].

The GWP associated with electricity production seemingly doubles (7.7 vs. 14.2 kg CO_2 eq/FU) when C6 bioethanol is integrated into the CHP (scenario II) (Fig. 7a). Considering the relatively similar net power capacities for the referred scenarios (~363 vs. 346 kW h/FU; Table A1), the large differences in the electricity GWP could be a reflection of the relatively high economic allocation factor (~25-folds higher) for electricity vs. that of bioethanol (Table A2), attributed to the high total revenue for scenario (II) electricity (~US$ 312 million) compared to bioethanol's (~US$12 million) (Table A2).

Comparing scenario (III) to (IV) revealed that the diversion of

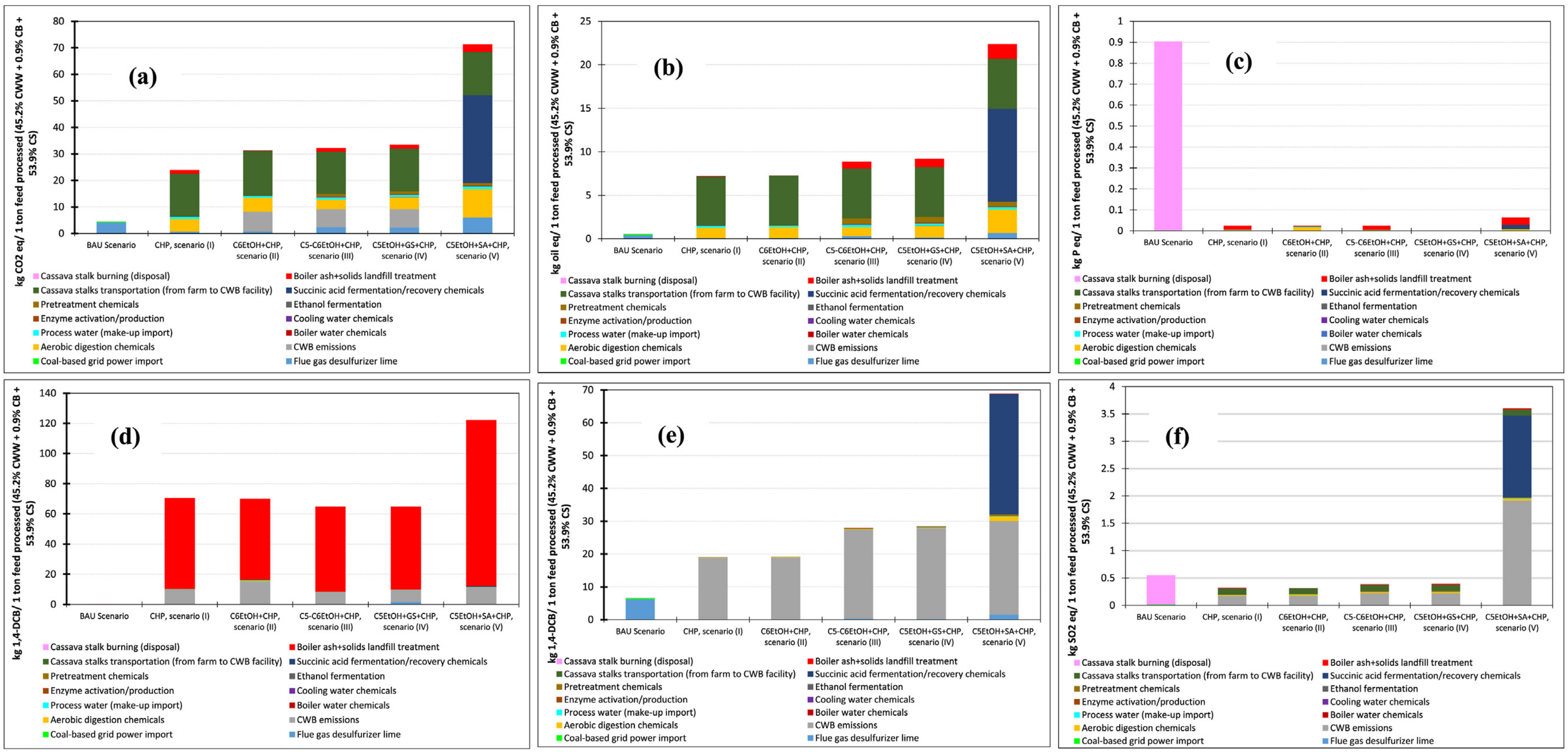


Fig. 7. Results of the Life Cycle Assessments for the cassava wastes biorefineries, based on the method of ReCiPe (2016) midpoint (H) v1.03/World (2010) H/Characterization. (a) Global warming, (b) Fossil resource scarcity, (c) Freshwater eutrophication, (d) Freshwater ecotoxicity, (e) Terrestrial ecotoxicity, (f) Terrestrial acidification. In the Figure, BAU = business-as-usual, C5EtOH = pentose based bioethanol, C5−C6EtOH = pentose + hexose based bioethanol, C6EtOH = hexose based bioethanol, CHP = combined heat and power, CWB = cassava wastes biorefinery, GS = glucose syrup, SA = succinic acid.

the C6 sugars for glucose syrup conversion barely increased the GWP (increased by 3.7%) (Fig. 7a). This could be explained by the similar amounts of chemicals, enzymes, nutrients and non-renewable inputs to both scenarios (Nanda et al., 2015), with the minor differences occurring in the ethanol fermentation and CHP operations, such as fermentation chemicals (CSL, DAP) and boiler/cooling tower chemicals (Fig. 7a; Appendix, Table A1). Conversely, the conversion of the diverted C6 sugars to SA, which was modelled in scenario (V) (Fig. 5), increased the GWP by 121% when compared to scenario (III) (Fig. 7a). As shown in the breakdown of the GWP for scenario (V) (Fig. 7a), the SA production section accounted for approx. 64% of the GWP, which is largely due to the high volumes of non-biogenic chemical consumptions, particularly H_2SO_4 (29.44 kg/FU) and NaOH (24.61 kg/FU) in SA fermentation and recovery (Fig. 7a; Appendix, Table A1) (Cok et al., 2014).

In general, the GWP increased with the number of products (Fig. 7a). Interestingly, for all the CWBs, the trend of FRSP was similar to the GWP's (Fig. 7a vs. Fig. 7b), which is corroborated by similar findings for sugarcane biorefineries (Gnansounou et al, 2015, 2017). The similar FRSP trends support assertions that the GWPs are largely due to the fossil based inputs, while for FRSP, the extent of fossil based inputs corresponds with the number of products (Fig. 7b).

4.1.2. Freshwater eutrophication potential (FEP)

FEP refers to excessive nutrient enrichment of freshwater ecosystems with resultant increase in growth of aquatic plants or algae that reduces water quality (Shepherd et al., 2003). Relative to the studied CWBs, nitrogen (N) and phosphorous (P) are the major potential eutrophication nutrients, which could originate from operations such as volatilization of nitrogen based inputs (e.g. NH_3 & DAP in the cellulase enzyme production, and ethanol/SA fermentations), emission of NO_x from combustion units, and release of phosphates from biofuel combustion and ash treatments at landfills (Cherubini and Jungmeier, 2010; Widder et al., 2011).

The CWBs demonstrate potential for substantial reductions in the FEP relative to the base case BAU scenario. The BAU's FEP (0.9 kg P eq) was shown to be 36.92, 37.16, 37.54, 36.79, and 14.25-folds higher than scenarios (I)–(V) respectively (Fig. 7c). From the BAU FEP breakdown (Fig. 7c), open burning of CS accounted for 97% (Fig. 7c), which may be justified by the high air and land emissions due to the absence of treatment of the flue gas and ash (Fig. 2a) (Widder et al., 2011). Furthermore, taking into account the 85% COD removal presumed in the AD simulation (Padi and Chimphango, 2020b), the minimal contribution of the AD biogas-SDHA process to the BAU's FEP (3%, Fig. 7c) could be explained by the relatively low nutrient content of the AD digestate disposed into waterbodies (Fig. 2a).

The extra nutrient emissions in scenario (I), due to the additional power capacity (Appendix, Table A1), is equivalent to the nutrient emissions associated with the integrated GS and/or bioethanol in scenarios (II), (III) or (IV), but ~62% lower than the integrated SA's in scenario (V). From the FEP results (Fig. 7c), equal performances (~0.024 kg P eq) were shown for scenarios (I)-(IV), which increased to 0.063 kg P eq in scenario (V).

Hence, for the BAU, CS burning represents hotspot for FEP, despite there being no value derived from the burning. Therefore, considering the substantial reductions in the FEP for CWBs vs. the BAU (Fig. 7c), the suggested integration of the CS (farm wastes) with the CSF's wastes (CWW + CB) for biorefinery exploits could be a beneficial strategy for value-addition to waste resources while safeguarding against water resource contaminations.

4.1.3. Freshwater ecotoxicity potential (FETP), terrestrial ecotoxicity potential (TETP), and terrestrial acidification potential (TAP)

FETP and TETP relate to the environmental impacts of released toxic materials on freshwater or terrestrial ecosystems respectively, whereas TAP measures the impacts of acidifying pollutants released on land (Bare et al., 2003; Widder et al., 2011). Thus, in addition to the FEP emissions in the CWBs (section 4.1.2), SO_x emissions from the fuel combustions, life cycle of H_2SO_4 (pre-treatment/SA fermentation) (Falano et al., 2014), volatilization of Na_2SO_4 salts (SA fermentation), metals in combustion flue gas or boiler ash, $CaSO_4$ salts from FGD (Figs. 2–4), toxic or acidic compounds such as cyanide & propionic acids in the AD digestate (Appendix, Table A1), which invariably end up in water bodies or land, contribute to the FETP and TETP/TAP respectively (Cherubini and Jungmeier, 2010; Koornneef et al., 2008; Singh et al., 2011).

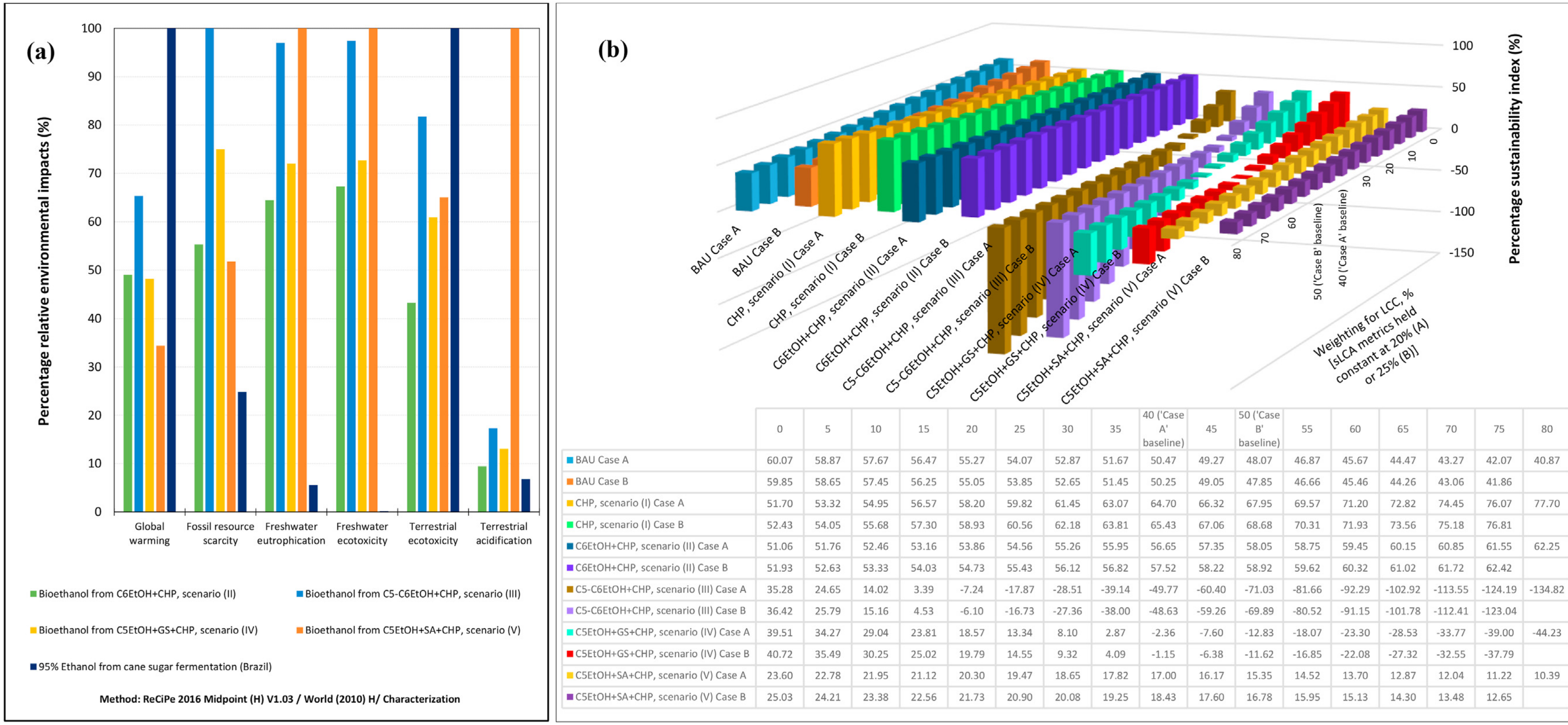
Comparable FETPs, TETPs and TAPs were shown for scenarios (I) vs. (II) and (III) vs. (IV), attributable to the minimal differences in the chemical demands (Appendix,Table A1), as well as the similar approaches to handling process wastes or emissions (Figs. 2–4). The FETP for the BAU, projected at 0.187 kg 1,4-dichlorobutane (DCB) (Fig. 7d), was considerably low compared to the CWBs' values at ~70 kg 1,4-DCB for the (I)-(II), ~64.8 kg 1,4-DCB for (III)-(IV), and ~122 kg 1,4-DCB for (V) (Fig. 7d). In relation to the TETP, compared to the BAU, higher values were shown for the CWBs, which were comparable for scenario (I) vs. (II) and (III) vs. (IV) (Fig. 7e). For the TETP, the BAU's TETP (6.63 kg 1,4-DCB) mainly emerged from the AD biogas-SDHA process (99.7%), with only 0.3% contribution from the open burning of CS (Fig. 7e). In contrast to the FETP and TETP trends, the BAU's TAP (0.547 kg SO_2 eq) was 1.7-folds that of (I)-(II) and 1.4-folds that of scenarios (III)-(IV) (Fig. 7f).

The proposed strategy for GWP mitigations, comprising reduction in transportation distance via cultivation of high CS cultivars (section 4.1.1), could equally minimise the TAPs of the biorefineries. Relevant to the TAP, the substantial contributions from CS transportation, particularly for scenarios (I)-(II) (~35%) and (III)-(IV) (~29%) (Fig. 7f), is imperative to policy designs for TAP mitigations.

4.1.4. Comparing the environmental impacts for the bioethanol production sections

As implied in section 1, biorefinery processes based on edible crops (e.g. cassava) and non-food crops or residues as feedstock (e.g. switch grass, CS) can be classified as first generation (1G) and second generation (2G) biorefineries respectively (Özdenkçi et al., 2017). The 1G is a well-developed technology with widespread commercial applications, such as the sugarcane molasses-based ethanol industry in Brazil (Tao and Aden, 2009). Conversely, 2G biorefineries are generally in development stages (E4tech et al., 2015). The 2G has received considerations over 1G regarding food security impacts (Kim and Dale, 2004). However, the environmental performances for 1G vs. 2G processes are inconsistent, attributable to the diversity in conversion technologies for both processes (Honnery et al., 2013; Özdenkçi et al., 2017). For instance, 1G sugarcane molasses-based ethanol process consists of acid hydrolysis, yeast fermentation & ethanol recovery (Quintero et al., 2012), whereas 2G sugarcane bagasse & trash-based bioethanol consists of pre-treatment/EH, fermentation & ethanol recovery (Farzad et al., 2017).

Therefore, to analyze the environmental potentials of the proposed CWBs vs. the established 1G industries, the environmental impacts of the 2G bioethanol production from the CWBs [i.e. scenarios (II)-(V)] have been compared to the commercial ethanol from molasses fermentation (1G ethanol) in sugarcane biorefineries in Brazil (Fig. 8a) (Ecoinvent, 2018). It was shown that 1G ethanol presents the lowest impacts for FRSP, FEP, FETP & TAP, and vice versa for GWP & TETP (Fig. 8a). Thus, compared to the CWB bioethanol, ethanol from the 1G molasses process is more sustainable for most impacts. However, the potential benefits of substantial reductions in TETP and GWP by the CWB bioethanol is



	0	5	10	15	20	25	30	35	40 ('Case A' baseline)	45	50 ('Case B' baseline)	55	60	65	70	75	80
BAU Case A	60.07	58.87	57.67	56.47	55.27	54.07	52.87	51.67	50.47	49.27	48.07	46.87	45.67	44.47	43.27	42.07	40.87
BAU Case B	59.85	58.65	57.45	56.25	55.05	53.85	52.65	51.45	50.25	49.05	47.85	46.66	45.46	44.26	43.06	41.86	
CHP, scenario (I) Case A	51.70	53.32	54.95	56.57	58.20	59.82	61.45	63.07	64.70	66.32	67.95	69.57	71.20	72.82	74.45	76.07	77.70
CHP, scenario (I) Case B	52.43	54.05	55.68	57.30	58.93	60.56	62.18	63.81	65.43	67.06	68.68	70.31	71.93	73.56	75.18	76.81	
C6EtOH+CHP, scenario (II) Case A	51.06	51.76	52.46	53.16	53.86	54.56	55.26	55.95	56.65	57.35	58.05	58.75	59.45	60.15	60.85	61.55	62.25
C6EtOH+CHP, scenario (II) Case B	51.93	52.63	53.33	54.03	54.73	55.43	56.12	56.82	57.52	58.22	58.92	59.62	60.32	61.02	61.72	62.42	
C5-C6EtOH+CHP, scenario (III) Case A	35.28	24.65	14.02	3.39	-7.24	-17.87	-28.51	-39.14	-49.77	-60.40	-71.03	-81.66	-92.29	-102.92	-113.55	-124.19	-134.82
C5-C6EtOH+CHP, scenario (III) Case B	36.42	25.79	15.16	4.53	-6.10	-16.73	-27.36	-38.00	-48.63	-59.26	-69.89	-80.52	-91.15	-101.78	-112.41	-123.04	
C5EtOH+GS+CHP, scenario (IV) Case A	39.51	34.27	29.04	23.81	18.57	13.34	8.10	2.87	-2.36	-7.60	-12.83	-18.07	-23.30	-28.53	-33.77	-39.00	-44.23
C5EtOH+GS+CHP, scenario (IV) Case B	40.72	35.49	30.25	25.02	19.79	14.55	9.32	4.09	-1.15	-6.38	-11.62	-16.85	-22.08	-27.32	-32.55	-37.79	
C5EtOH+SA+CHP, scenario (V) Case A	23.60	22.78	21.95	21.12	20.30	19.47	18.65	17.82	17.00	16.17	15.35	14.52	13.70	12.87	12.04	11.22	10.39
C5EtOH+SA+CHP, scenario (V) Case B	25.03	24.21	23.38	22.56	21.73	20.90	20.08	19.25	18.43	17.60	16.78	15.95	15.13	14.30	13.48	12.65	

Fig. 8. (a) Relative environmental impacts for 1-ton bioethanol production in the cassava waste biorefineries [i.e. only the scenarios (II)-(V) with bioethanol production sections] vs. 1-ton cane sugar ethanol from a sugarcane biorefinery (economic allocation basis) (Ecoinvent, 2018); (b) Sustainability index projections for the cassava wastes biorefineries for various weightings for LCC/eLCA metrics (0–80% for Case A; 0–75% for Case B) and fixed sLCA weighting (20% for Case A; 25% for Case B). In the figure, BAU = business-as-usual, C5EtOH = pentose based bioethanol, C5–C6EtOH = pentose + hexose based bioethanol, C6EtOH = hexose based bioethanol, CHP = combined heat and power, GS = glucose syrup, SA = succinic acid, eLCA = environmental life cycle assessment, LCC = life cycle costing, sLCA = social life cycle assessment.

imperative for considerations in mitigating climate change impacts of fossil transport fuels. Amongst the studied CWBs, inconsistent trends were shown for the environmental categories, with comparative differences ranging ~90% for the TAP and ~32–50% for all other categories (Fig. 8a). Hence, with the exception of the TAP, the predicted impacts for the bioethanol from the CWBs are fairly comparable (Fig. 8a). Variations in the process approach and economic allocation factors (Table A2) can be cited for the observed differences. For instance, while 1% H_2SO_4 pre-treatment and subsequent NH_3 conditioning of the starch wastes precedes enzymatic hydrolysis in the C5–C6EtOH process (III) (section 2.4), only enzymatic hydrolysis was employed in the C6EtOH process (II) (section 2.3).

4.2. Economic performances of the biorefineries

Compared to the investment costs for the CHP scheme (I), higher (up to 1.84-folds) upfront cost impacts could be projected for the integrations of the CHP with bio-products [(II)-(V)] (Table 2), which could influence CWB choices regarding investment decisions. The BAU demonstrates the least capital investment cost (TCI), while the CWBs' generally increased (up to 1.84-folds) from scenario (I) to (V) (Table 2). Similar trends were shown for the production costs (TPC) (Table 2). The BAU scheme, therefore, presents the lowest investment costs requirements, but at the detriment of limiting the economic potentials for the cassava wastes. Comparing the NPV estimates for the CWBs (Table 2), the scenarios (I)-(II) demonstrate better investment returns than the BAU.

A shift from the BAU to the CWB systems that produce CHP only or with SA and/or bioethanol [i.e. (I), (II), (V)] can help advance industrial growths in the CSIs. The positive NPV projections for (I), (II) & (V) demonstrate their potentials for profitable investment returns and vice versa for (III) & (IV) (Table 2). Coupling profitability with the substantial surplus power generation capacities (~196–363 kW h/FU; Appendix, Table A1) for the (I), (II) & (V), their integrations into CSFs could help overcome the constraints of unreliable energy supplies & costs to the industrial crop prospects for cassava in leading cultivation nations such as Ghana (Kleih et al., 2013) and Nigeria (Nang'ayo et al., 2005).

4.3. Social impact projections for the biorefineries

Collectively, an inconsistent trend was shown for the social impacts vs. the number of products in the CWBs, exemplified by the total sLCA projections (Table 2). The number of job creations correspond with the number of product integrations in the biorefineries [23–69 personnel, from BAU to (V); Table 2], which can be attributed to the matching increase in plant sections (Nieder-Heitmann et al., 2019). Conversely, comparable HTPs were projected for the (I)-(IV) [19.73–20.73 kg 1,4-DB eq; Table 2], with fairly similar contributions from their ash landfill treatments (~54.2%), CS transportation (~31.3%) and CWB inputs/emissions (~14.5%) (SimaPro predictions). This can be explained by the comparable chemical inputs & emissions for the referred CWBs (Table 2) and CS transportation considerations (section 3.2). Pertaining to the CWB's contribution to energy security, all scenarios [(I)-(V)] demonstrate substantial potentials for surplus power generation (~148–363 kW h/FU, Table 2), which decreased by up to ~59% for the scenarios co-producing CHP + bio-products [(II)-(V)] vs. CHP only (I) (Table 2).

4.4. Sustainability of the biorefineries

For all CWB scenarios, both the mutual investor-environmentalist (Case A) and the investor (Case B) stakeholder perspectives result in similar PSI trends with minor differences in magnitude (Fig. 8b), which suggests minimal differences in the impacts of the considered weightings on the PSIs. The Cases A & B baseline scenarios (section 3.3.2) showed comparable sustainability (PSI) rankings for the CWBs, with the predicted best-to-least scenario following the order (I) > (II) > BAU > (V) > (IV) > (III) (Fig. 8b). Additionally, the scenarios (I)-(II) favour the economic sustainability dimension than the environment's, and vice versa for the BAU, (III)-(V) (Fig. 8b). Under conditions of increasing the desired economic performance (i.e. increasing LCC weights) or decreasing the desired environmental performance (i.e. decreasing eLCA weights) (section 3.3.2), the predicted PSIs decreased for the BAU & (III)-(V), and increased for (I) & (II) (Fig. 8b). Therefore, under the context of demarcation of the system boundary at the biorefinery gate, the BAU scenario seemingly presents the best environmental scheme for the cassava starch wastes (Table 2), although with a negative consequence of limiting the economic potentials of the wastes (Table 2). For instance, comparing the NPV estimates in Table 2, the predicted order with regards to decreasing profitability potentials is (I) > (II) > BAU > (V) > (IV) > (III), which suggests scenarios (I) and (II) exhibit better economic incentives than the BAU.

From the TBL sustainability perspective, scenario (I) demonstrates greater incentives with higher economic gains and relatively low environmental impacts, followed by (II) > BAU > (V) > (IV) > (III) (Fig. 8b). Based on the ± magnitudes of the PSIs (Fig. 8b), scenarios (I), (II), BAU and (V) are promising for sustainable industrial expansions in CSIs, while the contrary is presented for scenarios (III)-(IV) (Fig. 8b). Considering the comparable environmental impacts for scenarios (III)-(IV) vs. (II) (Table 2), the non-sustainable tendencies of (III)-(IV) can be attributed to the downward economic performances, exemplified by their negative NPVs (Table 2).

Governmental policies aimed at motivating green power tariffs may be relevant for encouraging near-term adoption of the proposed CWBs (IRENA, 2018). Risks to the sustainability of the CWBs would depend mainly on the derived profitability targets by stakeholders, particularly for scenarios (III) and (IV). For scenarios (III) & (IV), the considered Cases A & B both displayed possibilities to negate the sustainability (Fig. 8b). However, compared to the prevailing BAU scenario, the proposed uses of integrated cassava starch wastes for biorefinery conversions [(I)-(V)] would result in increased environmental savings when the avoided GWP from the equivalent fossil-based products is taken into consideration, thus, enhanced environmental uses of the wastes. For instance, per the Net GWP predictions (Table 2), the projected best-to-least scenarios follows the order (I) > (II) > (IV) > (V) > (III) > BAU. Therefore, taking into account the global interests to support green energy & products towards mitigating environmental impacts of dominant fossil-based alternatives (IEA, 2013; IRENA, 2018), and the substantial bio-products & net power capacities of the studied CWBs (Appendix, Table A1), a promising prospect for tariff supports for sustainable developments of the CWBs can be envisaged.

4.5. Reliability of the sustainability projections and avenues for future improvements

The sensitivity analysis (section 3.3.3) revealed the sub-metric weightings (Sc. 1–14) influence the sustainability rankings for the CWBs, especially the BAU and (I) which could switch positions (Fig. 9). Comparing the PSIs for the examined sub-metric weightings (Sc.1–14) vs. the 'Case A' baseline, the decreasing order of the biorefineries regarding robustness of the PSI to changes in the sub-metric weights was in the order: (V) > (II) > BAU > (I) > (IV) > (III) (Fig. 9). Scenarios (III) & (IV) were the most susceptible CWBs to changes in the sub-metric weights, with possibilities to reverse their sustainability (±PSI) (Fig. 9). Relative to the sustainability

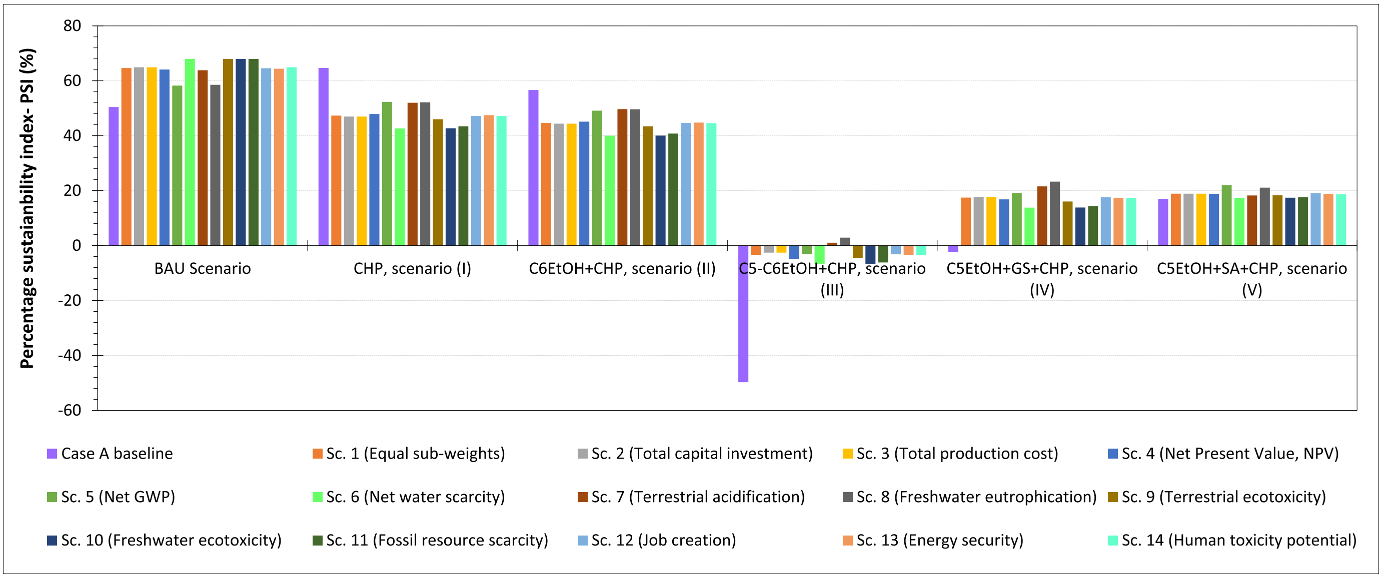


Fig. 9. Sensitivity assessments of the sub-metrics' weighting impacts on the sustainability index projections for the cassava wastes biorefineries. [NB: 'Case A baseline' scenario represents a 40% LCC, 40% eLCA & 20% sLCA weighting perspective, with the sub-weightings depicted in Table 2); Scenarios 1–14 (Sc.1–14) each represents prioritized weightings for the sub-metric (dominant sub-metric) in the bracket (see Appendix, Table A4)]. In the figure, BAU = business-as-usual, C5EtOH = pentose based bioethanol, C5–C6EtOH = pentose + hexose based bioethanol, C6EtOH = hexose based bioethanol, CHP = combined heat and power, GS = glucose syrup, SA = succinic acid, eLCA = environmental life cycle assessment, LCC = life cycle costing, sLCA = social life cycle assessment.

categories (i.e. LCC, eLCA, sLCA), the eLCA sub-metrics' weightings represent the main avenue to uncertainties in the PSIs, especially the TAP and FEP for scenario (III) (Fig. 9). Therefore, the considered sub-metric weightings in the LCSA, particularly the environmental category's, is crucial to the credibility of the estimated PSIs.

Future improvements of the PSI tool may target reliable sub-weight estimates, achievable through consensus building among related experts and stakeholders (Ren et al., 2018), and the use of advanced numerical tools that minimizes uncertainties in the outcomes such as the proposed Non-Linear Fuzzy Prioritization (NLFP) & interval multi-attribute decision analysis method (Ren and Ren, 2018). In addition, the reliability of the PSI tool may be enhanced through the inclusion of other powerful sustainability indicators such as energy efficiency and exergy thermodynamic indicators (Aghbashlo et al., 2018). Juxtaposed to the conventional energy analysis, which only shows how energy flows through a system, exergy analysis further identifies the avenues, magnitudes, and sources of process inefficiencies in energy and material conversion systems (Aghbashlo et al., 2017; Rosen, 2002) such as the biorefinery system. Thus, exergy analysis presents superior thermodynamic performance indicator than the conventional energy analysis, and has gained popularity in sustainability assessments for biorefineries (Aghbashlo et al., 2018; Dogbe et al., 2018). Integrations of the exergy analysis with related economic (exergoeconomic analysis) and environmental impact assessments (exergoenvironmental analysis) has proven useful for advanced and reliable sustainability assessments (Aghbashlo and Rosen, 2018). These sustainability approaches could, therefore, be explored in future sustainability evaluations for the CWBs.

5. Conclusions and future research

Comparative TBL sustainability assessment for CSI's conventional waste management (BAU scenario) and five CWB scenarios [(I) CHP, (II) C6EtOH + CHP, (III) C5–C6EtOH + CHP, (IV) C5EtOH + GS + CHP, and (V) C5EtOH + SA + CHP] has been achieved using a designed PSI estimation tool based on the principles of LCSA. The CWBs present better environmental uses for the wastes vs. the BAU, which could be enhanced by selecting biorefinery products with benign inputs or processing paths. Within the CWB gate boundaries, the environmental impacts generally increase with the number of products. However, allowing for the prospective avoided GHG emissions from the existing fossil-based equivalent products, the CWBs show potentials for higher environmental savings vs. the BAU.

Sustainability of the CWBs depend largely on the targets for the derived environmental or economic performances, with the scenarios (I)-(II) favouring the economic dimension vs. the environment's, and vice versa for the BAU, (III)-(V). Furthermore, positive PSI projections for the (I), (II), BAU & (V) revealed their potentials for sustainable developments in the starch industries, while the contrary was shown for scenarios (III)-(IV) (negative PSIs). The latter's unsustainable tendencies are attributable to the poor economic performances [NPVs, US\$ −1 billion (III) & −388.5 million (IV)]. Hence, considering the potentials for substantial fossil emissions reductions and net power generation by the CWBs, governmental incentives of green power tariffs could enhance economic profitability of the CWBs for near-term applications. Implementation of the CWBs could, therefore, enhance sustainable industrial developments in CSIs.

The PSI tool could, therefore, provide preliminary decision support in the selection of sustainable CWB processes. Future research may focus on improving the reliability of the PSI tool via the incorporation of more dependable sustainability indicators (e.g. thermodynamic exergy), as well as establishing reliable weights for the indicators through stakeholder consensus and advanced numerical tools for minimizing uncertainties (e.g. Non-Linear Fuzzy Prioritization). Alternative reliable sustainability assessments tools (e.g. exergoeconomic & exergoenvironmental analysis) could also be explored for the CWBs.

CRediT authorship contribution statement

Richard Kingsley Padi: Conceptualization, Investigation, Methodology, Formal analysis, Writing - original draft. **Annie Chimphango:** Conceptualization, Funding acquisition, 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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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.jclepro.2020.125171.

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