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Towards a more holistic sustainability assessment framework for agro-bioenergy systems — A review

Oludunsin Arodudu [a,b,*], Katharina Helming [a,d], Hubert Wiggering [a,b], Alexey Voinov [c]

[a] Leibniz-Centre for Agricultural Landscape Research (ZALF), Eberswalder Straße 84, 15374 Müncheberg, Germany
[b] Potsdam University, Institute of Earth and Environmental Sciences, Karl-Liebknecht-Straße 24-25, 14476 Potsdam, Golm, Germany
[c] ITC, Faculty of Geoinformation and Earth Observation, University of Twente, Hengelosestraat 99, Enschede, The Netherlands
[d] Faculty of Landscape Management and Nature Conservation, University for Sustainable Development (HNEE), Schickler Strasse 5, 16225 Eberswalde, Germany

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ABSTRACT

The use of life cycle assessment (LCA) as a sustainability assessment tool for agro-bioenergy system usually has an industrial agriculture bias. Furthermore, LCA generally has often been criticized for being a decision maker tool which may not consider decision takers perceptions. They are lacking in spatial and temporal depth, and unable to assess sufficiently some environmental impact categories such as biodiversity, land use etc. and most economic and social impact categories, e.g. food security, water security, energy security. This study explored tools, methodologies and frameworks that can be deployed individually, as well as in combination with each other for bridging these methodological gaps in application to agro-bioenergy systems. Integrating agronomic options, e.g. alternative farm power, tillage, seed sowing options, fertilizer, pesticide, irrigation into the boundaries of LCAs for agro-bioenergy systems will not only provide an alternative agro-ecological perspective to previous LCAs, but will also lead to the derivation of indicators for assessment of some social and economic impact categories. Deploying life cycle thinking approaches such as energy return on energy invested-EROEI, human appropriation of net primary production-HANPP, net greenhouse gas or carbon balance-NCB, water footprint individually and in combination with each other will also lead to further derivation of indicators suitable for assessing relevant environmental, social and economic impact categories. Also, applying spatio-temporal simulation models has a potential for improving the spatial and temporal depths of LCA analysis.

Contents

1. Introduction

Bioenergy has gained prominence amongst many policy stakeholders in the face of pressing energy security challenges, as well as in search for safer and more renewable energy sources for meeting global climate change mitigation and emission reduction targets (Fischer and

* Corresponding author at: Leibniz-Centre for Agricultural Landscape Research (ZALF), Eberswalder Straße 84, 15374 Müncheberg, Germany.
E-mail address: Oludunsin.Arodudu@zalf.de (O. Arodudu).

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Schrattenholzer, 2001; Berndes et al., 2003; Fischer et al., 2010). However, despite its high ratings amongst decision makers, there are still many arguments, both for and against the proliferation of bioenergy infrastructures and services (Mol, 2007; UNFCCC, 2008; Arodudu et al., 2013). Several findings claim that bioenergy is one of the most effective means for reduction of global crude oil dependencies, and turning back climate change and global warming trends, through the replacement of fossil fuels and the reduction of greenhouse gas emissions (Dincer, 1999; Bertil et al., 2004; Arodudu et al., 2013). Conversely, some other findings show that bioenergy will compete indiscriminately with other important biomass supply chains including food, animal feed, industrial raw materials (Groom et al., 2008; Searchinger et al., 2008), and put further pressure on ecosystem services (Wiens et al., 2011; Koizumi, 2013). Some also claim that bioenergy will contribute to the greenhouse gas emission levels through discharges from indirect fossil energy investments across the bioenergy production chain, e.g. in production of synthetic fertilizers, pesticides, lime (Pimentel, 2003; Hill et al., 2006; van Duren et al., 2015). Current uncertainties and diversities of opinions generate debates that require holistic sustainability assessments, in order to find pathways that will orientate policy making regarding bioenergy production towards sustainability (Ness et al., 2007; Helming et al., 2011).

With respect to agriculture based bioenergy (agro-bioenergy) specifically, its sustainability has often been brought into question by its overall high energy requirements, low net energy gain and efficiency, positive net greenhouse gas emission status, and the high water footprint associated with production across the value chains (Groom et al., 2008; Searchinger et al., 2008; Gerbens-Leenes et al., 2009). Even though this is true within the contexts previously considered, most studies that have come to these conclusions assume bioenergy production to be fossil fuel dependent, energy intensive, commercially focused and essentially a product of industrial agriculture settings. Industrialized agriculture system, which is widespread in most parts of the US and Europe, favours big farm holds and large expanses of land (Patzek, 2004; Pimentel et al., 2009). It also supports the pushing of the boundaries of agricultural production for profit by whatever means possible (Hall et al., 2011; Murphy et al., 2011) even at the expense of the degradation of the environment. Examples include precision irrigation-sprinkler irrigation, drip irrigation etc., increased synthetic fertilizer and pesticide application, use of improved seeds-hybrid cultivars, genetically modified cultivars, deployment of heavy machineries and more rigorous tillage techniques (Altieri et al., 2012; Altieri et al., 2015).

An alternative to the industrial agriculture that is not usually considered within most sustainability assessments for agro-bioenergy system is the ecological agriculture or the agro-ecological production (Chappell and LaValle, 2009; Blesh and Wolf, 2014). It is widespread in Cuba, Chile and most parts of the Latin America (Wittman, 2009; Aerni, 2011). Ecological agriculture advocates degrowth and decarbonization principles such as small scale production on small fragmented land holdings that are owned by rural communities and cooperatives. This prevents local community holders from becoming landless. It is also characterized by shorter transport distances (usually less than 20 km), deployment of smaller tractor implementations (single axle tractors) or human or farm animal (e.g. ox, buffalo, horses, donkeys, mules, camels etc.) labour (Smith, 2009; Wezel et al., 2009). Agro-ecological systems also encourages management using conservation practices and principles such as reduced or no tillage operations, use of mostly native seeds, as well as less energy intensive and more organic fertilizers, limes and pesticides sourced from agricultural waste sources (e.g. manure, biogas digestates etc.) (Altieri et al., 2012; Altieri et al., 2015).

In this study, we reviewed and suggested methodologies that could be adapted within local and regional sustainability assessment framework for assessing agro-bioenergy system from an agro-ecological point of view. This is expected to provide more balanced perspective of the sustainability of agro-bioenergy systems.

However, aside the need for an agro-ecological perspective within sustainability assessment frameworks for agro-bioenergy systems, assessment methodologies usually do not answer all the questions needed for more accurate decision making. In order to bridge this methodological gap, this study first suggested methodological improvements to some of the previously used tools and methods, before discussing those specifically related to sustainability assessment of agro-bioenergy systems from an agro-ecological point of view. Section 2 reviewed the structure of what a holistic sustainability assessment framework should look like and listed impact categories relevant for agro-bioenergy systems. Section 3 and its subsections reviewed how life cycle assessment (LCA) fits into the mould of a holistic sustainability assessment framework; identified its currents weaknesses as a holistic sustainability assessment framework; and suggested improvements that can assist in bridging the methodological weaknesses identified. While Sections 3.1.1 and 3.1.2 focused on methodological improvements that are applicable to agro-bioenergy systems both from an industrial agriculture and agro-ecological point of view, Section 3.1.2 focused on methodologies for assessing agro-bioenergy systems from an agro-ecological point of view only.

2. Sustainability assessment framework for assessing agro-bioenergy systems

Since sustainability assessment is a process that aims at directing management and policy making towards sustainability, it requires answering specific what- (impacts), where- (space), when- (time) and who- (stakeholders) questions (Ness et al., 2007).

Depending on the level of detail required, the sustainability questions requiring answers could be further divided into local, regional and global for the space dimension of the sustainability assessment with local factors being part of regional and global processes or global factors influencing a system at regional or local scales (Voinov, 2008; McLellan et al., 2014; Nyerges et al., 2014). Time dimension of sustainability assessment can be divided into short, mid and long term for the time element of the sustainability assessment (Filar et al., 2009; Handoh & Handoh and Hidaka, 2010). Stakeholder dimension of the sustainability assessment can be split into decision maker and decision taker sub-divisions (Lahdelma et al., 2000; Mendoza and Martins, 2006). The impact dimension of the sustainability assessment can be delineated into environmental, social and economic impacts (Morris et al., 2011; den Herder et al., 2012). This is illustrated in Fig. 1 (Arodudu et al., 2017).

Defining and measuring sustainability by answering these questions require the use of appropriate indicators that are systemic with respect to the different impact categories concerned, sensitive to the impacts of policy or activity examined over space and time, and reflective of different stakeholder group points of view (Helming et al., 2011). We have identified from literatures a cross section of impact categories that are relevant to agro-bioenergy systems, across the three major sustainability impact divisions namely environmental, social and economic impacts (Fig. 2).

3. Extending LCA for sustainability assessment of agro-bioenergy systems

Life-cycle assessment typifies the group of tools that account for the flow of inputs and outputs of energies and materials accompanying all the different stages of a product's life (EPA, 2006; Firrisa et al., 2014). Even though LCA studies have general standards like the ISO 14040:2006 and ISO 14044:2006 (ISO, 2006a; ISO, 2006b), which practitioners often refer to as a guide, each LCA still has its own unique boundaries and settings reflecting the goals or questions in focus (Weidema, 2009; Wolf et al., 2012). The life cycle of agro-bioenergy system is from raw material extraction (plant cultivation and harvesting inclusive) through material processing, manufacture, distribution, use,

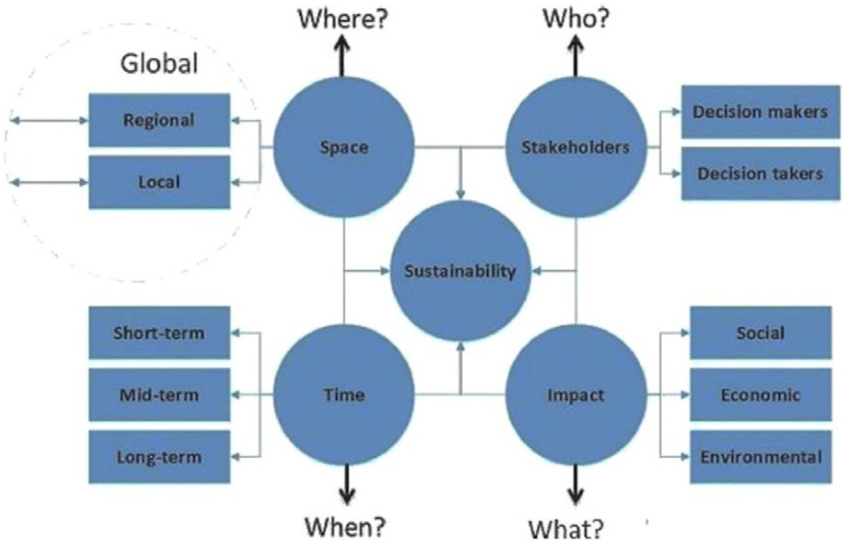

Fig. 1. The basic elements and questions of sustainability assessment processes (Arodudu et al., 2017)

repair/maintenance and disposal/recycling (Kim and Dale, 2005; Čuček et al., 2011). The deployment of LCA within sustainability assessment framework for assessing several production cycles, albeit relevant still has several weaknesses, which are currently being explored under an emerging research field called Life cycle sustainability assessment (LCSA) (Loiseau et al., 2013; Zamagni et al., 2013). This was assessed under this study with respect to agro-bioenergy systems.

Aside of being an environmental impact assessment tool (Canals et al., 2007; Koellner et al., 2013), LCA is mostly a decision maker tool, as decision takers views (mostly social and economic impacts) are usually not factored in (Mulder and Hagens, 2008; Hall et al., 2014). This is because they are often hard to quantify. Although not a locational tool like GIS or remote sensing, LCA often assesses environmental impacts based on local or regional conditions (Herrchen, 1998; de Baan et al., 2013). LCA also has the weakness of being static and therefore not able to predict mid-term to long term changes in impacts considered (Reap et al., 2003; Ness et al., 2007). Retrospectically speaking, deploying LCA as a sustainability assessment tool will only address decision maker's questions concerning local and regional environmental impacts in the short term. This implies that it covers only five out of the ten sub-elements of sustainability assessment described in Fig. 1. This is illustrated in Fig. 3.

Even though traditional LCA presently has established protocols (e.g. ISO 14040:2006 and ISO 14044:2006, CML-Guinée et al., 2002; Eco-indicator 99-Goedkoop and Spriensma, 2001; TRACI-Bare et al., 2003; EDIP2003-Hauschild and Potting, 2005) for evaluating several relevant environmental impact categories, most of them are either pollution or toxicity based (EPA, 2006; de Souza et al., 2015). These include global warming, terrestrial ecotoxicity, freshwater and marine aquatic ecotoxicity, eutrophication, acidification, stratospheric ozone depletion, photochemical oxidation or photochemical ozone (smog) formation etc. (Loiseau et al., 2013; Cabal et al., 2005). Noteworthy is the fact that environmental impact categories important within the agro-bioenergy contexts such as biodiversity and land use efficiency are still not satisfactorily addressed by LCA (de Baan et al., 2013; Koellner et al., 2013). Furthermore, LCA does not yet address most social and economic impact categories associated with agro-bioenergy systems such as the food, water and energy security nexus, job provision, quality of life, sustainable livelihood, sustainable production/consumption, resilient economic support base etc. (Mulder and Hagens, 2008; Hall et al., 2009). Consequently, in this study we have focused on methodological improvements that can compensate for the current weaknesses of LCA as a tool within sustainability assessment framework for assessing agro-bioenergy production system.

3.1. Proposed methodological improvements within LCA frameworks for sustainability assessment of agro-bioenergy systems

This section focused on methodological improvements proposed for LCAs intended for sustainability assessment of agro-bioenergy systems. Section 3.1.1 focused on methodologies for handling spatial and temporal deficiencies of LCAs within sustainability assessment framework for agro-bioenergy systems, Section 3.1.2 focused on life cycle thinking (LCT) indicators for measuring sustainability impacts relevant for agro-bioenergy systems, while Section 3.1.3 focused on indicators for quantifying stakeholder's social and economic impact categories in LCA.

3.1.1. Handling spatial and temporal deficiencies of LCAs within sustainability assessment frameworks for assessing agro-bioenergy systems

Spatial and temporal deficiencies in LCA can be addressed in a number of ways. While remote sensing/GIS based spatial information, and spatially explicit models (e.g. CLUE framework etc.) can be used to answer local and regional spatial questions associated with LCA (Verburg and Overmars, 2009; Loiseau et al., 2013); time slices of remote sensing information (i.e. multi-temporal imageries) and spatio-temporal models (e.g. LPJ ml, EPIC, Miami etc.) can answer more questions spatiotemporally (i.e. over space and time) (Los et al., 2006; Los, 2015). Even though the use of spatio-temporal models often comes with uncertainties that is usually created by underlying assumptions, many of such models have proven to be useful tools in simulating energy (e.g. solar radiation fluxes, fossil fuel etc.), nutrients (e.g. N, P, K, Ca etc.) and material fluxes (e.g. carbon, water, biomass etc.) within natural and human environment, over space and time (Bondeau et al., 2007; Balkovic et al., 2014). The impacts of nature-induced and human-induced events on the environment and society over space and time, has often been predicted and estimated using spatio-temporal models (Guinée et al., 2002; Nyerges et al., 2014). Nature-induced events are biophysical in nature e.g. global climate change, drought, sea level rise, flooding etc.; while human induced events are socio-economic in nature e.g. food security, peak oil, peak phosphorus, renewable energy targets etc. Information (parameter result estimates) obtainable from these models (e.g. N and P outflows from fields, change in soil organic carbon in soil etc.) are often needed within the LCA framework to supply information on energy, nutrients and material fluxes involved in production processes (Azevedo et al., 2013; van Goethem et al., 2013). On the other hand, such spatiotemporal information can deepen the spatial and temporal contexts of LCAs, as often

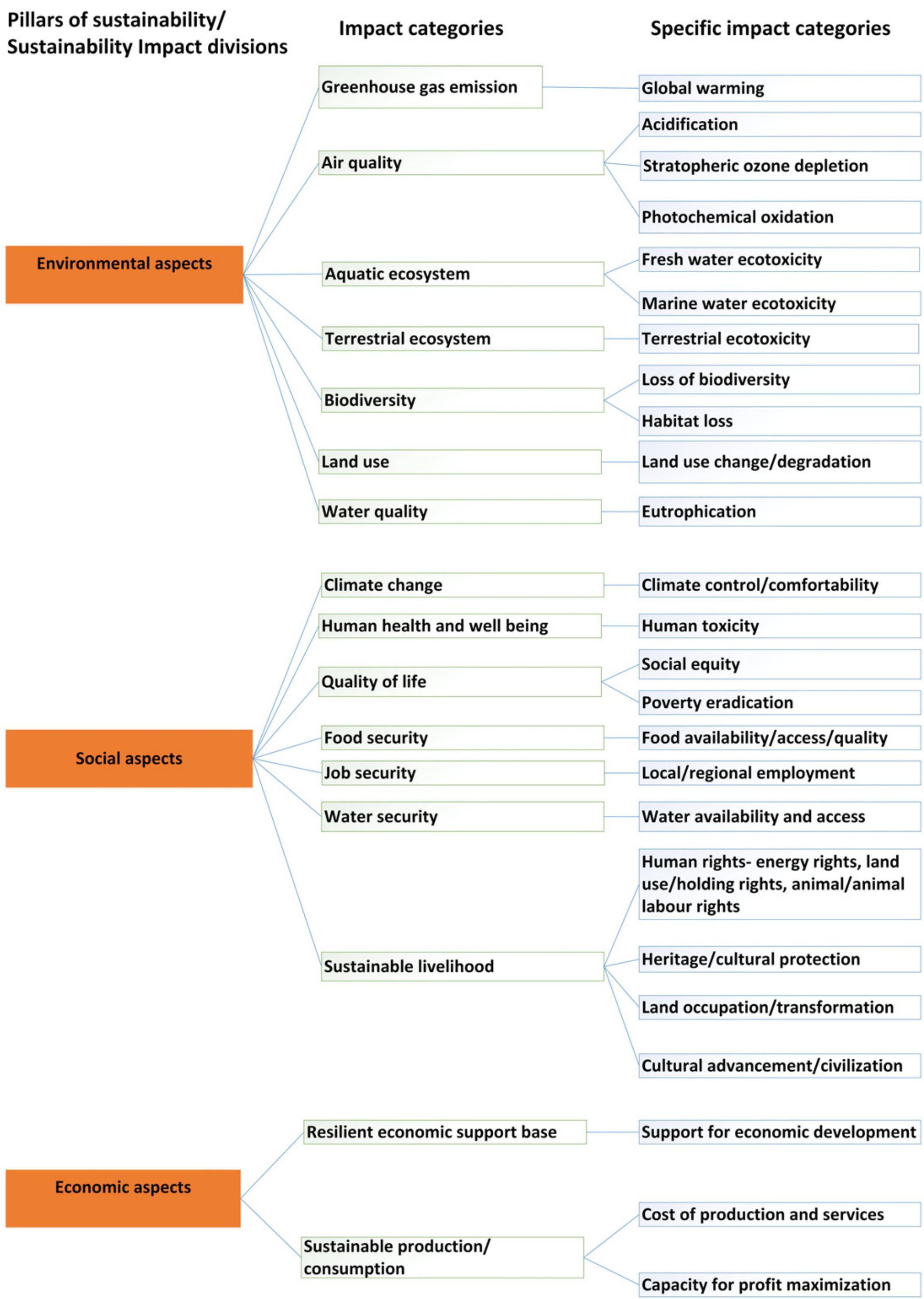

Fig. 2. Environmental, social and economic impact categories of agro-bioenergy systems (Cabal et al., 2005; Solomon, 2010; GBEP, 2011; Bizikova et al., 2013; Loiseau et al., 2013; Fetzel et al., 2015; Lambert et al., 2014)

required by policy stakeholders within decision making processes (Apfelbeck et al., 2007; McLellan et al., 2014).

Deriving the appropriate LCA parameters from such models requires careful spatial and temporal aggregation. While LCA is usually static and spatially homogeneous, information from spatio-temporal models are spatially distributed and changes in time. Appropriating such spatio-temporal information within an LCA will require precautionary principles such as choosing extreme parameter values (maximal or minimal), variances or central tendencies (using average or median values) across space and time; Or making the whole LCA spatial and dynamic? That may require a lot more complicated calculations, and, as a result, more uncertainty. But then perhaps it is not that we are adding uncertainty, but rather it is that we are making it more explicit, since in reality all the spatial and temporal variability is present, and we simply choose to ignore it in our standard LCA applications (Weidema, 2009; Mattila et al., 2012).

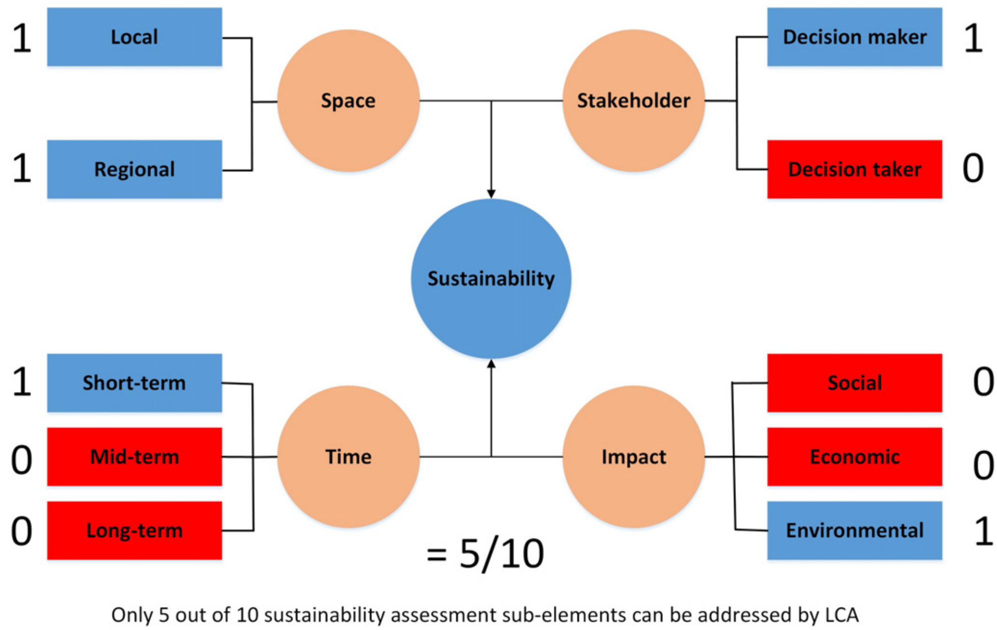

Fig. 3. Graphical illustration of the strengths and weaknesses of LCA as a sustainability assessment framework.

3.1.2. Life cycle thinking (LCT) indicators for measuring sustainability impacts of agro-bioenergy systems

In measuring the different sustainability impacts of a production system, LCA methodologies often take inventory of the flow of substances (i.e. energies, materials and releases) involved, with the aim of investigating their possible impact on the environment, society and economy (Zamagni et al., 2013; de Souza et al., 2015). However, environmental impact categories such as biodiversity and land use, as well as majority of social and economic impact categories are yet to receive adequate attention within LCA frameworks, as well as other sustainability assessment methodologies for assessing agro-bioenergy systems (FAO, 2012; Chen et al., 2014). To create LCA templates for assessing these impact categories, we evaluated the applicability of related life cycle thinking approaches/concepts i.e. non-traditional, life cycle approaches or concepts that incorporate the consideration of all activities or stages leading to the production of an end-product in decision making. Also, we examined the possibilities that could be associated with coupling different life cycle thinking approaches for the derivation of potential indicators, which might be well-suited for assessing impact categories relevant within sustainability assessment framework for agro-bioenergy systems.

Since the major substance flows of the agro-bioenergy production cycle are energies, greenhouse and non-greenhouse gases (greenhouse gases are however very relevant for bioenergy and climate change mitigation functions), water and biomass (Hill et al., 2006; Gerbens-Leenes et al., 2009; Hall et al., 2011), we took a look at life cycle thinking approaches that account for these flows and attempted to connect them with different impact categories important within the context of sustainability assessment of agro-bioenergy systems. The most well-known life cycle thinking approaches reviewed by this study included the surplus energy or the energy return on energy invested-EROEI, which accounts for energy flows across the production chain; the net greenhouse gas or net carbon balance-NCB, which accounts for the flow of greenhouse gases across the production chain; the water footprint, which accounts for the flow of water across the production chain; and the human appropriation of net primary production-HANPP, which accounts for flow of biomass across the production chain.

- Surplus energy or energy return on energy invested (EROEI) framework

This indicator framework is an LCT based energy input-output analysis framework which attempts to account for all the energy inputs and energy outputs of the chains of activities involved in energy production, in order to determine the net or surplus energy that can be made available for societal benefit (Mulder and Hagens, 2008; Voinov et al., 2015). It can be estimated in two major ways namely:

(i) As net energy gain (NEG) or net energy balance (NEB) or simply as net energy (NE). This is estimated by subtracting the total energy input into an energy production activity (across all the value chain involved) from the total energy output from that particular energy production activity (Hill et al., 2006; Arodudu et al., 2013). The units of NEG or NEB is usually in Joules (J) or Joules/hectare (J/ha) cultivated or Joules/tonnes (J/t) of biomass used, depending on the reference system used. Net energy gain is expressed as follows in Eqs. (I) and (II):

$$\text{NEG (J or J/ha or J/t)} = \text{Energy Output (J or J/ha or J/t)} - \text{Energy input (J or J/ha or J/t)} \quad \text{(I)}$$

(ii) As energy return on energy invested (EROI or EROEI), which is the ratio of total energy output to the total energy input of a particular energy production chain under investigation (Grandell et al., 2011; Hill et al., 2006). It has no unit but is rather the fraction of energy obtainable from an energy production activity. Energy return on energy invested (also known as the energy efficiency) is expressed as follows:

$$\text{EROEI (no units)} = \text{Energy output (J or J/ha or J/t)/Energy input (J or J/ha or J/t)} \quad \text{(II)}$$

NEG and EROEI accounts for the capacity or resilience of an energy provisioning source or services to withstand continuous socio-economic function and confer net benefit to the society, even in the face of externalities such as soil erosion and toxicity, ground and surface water pollution, loss of habitat, loss of food production capacity, provision or loss of jobs, improvement or decline of rural economy etc. (Hall et al., 2009; Arodudu et al., 2013). They are therefore a measure of the energy security of an energy source, i.e. an indicator of how much we could rely on an energy source to provide continuous socio-economic function to a society or an economy (Cleveland, 2005; Hu et al., 2013).

Much more than conventional LCA, recent developments within the EROEI framework extend its previous boundaries to include additional requirements and preferences (Grandell et al., 2011; Brandt et al., 2013). Energy consumed in terms of quantity and price value has direct relationship with economic impact categories important for establishment of agro-bioenergy systems such as the cost of production or cost of doing business, profit level taken by producer, general price level within an economy, cost of services procured by consumers etc. (Gagnon et al., 2009; Freise, 2011; Grandell et al., 2011). EROEI on the other hand has direct relationship with energy consumed, hence giving room for quantification of social and economic impact categories measurable in terms of money cost of energy consumed (Hall et al., 2009; Hu et al., 2013; Poisson and Hall, 2013). Example of such social impact category is the quality of life (in terms of wealth or poverty level of a nation) (Hall and Klitgaard, 2012; Lambert et al., 2014). Previous EROEI analyses have reinforced the notion that the discovery of new energy sources (e.g. from agro-bioenergy systems) leads to the emergence and sustenance of new forms of civilizations as seen in the economic boom and cultural evolution that followed the industrial revolution of the 18th and 19th Centuries (Hall et al., 1986; Guilford et al., 2011). This is also expected with the advent of agro-bioenergy and other renewables. EROEI studies have shown that a decline in the EROEI of energy deliverable to the society (e.g. from agro-bioenergy systems) will affect continuous and steady supply of energy i.e. economic support base (Hall et al., 2008; Freise, 2011), lead to an increase in the general cost of production and services, reduce the ease of doing business, limit the capacity to maximize profit within an economy, and eventually precipitate an economic collapse, since energy is the building block of every economy (Hall et al., 2014). This is why the collapse of the former civilizations (e.g. the Roman Empire) coincided with a period of declining EROEI occasioned by low grain yield as a result of loss of fertility, soil erosion and other degradation activities (Tainter, 2003; Hall and Klitgaard, 2012).

Energy (in Joules) is however preferred to monetary prices as overall units within the EROEI framework. This is because monetary prices are very volatile and susceptible to fluctuations. Like traditional LCA impact categories with normalized working units, e.g. kg of CO_2 equivalent for global warming, kg of SO_2 equivalent for acidification etc. (Cabal et al., 2005; EPA, 2006), within the EROEI framework the monetary data and other forms of information deployed are normalized into energy equivalent (mostly in Joules) (Hall et al., 2009; Lambert et al., 2014). In order to account for temporal nature of sustainability impacts assessed, EROEI analyses often deploy available historical and economic data together with energy consumption and output data (Hall et al., 1979; Gagnon et al., 2009).

- Net greenhouse gas or net carbon balance (NCB) framework

Net greenhouse gas (or net carbon) balance indicator framework is an LCT based indicator framework devoted to accounting for greenhouse gases released into the environment or society through production activities (De Oliveira et al., 2005; Hill et al., 2006). Embodied carbon that is stored or sequestered by materials or mediums (e.g. soils or living/harvested biomass etc.) within the production chain is also considered (depending on the scope and system boundaries defined) (McBratney and Stockmann, 2011; Follett et al., 2012). Total soil organic carbon (usually in t C/a or kg C/a), that is added annually as a result of planting an energy crop is usually accounted for within the net greenhouse gas balance of agro-bioenergy systems (IPCC, 2014; Tilman et al., 2006). Within agro-bioenergy systems, embodied carbon in the harvested biomass is not included in the inventory because it is released back to the atmosphere as the biomass is being denatured or burned up. However, carbon saved from the substitution effects of co-products of agro-bioenergy production (e.g. biogas digestate from biogas production process) may be included as avoided emissions depending on the defined system boundary being investigated (Dale et al., 2010; Wiens et al., 2011).

Advancement over the years within the greenhouse gas accounting framework has led to the normalization of all greenhouse gases relevant for the global warming potential (GWP) impact category to single normalized units of tonnes or kilogram of carbon dioxide or carbon equivalent (t or kg CO_2, OR t C or kg C) (Cabal et al., 2005; Gelfand et al., 2013). The GWP characterization or conversion factors for the three greenhouse gases with significant global warming potentials are 1, 21–25, and 265–310 for carbon dioxide-CO_2, methane-CH_4, and Nitrous oxide-N_2O gases, respectively, during a period of 100 years (in normalized t or kg of CO_2 or C units) (EPA, 2006; IPCC, 2014). Total soil organic carbon is usually adjudged negative when there is a net gain in total soil organic carbon, as a result of the cultivation of an energy crop during the planting season or year (McBratney and Stockmann, 2011; IPCC, 2014). It is however adjudged positive when there is a net loss in total soil organic carbon, as a result of the cultivation of an energy crop during the planting season or year (Follett et al., 2012; IPCC, 2014). This implies that a net loss in total soil organic carbon is characterized in the same way as other life cycle greenhouse gas emissions within the agro-bioenergy system (Hill et al., 2006; IPCC, 2014). Estimating net greenhouse gas or carbon balance for agro-bioenergy systems, in terms of normalized kg CO_2 equivalent per annum units (kg CO_2 equivalent/a) can be described as follows in Eq. (III):

$$\begin{aligned}\text{NCB (kg CO}_2\text{ equivalent/a)} = {} & \text{Total greenhouse gas emissions (kg CO}_2\text{ equivalent/a)} \\ & -\text{Total soil organic Carbon (kg CO}_2\text{ equivalent/a)} \\ & -\text{Greenhouse gas saved as a result of substitution effects of co} \\ & -\text{products (kg CO}_2\text{ equivalent/a)}\end{aligned} \tag{III}$$

Net carbon balance can therefore be an indicator for assessing greenhouse gas emission reduction and/or climate change mitigation efforts of an agro-bioenergy system.

- Water footprint (WF) framework

Water footprint indicator framework is an LCT based framework for accounting for the different forms of water consumed during the different stages of the production of a good or service (Pfister et al., 2009; Boulay et al., 2011). Such water may be consumed directly during the production stage or indirectly during the production of other materials or even embodied in energy used for the production of the product under consideration (Gerbens-Leenes et al., 2009; Mekonnen and Hoekstra, 2010). Water consumed during the production of a good or service may be in form of fresh water from surface and groundwater sources (also known as blue water), OR atmospheric water from rain and evaporating water and plant bodies (also known as green water) (Hess, 2010; Mekonnen and Hoekstra, 2011). More than traditional LCA does, water consumed for remedying (dissolving or diluting) pollutants generated during production processes is classified under a different category within the water footprint framework (also known as grey water) (Gerbens-Leenes et al., 2012; Hoekstra and Mekonnen, 2012).

In order to account for water provision and security as an impact category within a sustainability framework for agro-bioenergy systems, we must recognize the fact that indirect water for production of fossil fuel and electricity consumed, irrigation water and remediation water (grey water demand) places a sizable demand on the earth's limited fresh (blue) water reserves and by extension availability (or provision) for other human domestic and industrial uses (Mielke et al., 2010; Vanham et al., 2013). Green water supplies for crop evapotranspiration and growth may be assumed to be free from nature and a product of many other sources other than fresh water reserves, hence its exclusion from the framework. In estimating the net blue water balance, water saved as a result of substitution effects of co-products may be subtracted from the total blue water footprint, depending on the system boundaries of the study. Based on these assumptions, the net blue water balance (NBWB) of agro-bioenergy production system relative to other equally important domestic and industrial uses can be a normalized indicator for assessing water provision impact category of agro-bioenergy systems. This is described as follows in Eq. (IV):

$$\begin{aligned}\text{NBWB}\ (\text{m}^3\ \text{of water}) = {} & \text{Indirect water consumed for fossil fuel use}\ (\text{m}^3\ \text{of water}) \quad (\text{IV})\\ & +\text{Indirect water consumed for electricity consumed}\ (\text{m}^3\ \text{of water})\\ & +\text{Water consumed for irrigation (if applicable)}\ (\text{m}^3\ \text{of water})\\ & +\text{Water consumed during conversion process}\ (\text{m}^3\ \text{of water})\\ & +\text{remediation or grey water demand for dilution of pollutants}\ (\text{m}^3\ \text{of water})\\ & -\text{Water saved as a result of substitution effects of co-products}\ (\text{m}^3\ \text{of water})\end{aligned}$$

- Human appropriation of net primary production (HANPP) framework

HANPP framework is an LCT approach that has previously being deployed to account for the effect of human activities on different ecosystems, agricultural ecosystems inclusive (Haberl et al., 2005; Gavrilova et al., 2010). It is mostly a social metabolism concept for biophysical and socio-economic accounting of biomass flows between nature and society over space and time (Erb et al., 2009; Fischer-Kowalski et al., 2011). HANPP is an economy-wide biomass inventory framework which delineates what is taken away from nature, as against what is retained by and/or returned back to the nature, as well as what is received and consumed by the society, all within the context of discussing complex sustainability challenges associated with agro-bioenergy systems (Rojstaczer et al., 2001; Fetzel et al., 2015).

Different components of the HANPP framework) account for different nature-society interactions over stipulated space and time frames (Vitousek et al., 1986; Lauk and Erb, 2009). Within agricultural or agro-bioenergy systems, HANPP components may be measured in kg or tonnes of biomass per annum (kg/a) or kg or tonnes of biomass per hectare per annum (kg/ha/a). The accounting system of HANPP is described in Figure 4 and the following Eqs. V to IX.

Potential NPP (NPPpot) for annual crops accounts for maximum potential biomass yield under optimum natural or agro-ecological conditions, while actual NPP (NPPact) accounts for actual biomass yield obtainable (comprising of yield and residue) after certain management practices must have been implemented (Haberl et al., 2011; Krausmann et al., 2012). Aboveground NPP (aNPP) accounts for the standing shoot, while belowground NPP (bNPP) accounts for the underground root system (Ajtay et al., 1979; Niedertscheider et al., 2012). Human appropriated NPP (HANPP) accounts for harvested or used biomass (harvested grain-HG and extracted residue-ER) and biomass yield lost or gained as a result of management practices adopted (ΔNPPlc or HANPPluc) (Haberl et al., 2009; Scarlat et al., 2010); while ecosystem NPP (NPPeco) accounts for the biomass remaining or left on the ecosystem after harvest (Erb et al., 2009; Krausmann et al., 2012), this may include harvest losses (HL), belowground NPP-bNPP, which are approximately 15% of harvested grain and 13% of actual NPP on cropland respectively (Ajtay et al., 1979; Rojstaczer, 2001). Unused extraction or backflows to nature (UnE) is usually the unextracted, post-harvest residue left on the field, while used extraction (UE or HANPPue) comprise of harvested grain and extracted residue (Haberl et al., 2012; Fetzel et al., 2015). HANPPharvest refers to the biomass utilized during harvest period; it comprises essentially of extracted biomass (UE or HANPPue) taken off field and those left on the field post-harvest (UnE) (Haberl and Geissler, 2000; Bais et al., 2015).

Unlike traditional LCA that is spatially and temporally deficient, HANPP deploy spatio-temporal models for addressing spatial and temporal questions. A change in NPPpot over space and time is also important indicator of what is obtainable for appropriation under optimum agro-ecological conditions, especially within sustainability discuss involving food security, energy security (bioenergy forming an integral part) and transitions to bio-economy, as well as their associated ecosystems services (namely food, energy and raw material provision) (Haberl, 2001; Niedertscheider et al., 2012). This is because it represents the maximum amount of biomass obtainable from nature for supplies to this functions or services per time (Vitousek et al., 1986; Haberl et al., 2010). NPPeco on the other hand could be an indicator accounting for how much is left for nature to sustain the food chain of primary consumers and decomposers, as well as for the stabilization and protection of soils from erosion activities (Haberl et al., 2004; Haberl et al., 2005). NPPeco could however as well be an indicator of an additional source of biomass for meeting future animal feed, bioenergy and industrial raw materials demands within the context of sustainability discussions

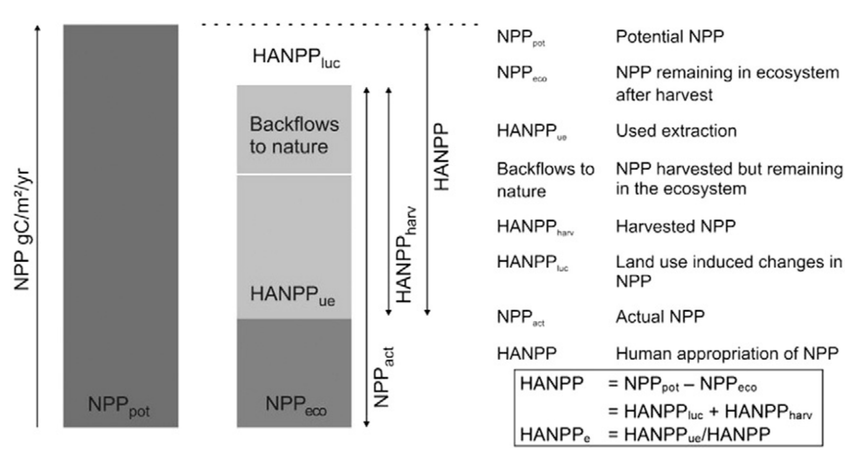

Fig. 4. Figure showing the different components of the HANPP framework (Source: Fetzel et al., 2015).

$$\text{NPPpot} = \text{NPPact} + \Delta\text{NPPlc (or HANPPluc)}$$
$$\text{NPPharvest} = \text{HG} + \text{ER}$$
$$\text{HANPP} = \text{UE} + \Delta\text{NPPlc OR HANPP} = \text{NPPharvest} + \Delta\text{NPPlc (or HANPPluc)}$$
$$\text{NPPact} = \text{aNPP} + \text{bNPP}$$
$$\text{NPPeco} = \text{UnE OR NPPeco} = \text{UnE} + \text{HL OR NPPeco} = \text{UnE} + \text{HL} + \text{bNPP (depending on the system boundaries and contexts)}$$

regarding food security, energy security and transition to bio-economy (or substitution of depleting raw material reserves with biomass) (Scarlat et al., 2010; Haberl et al., 2011). Within the context of future agro-bioenergy systems, NPPeco represents a trade-off indicator between ecosystem maintenance and additional bioenergy supply, or a trade-off indicator between ecosystem maintenance and food supply (if used as animal feed) (Lauk and Erb, 2009).

Interactions between different HANPP components can be relative indicators that help answer questions related to the use of biomass for meeting future food, energy and material security needs. Biomass production efficiency (NPP Peff) measured as NPPact (actual NPP)/NPPpot (potential NPP) could be an indicator for the potential of an area (or region) to reach maximum potential biomass yield under future food scarcity (or security) events (Krausmann et al., 2003; Krausmann et al., 2013). Biomass production efficiency can therefore be a relative indicator for measuring the food security impact category in relations to the use of agro-biomass for bioenergy production. This is because increased production efficiency could make more harvested grain available for meeting food security demands and/or energy security demands on the one hand; as well as more unextracted and/or reusable extracted residues available for meeting energy security demands.

NPP (or biomass) use or appropriation efficiency (NPPE Ueff) measured as UE (unextracted residue) / NPPact (actual NPP) or HANPPus (unextracted residue) / NPPact (actual NPP) could be an indicator of what is left or available to be used for meeting emerging biomass demands for animal feed, bioenergy and industrial raw materials, especially within the context of sustainability discussions centred on food security, energy security and transition to bio-economy (Krausmann, 2001; Erb et al., 2012). Biomass appropriation or use efficiency will be a relative indicator for measuring the energy security impact category of agro-bioenergy systems, because it will be a fair measure of the relative availability of biomass resources for meeting emerging energy security demands.

Land use or management efficiency (HANPPe) measured as UE / HANPP or HANPPue / HANPP could be an indicator of human efficiency at appropriating NPP or biomass, considering the management practices it employs (Haberl et al., 2010; Niedertscheider and Erb, 2014). In the event of high demand for bioenergy from agricultural sources, land use efficiency can also be a relative indicator for measuring the energy security impact category of agro-bioenergy systems. However, the difference between biomass use efficiency and land use efficiency is that while biomass use efficiency estimates actual biomass available for meeting energy security and other needs, land use efficiency estimates potential biomass available for meeting the energy security needs under different land use management conditions.

- Coupling Life Cycle Thinking (LCT) indicators

Coupling HANPP methodology with other LCT approaches (namely EROEI, NCB and water footprint) for the sustainability assessment of agro-bioenergy systems, entails conducting a three-prong LCA inventory of the energy, greenhouse gas and water flows involved in extracting the three major HANPP components relevant for bioenergy production (namely HG-harvested grain, ER-extracted residue and UnE-unextracted residue). The coupling process involves the deployment of relevant conversion factors for the life cycle assessment of the embodied energy, greenhouse gas and water flows associated with extracting the different HANPP (or biomass) components relevant for agro-bioenergy.

Material flow inventory (in terms of energy, greenhouse gas and water) for the extraction of harvested grain (HG) for bioenergy is relevant to the food security sustainability discussion, because its use for bioenergy competes directly with food supply chains (Scarlat et al., 2010; Haberl et al., 2012). The life cycle inventory process will consider the energy, greenhouse gas and water inputs into the bioenergy production systems, the energy outputs of main and co-products, as well as the greenhouse gas and water saved by the utilization of the co-products. Relevant LCT based indicators (namely NEG, EROEI, NCB and NBWB) for the extraction of harvested grain for bioenergy will be calculated from the results obtained from the life cycle inventory process. This is illustrated in Figs. 5 (1), 6 (1) and 7 (1) respectively.

An inventory of energy, greenhouse gas and water flows associated with the reuse of extracted residues (ER) for bioenergy is important because the reuse of already extracted residue increase biomass use efficiency and enhances transition to global bio-economy (Krausmann et al., 2008; Scarlat et al., 2010). An example of reuse of extracted residue can be found in the reuse of residues previously used for animal beddings as feedstock for bioenergy production. Similar to the life cycle inventory for the use of harvested grain for bioenergy, even though the individual activities and stages involved are different, the energy, greenhouse gas and water flows associated with the reuse of extracted residues for bioenergy can be done in order to estimate their respective NEG, EROEI, NCB and NBWB indicator values. This can be seen in Figs. 5 (2), 6 (2) and 7 (2).

Life cycle energy, greenhouse gas and water flow inventory of extracting unextracted residues (UnE) is also worthy of consideration because it represents a trade-off between ecosystem maintenance (biodiversity conservation) and harnessing additional bioenergy potentials within the context of emerging biomass demands (Haberl et al., 2005; Haberl et al., 2007). Although the life cycle stages and activities involved are different, the material flow inventory processes, which accounts for relevant energy, greenhouse gas and water flows, and leads to the calculation of NEG, EROEI, NCB and NBWB indicator values for the use of unextracted residues for bioenergy can be done in the same manner as those of harvested grain and extracted residue. This is shown in Figs. 5 (3), 6 (3) and 7 (3).

This approach will not only lead to the derivation of individual indicators that measure the effects of using any of the three relevant HANPP flows on important sustainability impact categories such as food, energy and water security, it also hand-in-hand gives room to analyze the effect of agro-bioenergy systems as a whole on the food, water and energy security nexus.

Coupling the water footprint framework with EROEI and NCB frameworks can also result in the derivation of useful indicators such as the energy return on water invested (ERoWI) which accounts for energy returned on water invested into agro-bioenergy systems for irrigation and/or remediation of toxic pollutants; and the NCB (WI) which accounts for greenhouse gas exchange as a result of irrigation (Mulder et al., 2010; Murphy and Hall, 2010). This could also be relevant at the food, water and energy security nexus (Bizikova et al., 2013; Flammini et al., 2015).

However, it should be noted that while individual and coupled life cycle thinking approaches such as EROEI, HANPP, water footprint and NCB have taken and applied some working principles from the traditional LCA (e.g. normalization of units to energy for EROEI), traditional LCA principles is yet to adopt working principles developed from these LCT approaches (Maes and Van Passel, 2014), hence our suggestion for adaptation within future LCAs.

3.1.3. Elicitation of stakeholder's social and economic impact categories in LCA

To compensate for the lack of capacity of LCAs to measure social and economic impacts (which are often times the primary concern of the decision takers stakeholder groups), some other life cycle thinking approaches and concepts like the life cycle costing (LCC), the social life cycle assessment (SLCA) and the value chain approach or analysis can be deployed (Weidema, 2006; Zamagni et al., 2013). While Life cycle costing investigates economic impact across the different value chain with the production cycle, social life cycle assessment focuses on social impacts (Jørgensen et al., 2008; UNEP, 2009). Value chain approaches usually research into both social and economic impacts, depending of the peculiarities of the case study being investigated (Lindner et al.,

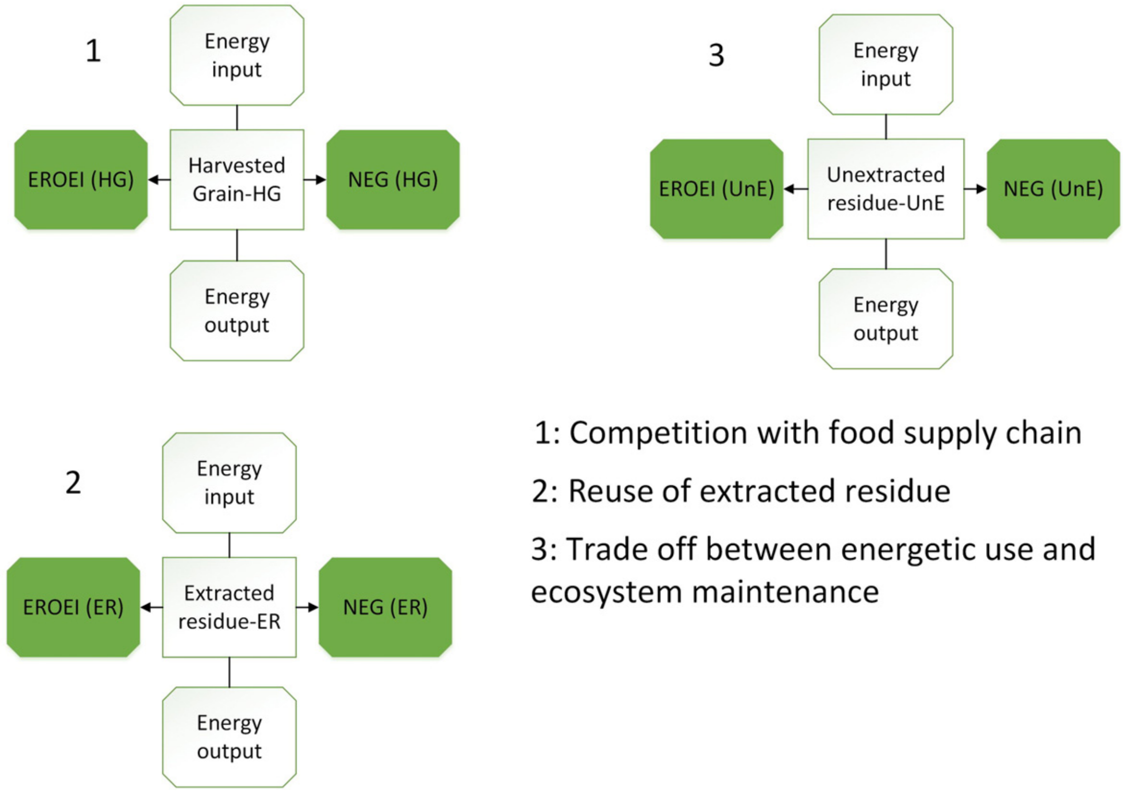

Fig. 5. Energy flow for extraction of HANPP components for bioenergy production.

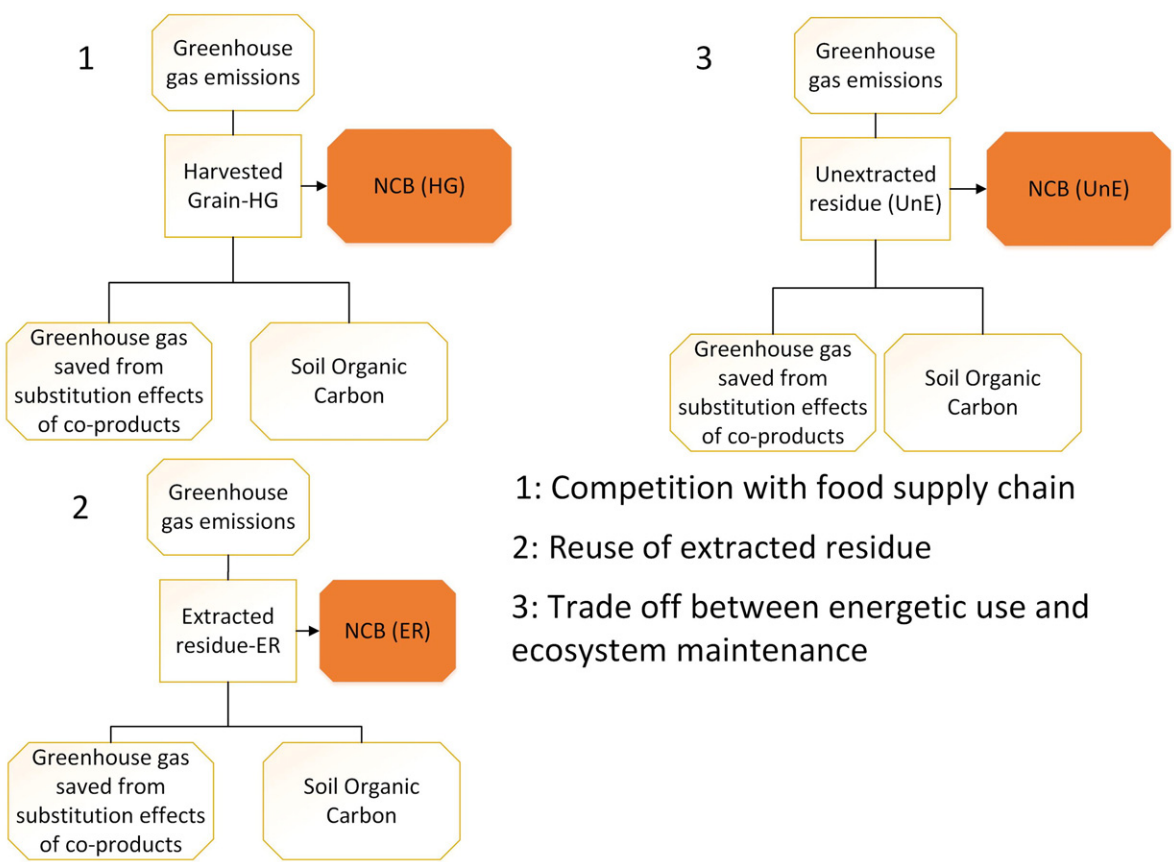

Fig. 6. Carbon flow for extraction of HANPP components for bioenergy production.

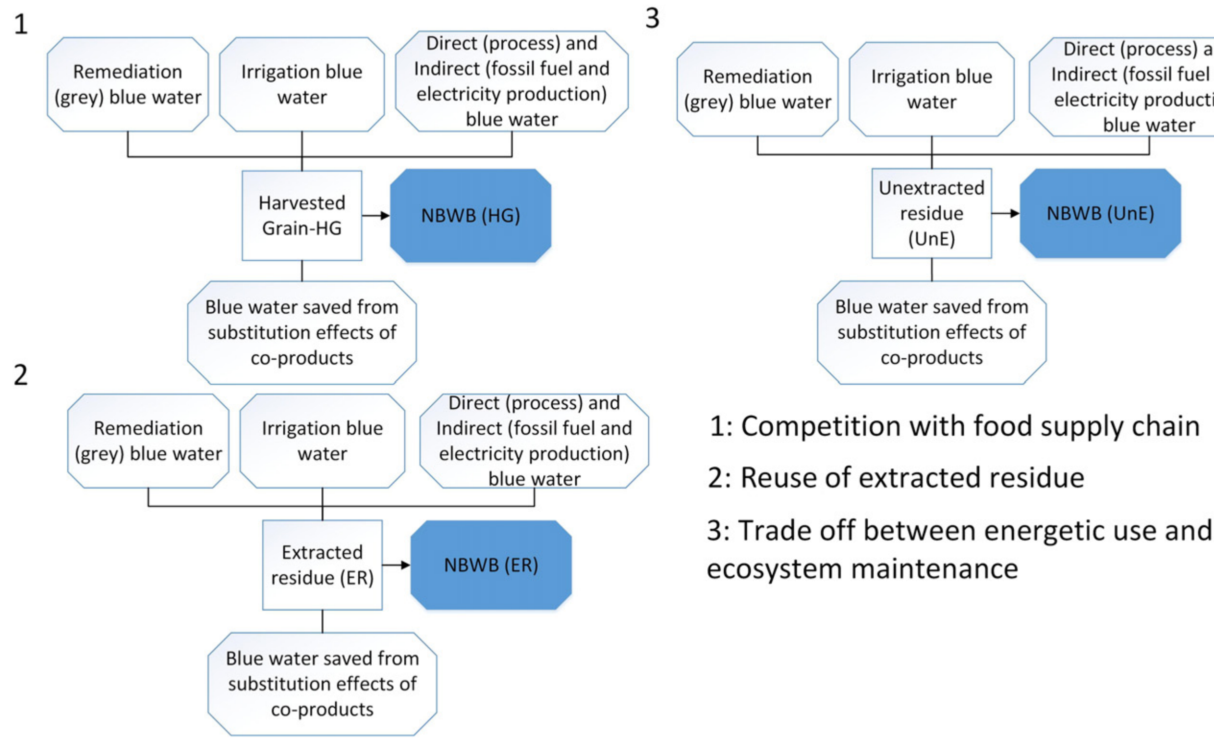

Fig. 7. Water consumption for extraction of HANPP components for bioenergy production.

2011; Soosay et al., 2012). The purpose of this group of life cycle thinking approaches is most often to maximize stakeholder's economic gains (profits), and/or to reduce adverse social impacts as defined by concerned stakeholders (Ness et al., 2007; den Herder et al., 2012).

Sustainability assessment of agro-bioenergy systems from an agro-ecological point of view entails consideration of relevant agronomic factors that reflect the characteristics of agro-ecological systems within its boundaries. This can be done by a substitution of elements of previous LCA which reflects the characteristics of agro-bioenergy systems from an industrialized agriculture point of view, with those that reflect the attributes of agro-ecological systems. An LCA that considers and/or implements changes in local or regional agronomic options within a sustainability assessment framework for agro-bioenergy systems will help quantify some of the stakeholder's concerns (mostly social and economic impacts) they touch on indirectly (Loiseau et al., 2013; Scipioni et al., 2013). While most of the previous LCAs for assessing agro-bioenergy systems align with the industrial agriculture system, assessing agro-bioenergy system from an agro-ecological point of view will not only allow for the implementation of alternative agronomic options which characterizes agro-ecological systems, especially within LCAs designed for sustainability assessment of agro-bioenergy systems; it will also by extension lead to the derivation of new LCA based indicators that can be indirect or proxy indicators for stakeholder's social and economic concerns or impacts (Jørgensen et al., 2008; Arvidsson et al., 2010). This is illustrated in Table 1.

Human and animal labour footprint can be accounted for using the maximum energy that they can exert daily multiplied by the number of days it takes them to perform individual, as well as all the tasks involved across the life cycle of the agro-bioenergy system under consideration. The amount of water taken or the amount of greenhouse gas emitted for the number of days involved during the life cycle of the agro-bioenergy system or annually could also be an acceptable means for quantifying human and animal labour footprints. The energy, greenhouse gas and water flows associated with food production for humans and animals for the number of days their labour was incurred within the life cycle of the agro-bioenergy system or throughout the year can also

Table 1
Potential LCA-derived indicators for stakeholder concerns and their proposed units.

Agronomic factors	Potential indicators	Stakeholder concern	Impact category divisions	Units
Human labour (instead of tractor labour)	Human labour footprint	Local and regional employment (Job security)	Social	Joules, g or kg or tonne of CO_2 or C equivalent, m^3 of water
Animal labour (instead of tractor labour)	Animal labour footprint	Animal rights (Sustainable livelihood-heritage protection)	Social	Joules, g or kg or tonne of CO_2 or C equivalent, m^3 of water
Tillage options (conventional VS. reduced VS. no-till)	Tillage or cultivation footprint	Farmer's cost of production (Sustainable production and consumption)	Economic	Joules, g or kg or tonne of CO_2 or C equivalent, m^3 of water
Seed sowing options (native VS. Hybrid VS. GMOs)	Seed production footprint	Environmental (indirect) and farmer's cost of production (Sustainable production and consumption)	Environmental and economic	Joules, g or kg or tonne of CO_2 or C equivalent, m^3 of water
Fertilizer options (synthetic VS. animal manure VS. biogas digestate)	Fertilizer footprint	Environmental (indirect) and farmer's cost of production (Sustainable production and consumption)	Environmental and economic	Joules, g or kg or tonne of CO_2 or C equivalent, m^3 of water
Pesticide options (synthetic VS. organic VS. biological)	Pesticide footprint	Environmental (indirect) and farmer's cost of production (Sustainable production and consumption)	Environmental and economic	Joules, g or kg or tonne of CO_2 or C equivalent, m^3 of water
Irrigation options (rain-fed VS. surface VS. sprinkler VS. drip)	Irrigation footprint	Environmental (indirect) and farmer's cost of production (Sustainable production and consumption)	Environmental and economic	Joules, g or kg or tonne of CO_2 or C equivalent, m^3 of water

be used as a benchmark for measuring their human and animal labour footprint. Cultivation, seed production, fertilizer, pesticide and irrigation footprints can be accounted for by quantifying the energy, greenhouse gas and water flow associated with their adoption across the life cycle of the agro-bioenergy system.

Also, implementing land use based frameworks like ecosystem services or land use functions in a participatory manner (side by side with LCA) can assist in factoring in both decision makers and decision takers stakeholder views on social and economic impacts into LCA (Helming et al., 2013; Loiseau et al., 2013). However it is noteworthy that, while ecosystem services framework is more suited for natural or semi-natural ecosystems (with ecological and socio-economic services provided by ecosystems as the central theme), land use functions framework is more suited for intensively used ecosystems (with focus on the services that come from the use of the land) (Wiggering et al., 2006; Schößer et al., 2010). Even though environmental, social and economic valuation of impacts is possible within both ecosystem services or land use functions frameworks, the most recent ecosystem services assessments trends towards economic analysis and quantification of stakeholder's concerns, i.e. through the Payment for Ecosystem Services concept (UNEP, 2008; Burkhard et al., 2014); while those of land use functions assessments trends toward social evaluation of stakeholder's concerns i.e. through the Framework for Participatory Impact Assessment methodology- FoPIA (König et al., 2013; Morris et al., 2011); hence offering complementary, integrated information, which enhances interpretation during decision making processes within the LCA framework (Zhang et al., 2010; de Baan et al., 2013). In providing information within sustainability assessment framework for enhancing decision making processes, LCA can either be used for assessing relevant ecosystem services and land use functions (Koellner et al., 2013; Loiseau et al., 2013), or in some other cases, ecosystem services indicators may be worked into LCA frameworks for assessing relevant impact categories (Cao et al., 2015; Rosa and Sánchez, 2015).

4. Discussion

Applying LCA for holistic sustainability assessment of agro-bioenergy systems requires compensating for the inherent weaknesses of LCA as a sustainability assessment framework, as presently being researched under a newly emerging field known as life cycle sustainability assessments (LCSA). In line with this objective and with respect to agro-bioenergy systems, this review highlighted the advantages of integrating alternative agronomic factors (i.e. alternative farm power, tillage, irrigation, fertilizer, seed sowing options) into LCAs. The integration of alternative agronomic factors into LCAs for agro-bioenergy systems do not only open up the possibility of implementing LCA from an agro-ecological point of view, it also provides a more balanced perspective of agro-bioenergy LCAs in general (Venkat, 2012; Meier et al., 2015). Aside providing a more balanced perspective for agro-bioenergy LCAs, relevant decision taker's social and economic impact categories can also be assessed by LCA indicators derivable from such assessments. This represents three out of the five sustainability assessment elements requiring methodological improvement as stated in Fig. 3. Examples of such social and economic impact sub-categories and categories assessed by integrating agronomic factors in the boundaries of previous LCAs include poverty, a sub-category under quality of life, local/regional employment, a sub-category under job security, animal/animal labour rights, a sub-category under sustainable livelihood etc. Some other decision taker's social and economic impact categories, also three out of the five sustainability assessment elements needing methodological improvement (as described in Fig. 3) can be quantified using life cycle thinking indicators. This has been exhaustively described in Section 3.1.2.

Aside assessing decision taker's social and economic impact categories, accounting for the substitution effects of returning the co-products of agro-bioenergy systems as fertilizers or pesticides under agro-ecological system is also an added value of considering LCA from an agro-ecological perspective. Since agro-ecological systems encourages the use of organic fertilizers and pesticides, EROEI NCB and NBWB assessments for agro-bioenergy systems from an agro-ecological point of view is expected to consider the substitution effects of returning co-products of agro-bioenergy systems as organic fertilizers or pesticides within agro-ecological systems. Such co-products are mostly accounted for as wastes or inefficient for optimal production within agro-bioenergy systems with industrial agriculture bias. An example of this is the substitution effects of returning biogas digestate (a co-product of biogas production) as fertilizer and lime replacement under agro-ecological systems. This do not only assist in investigating possible gains from co-products, it also reduces the energy consumption, greenhouse gas and water use burdens that their non-inclusion would have allocated solely to the environment, hence generating a more negative outlook.

While the boundaries of each LCA is expected to be drafted in relation to the sustainability questions each study is designated to answer, coupling of life cycle thinking indicators and the substitution of energies and materials (e.g. greenhouse gas/carbon flow, fresh water consumptions etc.) of previous LCA boundaries for agro-bioenergy systems, with those which reflect the effects of alternative agronomic factors, do not only lead to the derivation of new LCA based indicators for previously inadequately addressed impact categories, they also incorporate value judgments that are useful within sustainability discussions and decision making contexts. Examples of such value judgments include NCB ≤ 0, NBWB ≤ 0, NEG > 0, EROEI > 1 after conversion plant and/or ≥ 3 at the farm gate to be considered sustainable or not. This is further explained in Table 2.

While the use of coupled life cycle indicators may be instrumental to generating information needed at the food, water and energy security nexus, multiple consideration of several quantities within the same frame may create new data handling issues. This could occur while accounting for energy and greenhouse gas emissions associated with fresh water use, as well as greenhouse gas emissions and fresh water use associated with energy production and consumption. This might however not be a problem, if stakeholder data demand is restricted to one and not all of the energy/material flows involved. Lastly, the spatial and temporal deficiencies of LCAs within sustainability assessment frameworks for assessing agro-bioenergy systems i.e. the remaining two out of the five sustainability assessment elements requiring methodological improvement (as illustrated in Fig. 3) can be addressed using remote sensing/GIS based spatial information, spatially explicit models, time slices of remote sensing information (multi-temporal imageries) and spatio-temporal models.

Based on the increase in the number of indicators suggested within sustainability assessment framework for assessing agro-bioenergy systems, the workload, as well as uncertainties related to data used and estimates made should be envisaged. This can however be handled using available multi-criteria decision analysis (MCDA) tools, modelling approaches and software supports that have previously been deployed within environmental decision making contexts. Examples of multi-criteria decision analysis tools include AHP-Analytical Hierarchical Process, MAUT-Multi Attribute Utility Theory, ELECTRE-Elimination and Choice expressing Reality, PROMETHEE-Preference Ranking Organization Method for Enrichment Evaluations, DRSA-Dominance-based Rough Set Approach etc. (Lahdelma et al., 2000; Cinelli et al., 2014). Modelling approaches such as Bayesian belief networks (BBN), fuzzy logic/modelling, linear/non-linear multi-objective goal/compromise programming, ANP-analytical network process etc. are also applicable for these functions (Wolfslehner et al., 2005; Mendoza and Martins, 2006; Landuyt et al., 2013). Multi-faceted software platforms like the ToSIA (Tools for Sustainability impact assessment) framework, which can take in large body of stakeholder indicator information under a participatory process, while also incorporating many data uncertainty handling and modelling features could also be used (den Herder et al.,

Table 2
Explanation of value judgement of indicator systems within sustainability assessment framework for assessing agro-bioenergy systems.

Indicators	Explanation of value judgement of indicator systems
NEG (Net energy gain) = Energy output − Energy input (I) (Either for the use of harvested grain-HG, extracted residue-ER or unextracted residue-UnE for bioenergy)	It should be above 0 and comparable in scale to renewable energy targets or energy demand to deserve consideration as sustainable within energy security contexts (Arodudu et al., 2013, Arodudu et al., 2014, van Duren et al., 2015, Voinov et al., 2015). NEG from agro-bioenergy system is thermodynamically useless when below 0 (i.e. energy input is above energy output), energy neutral when equal to zero and an energy gaining activity when above 0 (i.e. energy output is above energy input).
EROEI (energy return on energy invested) = Energy output/Energy input (II) (Either for the use of harvested grain-HG, extracted residue-ER or unextracted residue-UnE for bioenergy)	It should be above 3 at farmgate or above 2 after refinery plant to be deserve consideration as sustainable within energy security contexts (Hall et al., 2009).
NCB (Net carbon balance) = Total GHG emissions − Total soil organic carbon (TOC) + Greenhouse gas emissions saved from substitution effects of co-products(III) (Either for the use of harvested grain-HG, extracted residue-ER or unextracted residue-UnE for bioenergy)	Final negative values indicates that the agro-bioenergy source is a net greenhouse gas (carbon) sink or storage i.e. able to reduce greenhouse gases and mitigate global warming/climate change (Hill et al., 2006, Tilman et al., 2006). A final zero value indicates that the agro-bioenergy source is carbon neutral. A final positive net carbon balance value indicates that the agro-bioenergy sources is a net greenhouse gas emitter or polluter. However, in cases comparing agro-bioenergy systems to the fossil fuel systems it is replacing, if NCB (bioenergy) < NCB (fossil energy replaced) it can be said to still have climate change mitigation potential, hence more sustainable comparatively, even if the final NCB value obtained is positive.
NBWD (Net blue water balance) = Indirect water consumed for fossil fuel use + Indirect water consumed for electricity consumed + Water consumed for irrigation (if applicable) + Water consumed during conversion process + remediation or grey water demand for dilution of pollutants − Water saved as a result of substitution effects of co − products (IV) (Either for the use of harvested grain-HG, extracted residue-ER or unextracted residue-UnE for bioenergy)	Like within the net greenhouse gas balance framework, a final positive net blue water balance value indicates that the agro-bioenergy sources is a net water consumer; a final negative net blue water balance value indicates that the agro-bioenergy sources is a net water sink (only possible if water saved from substitution effects of co-products is more than water consumed on the whole production process). A final net zero value indicates that the agro-bioenergy source is water neutral. However in cases where a process or an energy source (e.g. agro-bioenergy systems) confers comparative reduction in fresh water consumption compared to others (e.g. systems driven by fossil fuel), it may be seen as sustainable in the same way as water negative and water neutral processes or energy sources, even though a final positive NBWB value is obtained.
NPP P_{eff} (biomass production efficiency) = NPP_{act}/NPP_{pot} (X) NPP_{act}–Actual NPP NPP_{pot}–Potential NPP	Above 1 biomass production efficiency value means biomass yield is higher than potential yield; this means that the potential for meeting food security demands is high, even though it may be at the expense of significantly higher energy, carbon and nutrient inputs (Haberl et al., 2007, Kohlheb and Krausmann, 2009). Biomass production efficiency value less than but close to 1 implies that the potential for meeting food demand is high; values farther away from 1 means the potential for meeting food demand is low.
NPP U_{eff} (biomass appropriation/use efficiency) = UE/NPP_{act} OR $HANPP_{ue}/NPP_{act}$ (XI) UE or $HANPP_{ue}$–Used extraction NPP_{act}–Actual NPP	Use efficiency of 1 is not possible, a higher use efficiency (closer to 1) implies less biomass potential for meeting emerging biodiversity conservation, energy security and transition to bio-economy (substitution of depleting raw material reserves with biomass) demands (Krausmann et al., 2008; Haberl et al., 2009; lower use efficiency (farther from 1) implies higher potential for meeting biodiversity conservation, energy security and transition to bio-economy demands. In the event of high demand for bioenergy from agricultural sources.
$HANPP_e$ (land use or management efficiency) = UE/HANPP or $HANPP_{ue}$/HANPP (XII) UE or $HANPP_{ue}$–used extraction HANPP–Human appropriation of NPP	Land use or management efficiency of 1 or above 1 is rarely reached but possible if actual NPP surpasses potential NPP (usually at the expense of significantly higher energy, carbon and nutrient inputs). Land use or management efficiency of 1 (or above) may also indicate that humans are already using more than it is expected to receive from nature (usually at higher energy and environmental costs). A higher land use or management efficiency (closer to 1) implies that land space is already being utilized judiciously and therefore less biomass is potentially available for meeting emerging biodiversity conservation, energy security and transition to bio-economy demands (Krausmann et al., 2012; Fetzel et al., 2014). Lower land use and management efficiency (farther from 1) implies that the potentials of the land is yet to be fully utilized and therefore has high potentials for meeting biodiversity conservation, energy security and transition to bio-economy demands, if appropriate land use or management practices are employed.

2012). Although originally a stakeholder-based, forest value chain analysis tool, ToSIA offers a platform that can be applied for the sustainability assessment of agro-bioenergy systems (Lindner et al., 2010).

5. Conclusion

This paper reviewed indicator systems that could help assess specific environmental, economic and social aspects that have not been satisfactorily assessed within previous sustainability assessment framework for agro-bioenergy systems. It also proposed a methodology for implementing LCA of agro-bioenergy systems from an agro-ecological point of view within sustainability assessment contexts. This is through the adequate consideration of alternative agronomic factors using substitution approach within attributional LCA frameworks. The choices of impact categories and indicators selected for sustainability assessments of agro-bioenergy systems at local and regional scales is expected to be shaped by local and regional peculiarities, as well as the perception and set priorities of local and regional stakeholders concerned, hence the non-suggestion of any particularly new sustainability framework by this study. The suggestion of such a framework might amount to recommending a “one framework to fit all” agro-bioenergy system sustainability assessment problem, which is a kind of approach that has attracted lots of criticisms and have been widely discouraged among sustainability practitioners globally. Due to the varying nature of questions that may arise within sustainability assessment framework for agro-bioenergy systems, complimentary methods and tools e.g. life cycle costing, social life cycle assessments, cost benefit analysis, multi-criteria decision analysis tools, land use function assessment, ecosystem services assessment. ToSIA etc. may still be needed to ensure holistic sustainability assessments, which cover the four basic elements of sustainability assessments (namely space, time, impacts and stakeholders) adequately at the same time. Even though impact categories dealt with within this study are those related to agro-bioenergy systems, they could be modified and further applied to other cross-cutting sustainability issues as required.

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