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Life Cycle Assessment of carbon dioxide removal technologies: A critical review

† *Electronic Supplementary Information (ESI)*

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1 Afforestation and Reforestation (AR)

García-Quijano *et al.*[1] presented a case study of afforestation in the Western Cape Province in South Africa. Three management scenarios were analyzed: from natural vegetation to a multi-functional forest, from natural vegetation to a production forest (*i.e.* timber production while considering environmental services), and a non-conversion scenario (*i.e.* reference scenario). Carbon modelling demonstrated that more carbon is sequestered compared to the reference scenario. An LCA approach was used to quantify the environmental impact of Land Use Change (LUC) per tonne of sequestered C. The work of Gaboury *et al.*[2] evaluated GHG emissions for the conversion of open woodlands to black spruce forests in the region of Québec (Canada). A net carbon sequestration rate of 1.0 tC ha^{-1} $year^{-1}$ was reported, with an uncertainty range in carbon sequestration of 0.2 to 1.9 t C ha^{-1} $year^{-1}$, based on plantation yields and potential wildfires. GHG emissions reported from AR operations were 1.3 t CO_2-eq.ha^{-1}, *i.e.* a contribution of less than 0.5% of the biological C balance. Brunori *et al.*[3] examined the afforestation with an oak plantation in Central Italy after reclamation of a mining site. Their results revealed an average annual net carbon sequestration rate of 10.13 t CO_2-eq. ha^{-1} $year^{-1}$ (note the difference in unit compared to other studies). Finally, Lun *et al.*[4] examined the potential of plantation forests in North China to determine GHG emissions. The results showed that the plantation forest could serve as a carbon sink with 36 tC ha^{-1} (*i.e.* $\sim$0.9 tC ha^{-1} $year^{-1}$) for one complete rotation period of 41-years.

1.1 Carbon models

Besides LCA modelling choices, the application of different carbon models further decreases the comparability of papers on AR. For example, Gaboury *et al.*[2] used the CO2FIX model to determine carbon accumulation derived from afforestation and used an adapted module for carbon soil stocks. Lun *et al.*[4] used allometric growth models and carbon stock models to calculate carbon storage of trees. Soil and litter carbon emissions were derived from the Yasso module in the CO2FIX. Alternatively, Brunori *et al.*[3] used an older IPCC method (*i.e.* IPCC[5]) to calculate carbon stocks of their specific plantation. Not surprisingly, García-Quijano *et al.*[1] demonstrated that different carbon modelling decisions result in significant differences on the results.

1.2 Reversibility and (indirect) Land Use Change ((i)LUC)

CO_2 storage in forests can be reversible, since its permanency depends on many factors and potential disturbances.[2,6–10] In addition, direct and indirect Land Use Change (*i.e.* LUC and iLUC) can also greatly influence the GHG emission balance.[11,12] LUC refers to the conversion of land from one function to another function. iLUC can occur when land change in one place triggers unintentional changes in land on another place.[11,13,14] However, most AR studies exclude the reversibility of CO_2 storage and/or (i)LUC. We refer to

Wicke *et al.*[11] regarding (i)LUC and to Schmidt *et al.*[15] regarding the integration of iLUC in LCA.

1.3 Timing of emissions

In conventional LCA, the inventory results are aggregations of pollutants derived from various processes used in the product system over the assessed time horizon, without considering the temporal aspects of the occurrence of GHG emissions. This static modelling of emissions represents a shortcoming.[6,16] In reality, releasing a large amount of GHG emissions at once would have a different environmental impact over time than releasing the same amount of GHG emissions over a longer time horizon, since both have a different impact on the total radiative forcing.[6,7,16]

As discussed, carbon sequestration projects could be reversible due to disturbances, such as wildfires and flooding during the assessed time horizon of the project.[2,17] To demonstrate this, suppose that a wildfire appears at the end of the project and that this impact would have been included in the life cycle inventory. In this case, conventional LCA would assume zero carbon sequestration during the project, since the wild fire would emit at least the same amount (or more) of CO_2-eq. back to the atmosphere at the end-of the lifetime. In reality, there would be less radiative forcing between the start of the project and the point when the wildfire starts, since the forest grows, captures carbon hence removes CO_2 from the atmosphere, which temporary reduces radiative forcing over a certain number of years before the wildfire.[17] These temporal aspects would not have been captured with conventional LCA.

Alternatively, dynamic LCA has been proposed in Levasseur *et al.*[16]. A case study on bio-fuels revealed the critical importance of whether using dynamic LCA or not.[16] Another study on dynamic carbon methods of Levasseur *et al.*[17] demonstrated that carbon sequestration in afforestation is sensitive to disturbances during the project lifetime. Further, Albers *et al.*[7] emphasized that the exclusion of temporal aspects in biomass related CDR technologies, in terms of biogenic carbon flows, could result in an overestimation of environmental benefits.

All selected AR LCA studies excluded these temporal effect of carbon dynamics on radiative forcing (as GWP).

2 Biochar

Roberts *et al.*[18] presented a biochar (slow) pyrolysis system with three biomass feedstocks options: corn stover (crop residues), yard waste and switchgrass. Two scenarios were included to incorporate iLUC for bioenergy crops (*i.e.* switchgrass). The results revealed negative emissions from most feedstock scenarios, except the scenario where a large penalty was given for switchgrass due to the inclusion of iLUC (GHG penalty value based on Searchinger *et al.*[19]). The utilization of yard waste feedstock resulted in the largest negative emissions with -885 kg CO_2-eq. t^{-1} feedstock. Hammond *et al.*[20] conducted an LCA for different biochar systems in the United Kingdom. Biochar was derived from ten different feedstock options on three scales (small, medium and large). The authors included avoided emissions from UK electricity generation (due to use of by-products from the pyrolysis process). The results demonstrated carbon abatement rates for all scales, ranging from 0.7 to 1.3 t CO_2-eq. oven dry tonne $(odt)^{-1}$ feedstock. The best results were derived from waste feedstocks (*i.e.* forestry and sawmill residues). Peters *et al.*[21] presented an LCA of the slow pyrolysis of lignocellulosic energy crops to produce biochar and different energy products. Different biochar scenarios were considered based on the biochar stability in soils. The base case scenario assumed a biochar soil lifetime of 1000 years, and this scenario was compared with the direct combustion of biomass. The results exhibited negative emissions for the base case scenario (-1.22 t CO_2-eq. t^{-1} feedstock). However, the production of biochar, and its favorable result on GWP, came at the expense of environmental burdens on some other impact categories, *e.g.* on acidification and eutrophication potential.

3 Soil Carbon Sequestration (SCS)

A study of Adler *et al.*[22] analyzed bio-energy cropping systems on GHG emissions. Biggest net GHG-sinks were reported for Hybrid poplar and Switchgrass feedstocks (200 to 400 g CO_2-eq.-C m^{-2} yr^{-1}), respectively, due to SCS and the inclusion of the avoidance of gasoline or diesel consumption. Additionally, a study of Brandão *et al.*[6] compared environmental impacts of different energy crops. Soil quality was included as a separate impact category. The biggest negative GHG emissions were obtained from Miscanthus. Acidification and eutropication potential turned out to have the biggest impact for oilseed rape and Miscanthus, respectively. Moreover, the work of Wang *et al.*[23] examined three grazing strategies in the US for beef production: light continuous grazing, heavy continuous grazing and multi-paddock grazing (rotational grazing). The results showed that a shift of heavy continuous grazing to multi-paddock grazing (rotational grazing) results in the lowest C emissions with -2002.8 kg C ha^{-1} $year^{-1}$.

3.1 How to account for soil carbon changes in LCA?

Currently, there is no agreement in the LCA community on how to account for SCS,[24] although different approaches have been proposed (*e.g.* Petersen *et al.*[25]). This resulted in a wide variety of methods and concerns a variety of scales (*i.e.* local, regional and global), use of field data (site-specific, site-dependent, site-generic), applicability of the analysis (high to low, *e.g.* low with limited data availability) and type of models used in terms of complexity (emission factors, *e.g.* Tier 1,[26,27] carbon models, dynamic crop models and measurements).[24] Each type of model has its strengths and weaknesses.

Consequently, many approaches were used to include soil carbon changes in LCA, which makes a comparison between LCAs even more complex. Besides, some studies insufficiently clarified how (modelled) soil organic flows were linked to environmental flows in LCA. Further, each paper applied its own time horizon of assessment. Goglio *et al.*[24] provides guidelines and recommendations for best practices to include soil carbon changes in LCA.

4 Bioenergy with Carbon Capture and Storage (BECCS)

Fajardy and Mac Dowell[28] conducted a comprehensive assessment of BECCS, with dedicated energy crops and agricultural residues as biomass feedstock. Their work considered different regions, land and climate variations, and determined their water footprint, energy footprint and GWP (including iLUC). The results demonstrated case-specific differences between the scenarios, hence this revealed that the consideration of a wide set of parameters and local assessments is crucial to determine the overall environmental impact of BECCS systems. For example, three out of four case studies (based on different regions, feedstocks and mainly dependent on (i)LUC considerations) resulted in negative CO_2-emissions (ranging from -288 to -1124 t CO_2-eq. ha^{-1}). However, one case study (European willow feedstock on central grassland) demonstrated net positive CO_2-emissions, mainly due to environmental impacts generated from LUC and iLUC. Pour *et al.*[29] compared landfill gas combusted in a gas turbine (with CCS) with Municipal Solid Waste CCS (MSW-CCS). The results showed a net GWP of 0.6 and -0.7 kg CO_2-eq. per kg waste utilised for landfill gas combusted-CCS and MSW-CCS, respectively. The LCA results for other impact categories indicated environmental trade-offs coming along with the implementation of CCS. Beal *et al.*[30] examined the use of algae feedstock in a BECCS system. Algae feedstock could offer potential benefits compared to dedicated crop feedstock (conventional BECCS), since it could reduce competition of water and land consumption.[30] The results demonstrated a promising GHG performance for algae feedstocks systems, such as GHG emissions of -5,210 kg CO_2-eq. t^{-1} algae.

4.1 Carbon break-even time

During the start of a BECCS project, usually land-use change occurs, *i.e* a certain area of land is changed from an initial type of land use (*e.g.* grassland for livestock production) to a new type of land use for crop plantation. This change generates environmental impacts.[31] Besides, iLUC should be considered since there could be a reduction in agricultural production to be compensated at other locations shifting environmental impacts.[14] Therefore, LUC and iLUC have to be considered at the start of a BECCS project, which results in a 'carbon debt'.[31,32] From a temporal perspective, this carbon debt should be compensated during the BECCS project lifetime to result in negative GHG emissions.[31] More specifically, the following project years usually result in annual carbon sequestration due to growth of new trees/forests or from the plantation of crops and carbon capture and storage. The rate of carbon sequestration ultimately defines, if a BECCS project results in net positive or negative GHG emissions. The carbon break-even point is the particular moment in time (in number of years) when the carbon debt is compensated by the carbon sequestration.[31] This carbon break-even point should be shorter than the project lifetime to result in negative GHG emissions. Note that the inclusion of temporal aspects in this section excludes the timing of GHG emissions in relation to the impact on radiative forcing, as explained in Section 1.3.

5 Direct Air Carbon Capture and Storage (DACCS)

A study of de Jonge *et al.*[33] aimed to determine the life-cycle carbon efficiency of DAC systems (based on strong hydroxide sorbents), and to determine their environmental hotspots in terms of carbon removal efficiency, defined as the relative amount of CO_2 removed per unit of CO_2 captured. The results revealed a life-cycle carbon removal efficiency of 62%. Energy demand for the capture process turned out to generate most indirect emissions. Consequently, a switch to a renewable energy source (*e.g.* solar power) could increase the life-cycle carbon efficiency to 84% according to their analysis. Further, a study of Deutz and Bardow[34] assessed the DAC technology of Climework for two locations: Hinwil (Switzerland) and Hellishei∂i (Iceland). A detailed environmental analysis was presented for different adsorbent types used during CO_2 capture. The results demonstrated life-cycle carbon efficiencies of 85% and 93% for Hinwil and Hellishei∂i, respectively, with a big influence of the energy source used for CO_2 capture. In addition, a recent study of Terlouw *et al.*[35] assessed the DAC technology of Climeworks on eight locations for several system lay-outs, with a detailed analysis of the CO_2 storage stage. Different system configurations were considered for the DAC system: grid coupled (grid electricity and waste heat) and stand-alone (based on solar electricity and heat) configurations. A global and prospective analysis (year 2040) on GWP was included for grid-coupled system configurations. The results - for eight locations - demonstrated GHG removal efficiencies between $\sim$97% for the best case (Norway, with waste heat and grid-coupled), and $\sim$9% for the worst case (Greece, grid-coupled with a high-temperature heat pump). Besides, stand-alone system configurations showed a promising alternative - in countries with high annual solar irradiation - to avoid the absorption of GHG intensive grid electricity. The global sensitivity analysis showed net GHG emissions - instead of GHG removal - for grid-coupled DACCS systems in countries with GHG intensive grid electricity.

5.1 Proprietary Technology

The lack of transparency and detailed information resulted in some very generic assumptions in scientific literature on DAC, such as the utilization of alternative life cycle inventory to represent DAC systems. For example, van der Giesen *et al.*[36] approximated the life cycle inventory of DAC to be similar to the life cycle inventory of a passenger vehicle. Therefore, we argue for more transparency in the data provision of DACCS systems. For example, the work of Keith *et al.*[37] could be used as starting point to define new life cycle inventory for the high-temperature heat DAC technology of Carbon Engineering.

6 Enhanced Weathering (EW)

The goals of the one study by Lefebvre *et al.*[38] were the quantification of environmental impacts and the identification of hotspots from EW and carbonisation processes, and to estimate the carbon removal potential for a case study in Sao Paulo (Brazil). Sao Paulo State was indicated as an appropriate site for EW due to beneficial climate and soil conditions, the presence of basalt quarries and the occurrence of widely abundant basalt deposits.[38] The study uses an appropriate functional unit (per t CO_2-removed), multiple impact categories and consistent system boundaries. The authors reported 0.110 kg CO_2-eq. emissions per t of CO_2-removed, *i.e.* a GHG removal efficiency of 89%.

7 Rubrics and metrics

Table A1: Overview of classifications used for Table 2 and the generation of Figure 3 in the main article. 'None' (in case there are no LCA studies) and 'n.a.' (not available) are always assigned with a weight/value of 0.

Scale	Score	low-high	not relevant - relevant	hardly-always	less important - very important
-	0	low	not relevant	hardly	less important
o	0.5	medium		sometimes	important
+	1	high	relevant	always	very important

Table A2: Overview of rubrics/rules used for Table 2 and the generation of Figure 3 in the main article.

Indicator	*Definition*	*Scale*	*Rubric/rules*
Coverage in LCA literature	The number of LCA studies found per CDR technology.	low	Less than 3 LCA studies found.
		medium	More than or equal to 3, but less than 10, LCA studies found.
		high	More than or equal to 10 LCA studies found.
Coverage of LCIA categories	The variety and number of LCIA categories considered per CDR technology.	low	Less than 30% of the LCA studies included more than one environmental impact category.
		medium	More than or equal to 30%, but less than 70%, of the LCA studies included more than one environmental impact category.
		high	More than or equal to 70% of the LCA studies included more than one environmental impact category.
Multi-functionality	The number of functions the product system (of the CDR technology) provides.	not relevant	One primary function of the product system.
		relevant	Multi-functional output processes, which need allocation or system expansion.
Correct application of 'negative emission concept'	A proper application and use of the concept of 'negative emissions' complying with the definition of negative emissions as embedded in the CDR technology definition in the Introduction.	hardly	The vast majority of LCA studies does not properly distinguish between avoided and negative emissions.
		sometimes	Some of the LCA studies does not properly distinguish between avoided and negative emissions.
		always	The negative GHG emissions presented in the LCA papers present real negative GHG emissions in terms of CDR; a permanent removal of GHGs from the atmosphere.
Reliability of present LCA results	The quality and reliability of the results currently presented in CDR technology LCA studies.	low	Combination of the score of 'Coverage of LCIA categories' and 'Correct application of 'negative emission concept". Sum of the score (see Table A1) is less than 1.
		medium	Combination of the score of 'Coverage of LCIA categories' and 'Correct application of 'negative emission concept". Sum of the score (see Table A1) is more than or equal to 1.
		high	Combination of the score of 'Coverage of LCIA categories' at least 'medium' and 'Correct application of 'negative emission concept" = 'high'. Note: 'high' reliability is only assigned with a correct application of the 'negative emission concept', since we consider this of crucial importance. Sum of the score (see Table A1) is more than or equal to 1.5.
Importance of side-effects	Importance to include side-effects within the system boundaries of an LCA for a CDR technology.	less important	Comparatively minor side-effects are expected with the implementation of a CDR technology. Although it is still recommended to discuss side-effects and include them in the system boundaries of an LCA.
		important	Side-effects are important and should be considered in a proper LCA.
		very important	Side-effects are of crucial importance and should be considered in a proper LCA.

Table A3: Overview of CDR technologies and their evaluation results as well as explanation based on the rubrics presented in Table A2. ”none” refers to no coverage in a category. ”n.a.” means not available.

CDR technology		**Coverage in LCA literature**	**Coverage of LCIA categories**	**Multi-functionality**	**Correct application of ’negative emission concept’**	**Reliability of present LCA results**	**Importance of side-effects**
Scale		*low-high*	*low-high*	*not relevant-relevant*	*hardly-always*	*low-high*	*less important - very important*
Afforestation/Reforestation (AR)	Result	medium	low	not relevant	always	medium	important
	Explanation	4 LCA studies.	Mainly based on carbon footprint/GWP.	Usually, AR has one primary function; CDR.		’low’ (score=0) coverage of LCIA impact categories in combination with an ’always’ correct application of ’negative emission concept’ (score=1) results in a total of 1, hence ’medium’ reliability.	Negative side effects can be expected mainly due to intensive use of land needed for AR. However, the side-effects are expected to be less influential compared to other CDR technologies, since AR could improve for example the resilience of an ecosystem.
Biochar	Result	high	medium	relevant	hardly	low	very important
	Explanation	36 LCA studies.	Only 50% of the studies included another impact category than GWP.	Biochar leads to multi-functional output processes, such as biochar and energy production.		’medium’ (score=0.5) coverage of LCIA impact categories in combination with ’hardly’ correct application of ’negative emission concept’ (score=0) results in a total of 0.5, hence ’low’ reliability.	Substantial side effects can be expected mainly due to biomass utilization, such as (i)LUC, food and water competition, biodiversity and albedo change as well as ecosystem disturbances.
Soil Carbon Sequestration (SCS)	Result	high	medium	relevant	hardly	low	very important
	Explanation	35 LCA studies.	Only 30% of the LCA studies included another impact category than GWP.	SCS leads to multi-functional output processes, such as CDR and value from agriculture.		’medium’ (score=0.5) coverage of LCIA impact categories in combination with ’hardly’ a correct application of ’negative emission concept’ (score=0) results in a total of 0.5, hence ’low’ reliability.	Substantial side effects can be expected mainly due to intensive use of land and/or feedstock, resulting in implications on (i)LUC, food and water competition, biodiversity and albedo.
Enhanced Weathering (EW)	Result	low	medium	not relevant	always	high	very important
	Explanation	1 LCA study.	The LCA study included several environmental impact categories,though more studies are needed to confirm this. Note that this is exemption of the rules in the rubrics.	Primary function of EW is CDR.		’medium’ (score=0.5) coverage of LCIA impact categories in combination with ’always’ correct application of ’negative emission concept’ (score=1) results in a total of 1.5, hence ’high’ reliability. Note that this is based on one LCA study only.	The implementation of EW could potentially result in many side-effects. For example, EW modifies the chemical properties of ecosystems, such as soil, land and water and as such could influence ecosystems and agricultural productivity.
Ocean Fertilization (OF)	Result	none	none	not relevant	n.a.	n.a.	very important
	Explanation	no LCA studies.	n.a.	Primary function of OF is CDR.		lack of data.	There is a lack of long-term OF experiments. OF might include many undetermined side-effects, such as the possible reversibility of carbon storage.
Bio-energy with Carbon Capture and Storage (BECCS)	Result	high	high	relevant	sometimes	medium	very important
	Explanation	11 LCA studies.	80% of the LCA studies included at least one environmental impact category, with a focus on water depletion/scarcity.	BECCS leads to multi-functional output processes, such as CDR and the production of energy.		’high’ (score=1) coverage of LCIA impact categories in combination with ’sometimes’ a correct application of ’negative emission concept’ (score=0.5), results in a total of 1.5. However, indicated as ’medium’ reliability, since correct application of ’negative emission concept’ is not indicated as ’always’.	Substantial side effects can be expected mainly due to biomass utilization, such as (i)LUC, food and water competition, biodiversity and albedo changes.
Direct Air Carbon Capture and Storage (DACCS)	Result	medium	medium	not relevant	always	high	less important
	Explanation	3 LCA studies.	Two LCA studies are comprehensive in the number of environmental impact categories used. However, the results are currently based on low temperature DAC as well as a small number of LCA studies and conclusions need to be confirmed in future LCA studies.	Primary function of DACCS is CDR.		’medium’ (score=0.5) coverage of LCIA impact categories in combination with ’always’ correct application of ’negative emission concept’ (score=1) results in a total of 1.5, hence ’high’ reliability.	DACCS systems (potentially) cause less harmful environmental side-effects compared to other CDR technologies, since DACCS systems generally avoid (i)LUC, food competition and ecosystem implications.

8 Review tables

Table A4: Summary of LCA studies on afforestation and reforestation. The following acronyms mean: n.a. not applicable or specified. GWP global warming potential, C carbon, LUC land use change, ⋆ C flows are not considered as an actual LCIA method (in AR studies usually presented as carbon flows from carbon models).

Ref.	**Year**	**Country**	**Goal**	**Functional unit**	**Alloc.**	**LCIA method**	**Impact Cat.**			**GWP/C result**
							GWP	**C ⋆**	**Others**	
[1]	2007	S. Africa	Determine carbon sequestration and environmental impact of AR project	tC sequestered, tC emission reduction	n.a.	Peters *et al.* [39]		x	LUC	land use result: -1.02 to -1.21 ha year tC^{-1} (average of 30 years in Table 8)
[2]	2009	Canada	Estimate C balance and LCA on afforestation	ha	n.a.	IPCC 2001	x	x		-1.0 t C ha^{-1} $year^{-1}$. Total GHG-emissions of 1.3 t CO_2-eq.ha^{-1},less than 0.5% of biological C balance
[3]	2017	Italy	Determine carbon balance and GWP of anthropogenic activities of an oak plantation	ha of oak plantation	n.a.	IPCC 2013	x	x		-10.13 t CO_2-eq. ha^{-1} $year^{-1}$
[4]	2018	China	Determine carbon dynamics and emissions over rotation period	ha	n.a.	IPCC 2006		x		-0.9 tC ha^{-1} $year^{-1}$

Table A5: Overview of reviewed biochar LCA studies. s.e. system expansion or substitution, n.a. not applicable or (clearly) specified. GWP global warming potential, OP ozone depletion, HT human toxicity, AP acidification, EP eutrophication, ET ecotoxicity, LU land use, WD water depletion, EC energy consumption, ME marine eutrophication, FD fossil depletion, PMF particulate matter formation, POF photochemical oxidant formation, POCP photochemical ozone creation potential, SFP smog formation potential, IR ionizing radiation, AD abiotic depletion, RD resource depletion, FET freshwater ecotoxicity, MET marine ecotoxicity.

Ref.	Year	Country	Goal	Functional unit	Alloc.	LCIA method	Impact Cat.								GWP result
							GWP	OP	AP	EP	LU	WD	EC	Others	
18	2010	Global, US	Determine economic and environmental potential for different feedstocks	t of dry biomass managed	s.e.	IPCC 2007	x						x		Between +36 and -885 kg CO_2-eq. t^{-1} biomass feedstock, depending on type.
20	2011	UK	Quantifification of carbon abatement from pyrolysis biochar systems	odt feedstock, odt biochar produced, MWh electricity produced, ha land for feedstock, per pyrolysis facility	econ., s.e.	IPCC 2007	x								carbon abatement of 0.7 to 1.3 t CO_2-eq. odt^{-1} feedstock processed
40	2012	UK	Analyze carbon abatement of biochar treatment configurations of 10 feedstocks	t of feedstock	s.e.	Different sources	x								carbon abatement of 0.07 to 1.25 t CO_2-eq. t^{-1} feedstock
41	2012	Finland	Comparison of biochar and straw-bale construction for carbon storage	t of straw	s.e.	IPCC 2007	x								-0.9 t CO_2-eq. t^{-1} of straw for biochar
42	2013	Zambia	Compare conventional and conservation farming, with soil amendment of biochar	produced t of maize per year	n.a.	Recipe H Endpoint, Midpoint in ESI	x	x	x	x	x	x		many others in ESI	Only endpoint categories in ecopoints presented, wide variety of GWP results in ESI.
43	2013	Australia	Microalgae as feedstock for processing to electricity, bio-oil and biochar	MJ of pyrolysis biogas combusted for electricity, MJ of pyrolysis bio-oil combusted for electricity or extracted lipid refined for transport fuel, MJ of pyrolysis biochar combusted for electricity, t of pyrolysis biochar applied to soil as conditioner	Econ.	CML 2, Recipe H midpoint, Usetox, BPIC endpoint, WMO, IMPACT 2002+	x	x	x	x	x	x		ET, HT, AD	Wide variety result reported, ranging from -0.03 to 1.95 t CO_2-eq. t^{-1}
44	2013	Global, US	Comparison feedlot manure management practices: gasification vs. land application	t of dry feedlot manure	s.e.	IPCC 2007	x								-643 kg CO_2-eq. t^{-1} dry feedlot manure
45	2013	US	Propose a life-cycle and cost model of co-production of biochar by fast, slow pyrolysis and gasification	management of 1 dry Mg of biomass residue	s.e.	IPCC 2007;CO, non-methane hydrocarbons and PM from[46]	x								up to GHG mitigation of 1.4 Mg CO_2-eq. Mg^{-1} feedstock
47	2013	Poland	Evaluation of two emerging sludge-to-energy systems making use of fast pyrolysis and anaerobic digestion	daily treatment of 500 m3 liquid raw sewage sludge (5% solids content)	s.e.	IPCC 2006	x						x		-11.8 t CO_2-eq./daily treatment of 500 m3 liquid raw sewage sludge (SP system)
48	2014	Indonesia	Compare two types of biochar utilization: soil amendment and cooking	annual impact for average village household utilizing available cocoa waste	n.a.	Recipe H Endpoint, Midpoint in ESI	x	x	x	x	x	x		many others in ESI	-720 kg CO_2-eq/hh (presented in ESI as CC midpoint)
49	2014	Ghana	Combining composting with biogas and biochar production	one year of project operation at scale, unit of waste input	s.e.	n.a.	x								-230 tCO_2-eq. $year^{-1}$
50	2014	Canada	Determine potential of biochar production from different agricultural residues and wastes	t biomass dry weight	s.e.	IPCC 1996	x								Around -40 kg CO_2-eq.t^{-1} of forest residue and corn fodder (from figure)
51	2015	Canada	Impact of biochar-based bioenergy production and biochar land application	t of biochar produced from woody biomass	Mass	Eco indicator 99	x	x	x	x			x	many others	0.85 kg CO_2-eq. t^{-1} biochar produced
21	2015	Spain	Evaluation of slow pyrolysis system for production of energy and biochar	ha agricultural area, used for one yr	s.e.	CML	x		x	x			x	AD	-17.7 t CO_2-eq. ha^{-1} dry biomass feedstock
52	2015	USA	Determine perspective of conventional waste water treatment when making biochar and biosolids as co-product	dmt biosolids generated at a WWTP	pyrolysis allocated to biochar, s.e.	IPCC 2007	x						x		No value given, significant GHG-reductions achieved
53	2015	China	Compare use of agricultural residues to produce biochar or bioenergy	odt of straw	s.e.	IPCC 2007	x								carbon abatement of 1.06 Mg CO_2-eq. odt^{-1} straw
54	2016	Vietnam	Determine carbon footprint of two production pathways of biochar for the application on paddy fields	management of 1 Mg of dry rice straw	s.e.	IPCC 2013	x								GHG abatement of 0.27 and 0.30 Mg CO_2-eq Mg^{-1} rice straw, during spring and summer, respectively
55	2016	Vietnam	Comparison rice cropping systems using different residue management	production of 1 kg of milled rice	s.e.	IPCC 2013	x								1.11 and 3.85 kg CO_2-eq kg^{-1} rice, during spring and summer, respectively.
56	2016	Italy	Assessment of local slow pyrolysis plant	t of dry matter of feedstock	s.e.	IPCC 2013	x								-737 kg CO_2-eq. t^{-1} dried feedstock
57	2017	Vietnam	Compare three different pyrolysis technologies using rice husk to biochar	management of1 tonne of dry rice husk	s.e.	IPCC 2013, Recipe midpoint	x							PMF, HT	-0.74 to - 1.09 t CO_2-eq. t^{-1} dry rice husk, depending on system and season
58	2017	Zambia, Indonesia, Nepal	Compare 5 different pyrolysis methods to produce biochar	preparation and sequestration of 1 kg biochar	s.e.	Recipe H Endpoint	x	x	x	x	x			RD, IR, POF, PMF, HT, MET	No GWP values, only endpoint presented
59	2017	Spain	Determine environmental potential of biochar production from tomato biomass	production of 1 t of biochar with tomato plants as feedstock	Mass	IPCC 2013	x								in a best case scenario with 80% stable-C: -156 kg CO_2-eq. t^{-1} biochar
60	2017	Chile	Determine environmental hotspots in the life-cycle of biochar to soil	t of produced biochar	s.e.	Recipe 2016 (H) Midpoint	x							FD, HT, FET	between -2590 to - 2736 kg CO_2-eq. t^{-1} for different scenarios
61	2018	Spain, Italy, Belgium	Assess environmental impacts of recyling materials in agriculture by pyrolysis, composting and its combination	kg product, ha yr^{-1}	s.e.	CML 2002	x		x	x					No values presented, negative carbon emissions.
62	2018	Canada	Compare two pyrolysis systems: Scenario A, with a lower pyrolysis temperature and shorter solid residence compared to scenario B.	production of 1 Mg of biochar	s.e.	IPCC 2007	x						x		GHGs for A and B are -2110 and -2561 kg CO_2-eq. Mg^{-1} biochar , respectively
63	2018	Indonesia	Influence of spatial differentiation for the implementation of biochar as waste management strategy	kg of biogenic carbon from biomass residues	s.e.	Recipe endpoint, USEtox 2.02, IPCC 2013	x	x	x	x	x	x		PMF, IR, ET POF, MET, HT, RD, FET	No GWP results presented, only endpoint categories
64	2018	Indonesia, Australia	Use of oil palm waste to produce biochar and applied on three broad-acre crops	t of biochar	s.e., mass alloc. for palm oil products	UNFCC 2014	x								Average of -489 kg CO_2-eq. t^{-1} biochar (land-use change excluded)
65	2019	Sweden	Prospective LCA of large-scale biochar production in Stockholm	t dry woodchips acquired on the on global market	s.e.	IPCC 2013	x								between -696 to -1193 kg CO_2-eq. t^{-1} of woodchips, depending on biochar yields
66	2019	Belgium	Determine environmental impacts of biochar using willow and pig manure as feedstock	t of biochar	s.e.	IMPACT 2002+, CML	x	x	x	x	x	x	x	Many others	-2,063 and -472 kg CO_2-eq. t^{-1} of willow and pig manure, respectively
67	2019	Netherlands	Application of bioenergy and biochar to replace peat and natural gas	delivery of 1 MJ heat	s.e.	IPCC 2014, IMPACT 2002+	x	x	x	x	x	x		HT, RD	-0.011 kg CO_2-eq. MJ^{-1} for willow wood including biochar use.
68	2019	Denmark	Assess three scenarios for use of Winter oilseed rape including biochar production from pyrolysis	Mg dry seed of Winter oilseed rape	s.e.	IPCC 2013	x								171 (400 °C) and 111 (800 °C) kg CO_2-eq. t^{-1} dry seed of Winter oilseed rape
69	2019	Sweden	Determine land-use efficiency of maize digestion and willow-based pyrolysis systems	ha of land yr^{-1}, energy content of the biomethane in the delivered product	Physical, s.e.	IPCC 2013	x						x		Willow scenario a and b resulted in '-0.004 and -0.026 Mg CO_2-eq. GJ^{-1} biomethane, respectively.
70	2019	Sweden	Three thermochemical biosludge processing technologies compared to produce biochar; incineration (A), pyrolysis (B) and HTC (C)	t of dry matter biosludge	s.e.	IPCC 2013, CML-IA, Impact 2002+	x		x	x				ET	-0.89 (A), -1.43 (B), and -1.13 (C) t CO_2-eq. per tonne of biosludge
71	2019	Finland	Possibility of using biochar produced from side flows and buffer zones, to neutralize GWP impacts derived from crop production	t of oat flakes	s.e.	IPCC 2013	x								GHG mitigation of side flows and buffer flows of 350 and 390 kg CO_2-eq. t^{-1} of oat flakes, respectively
72	2019	Canada	Determine environmental impacts of sewage sludge considering heavy metals for pyrolysis	amount of sludge required to produce 1 t of biochar	n.a.	CML Baseline 2015	x							FET	Around -200 kg CO_2-eq. t^{-1} biochar (from graph)
73	2019	China	Determine carbon footprint of 7 sites for the application of biochar soil amendment	ha cultivation area, kg grain produced in cropland	n.a.	IPCC 2013	x								Wide variety of GHG emissions ranging between crops and scenarios

Table A6: Overview of reviewed SCS LCA studies. s.e. system expansion or substitution, n.a. not applicable or (clearly) specified. GWP global warming potential, OP ozone depletion, HT human toxicity, AP acidification, EP eutrophication, ET ecotoxicity, LU land use, WD water depletion, EC energy consumption, ME marine eutrophication, FD fossil depletion, PMF particulate matter formation, POF photochemical oxidant formation, POCP photochemical ozone creation potential, SFP smog formation potential, IR ionizing radiation, AD abiotic depletion, RD resource depletion, FET freshwater ecotoxicity, MET marine ecotoxicity.

Ref.	Year	Country	Goal	Functional unit	Alloc.	LCIA method	Impact Cat.								GWP result
							GWP	OP	AP	EP	LU	WD	EC	Others	
22	2007	US	Analyze bioenergy crops system on net GHG emissions	m^2 per year	s.e.	IPCC 2001	x								Net GHG sinks of 200 and 400 g CO_2-eq. C m^{-2} yr^{-1} for Hybrid poplar and Switchgrass feedstocks, for conversion to ethanol and gasification for electricity, respectively.
74	2009	US	Estimate environmental performance of corn cultivation under current tillage practices	kg of dry biomass	s.e., mass	IPCC 2001, CML, [75]	x		x	x			x		Between 254 and 824 kg CO_2-eq. kg_{-1} for corn grain and -39 and 89 kg CO_2-eq. kg_{-1} for corn stover
76	2010	Denmark	Determine environmental impacts of organic pig systems	kg of live weight pig delivered from the farm	n.a.	EDIP 97	x	x	x	x	x				Between 2830 to 3320 g CO_2-eq. kg^{-1} of live weight pig delivered
77	2010	Brazil	Evaluate impact of biofuel production	ha per year	s.e.	IPCC 2007	x								748 kg C-eq. ha^{-1} yr^{-1} , for unburned systems.
78	2011	UK	Comparison of land-use systems for energy production of crops	ha per yr, GJ energy	s.e.	CML 2001	x		x	x			x	Soil quality	Min. and max. of -402 to -11,096 kg CO_2-eq. ha^{-1} $year^{-1}$ for oilseed rape and miscanthus, respectively.
79	2012	US	Assess six production scenarios of activate cropland grown used to grown Miscanthus feedstock	MJ ethanol	s.e.	[80]	x								*e.g.* ranging between -26 to 22 g CO_2-eq. per MJ ethanol for scenario 4.
81	2012	US	Determine GHG emissions of 12 crop products based on organic and conventional methods	kg of product	n.a.	IPCC 2006, PAS 2050:2008	x								Wide variety of results, result mainly depended on type of feedstock and practice used
82	2013	US	Determine environmental impacts of bioenergy production on marginal lands	m^2 per year	s.e.	IPCC 2007	x								Between -241 to -932 g CO_2-eq. m_{-2} yr_{-1}, depending on cropping system
83	2013	US	Determine GHG emissions of wheat-based cropping systems	ha per year	n.a.	IPCC 2007	x								Values not presented but derived from figure between 0 and 0.7 t CO_2-eq. ha^{-1} $year^{-1}$
84	2014	UK, Ireland, US	Determine carbon footprints of grass-based dairy farms	t energy corrected milk	s.e., milk, mass, econ., protein, biologic., emission	IPCC 2007	x								Wide variety of results. *e.g.* based on milk alloc. the results ranged from 837, 884, 898 kg CO_2-eq. in Ireland (grass-based), UK and US (confinement), respectively.
85	2015	Spain	Determine GHG emissions of 6 crop types of organic and conventional systems	kg of product	Product, econ., mass, s.e.	IPCC 2006	x								Big variation of GWP results presented, mainly depending on practice, crop and allocation method
86	2015	France, Germany, Poland	Assess rapeseed production considering different management and fertilization practices	t of dry rapeseed	n.a.	CML 2000	x	x	x	x				AD, POF	Depends on scenario, from 651 to 896 kg CO_2-eq. t^{-1} of rapeseed.
87	2015	Spain	Determine carbon footprint of milk sheep farming systems of grasslands	L of FPCM	econ.	IPCC 2007	x								Ranging from 2.43 (IPCC, for intens. breeding) to -3.41 (IPCC, for extens. Breeding), kg CO_2-eq.
88	2015	Australia	Environmental assessment of farm management practices in wheat cropping systems	ha per year, t of feedstock	n.a.	IPCC 2007	x								Average of 475 kg CO_2-eq ha^{-1} for wheat production.
23	2015	US	Compare three grazing strategies on GHG emissions	marketed beef calf, hectare of rangeland	mass	IPCC 2007	x								between -2002.8 and -89.5 kg C ha^{-1} $year^{-1}$ (15 year scenario)
89	2016	Brazil	Determine economic and environmental performance of beef farms at farm level	kg of live weight	mass	IPCC 2007	x								Scenarios ranging from 17.8 to -53.6 CO_2-eq. kg^{-1} of live weight
90	2016	Sweden	Determine impact of barley production systems	t of harvested barley (DM)	n.a.	IPCC 2007, CML 2002	x		x	x			x		Between 325 to 372 kg CO_2-eq. t^{-1} of harvested barley (DM)
91	2016	Italy	Use of marginal soils for cultivation and evaluate environmental impacts	ha of giant reed, t of dry giant reed biomass	n.a.	CML 2015	x	x	x	x			x	POCP	between -3121 and -3943 kg CO_2-eq. ha^{-1} for MS and FS, respectively. And -172 and -107 kg CO_2-eq. t $d.m.^{-1}$ for MS and FS, respectively.
92	2016	US	Compare grazing-systems for beef production	ha per year, steer	n.a.	IPCC 2006	x								-2.05 to -2.11 for MOB and -0.45 to -1.07 for IRG in Mg C ha^{-1} yr^{-1} ,when a SCS rate of 3 Mg C ha^{-1} yr^{-1} was applied.
93	2017	Denmark	Evaluation of the environmental impacts of the prodiction of bioenergy feedstocks	t dry matter of harvested biomass	econ.	EPD, Recipe, ILCD	x			x	x		x	Soil quality, FET	84, 100 and 264 kg CO_2-eq. t DM^{-1} for Alfalfa, Willow and barley straw, respectively.
94	2017	US	Assess the effect of different feeding strategies in organic farms on GHG emissions	metric t of energy corrected milk	mass, milk sales	IPCC 2006	x								1,457 kg CO_2-eq. t^{-1} of ECM.
95	2017	Italy	Determine GHG emissions of small mountain (grassland) dairy farms	kg of FPCM, m^2 of agricultural Land	no alloc., physical	IPCC 2007	x								With no alloc.: 1.38 and 1.10 kg^{-1} FPCM for LLU and HLU , respectively. With phys. alloc.: 0.60 and 0.79 kg^{-1} FPCM for LLU and HLU, respectively.
96	2017	US	Compare GWP of seaweed and terrestrial bioethanol	kL of bioethanol	n.a.	Recipe	x								Best GHG-emission performance for switchgrass with 866 kg CO_2-eq. per kL^{-1} bioethanol.
97	2018	Japan	Compare carbon emissions of nature (NF), environmentally friendly (EF) and conventional farming (CF) systems	ha, kg of product	n.a.	[98]	x								Net carbon emissions of 1.98 and 1.81 CO_2-eq./ha for CF and EF, respectively. Carbon sinks could be achieved with no-till farming up to -1.87 CO_2-eq./ha.
99	2018	US	Determine GHG emissions of two beef finishing systems	kg of carcass weight	n.a.	IPCC 2014	x								6.12 and -6.65 kg CO_2-eq. per kg carcass weight for feedlot and adaptive multi-paddock grazing, respectively.
100	2018	Brazil	Environmental assessment of different sugarcane production strategies	t of sugarcane at the distillery	n.a.	Recipe H midpoint, endpoint	x	x	x	x				POF, PMF, FD, HT	Value not mentioned, around 30 kg CO_2-eq. t^{-1} of sugarcane for (SOC-20 years) derived from figure.
101	2018	Italy	Compare environmental profile and identify hotspots for fertilizer management	kg of dry biomass produced, ha cropped land	n.a.	Recipe H Midpoint	x	x	x	x		x		POF, PMF, FD, ME	-0.21 and -0.09 kg CO_2-eq. for Low input N and High input N kg^{-1} of dry biomass, respectively
102	2018	China	Indicate net GHG-balance of Chinese cropping systems including SCS and upstream CO_2-emissions	ha per year	n.a.	different factors and sources	x								between 328 to 7,567 kg C-eq. ha^{-1} year depending on cropping system
103	2018	Spain	Determine carbon footprint of agroforestry systems as extensive rangeland farms	kg live weight of product, litre of milk	mass	IPCC 2007	x								No specific value mentioned, we derived from figure and calculated: 6.34 kg CO_2-eq. kg^{-1} live weight for feedlot-finished beef cattle
104	2019	Denmark, UK, Austria	Compare organic and conventional dairy systems	kg of FPCM	ratio FPCM	ILCD Midpoint USEtox 2.02	x		x	x	x			Biodiversity, ME, RD, FET	From 0.74 to 1.01 kg CO_2-eq. kg_{-1} FPCM for organic grassland systems and conventional mixed systems, respectively
105	2019	England, Wales	Determine GHG emissions of organic food production	t of marketed crop product	Econ., s.e.	IPCC 2006	x								No specific numbers mentioned for the results including SCS.
106	2019	Bangladesh	Determine GHG emissions of rice production when changing land management	t of monsoon rice grain	n.a.	IPCC 2007	x								between 1.04 to 1.75 t CO_2-eq. t^{-1}, depending on cropping practice scenario.
107	2020	Italy	Determine carbon footprint of Alpine dairy production systems	kg of FPCM	n.a.	Recipe H mid- and endpoint, [108], IPCC2006, [109]	x							Biodiversity	Varietion of GWP results, between 0.98 to 1.33 kg CO_2-eq. kg^{-1} FPCM
110	2020	US	Determine environmental potential of high yield perennial energy crops on marginal lands	MJ of electricity, MJ of heat	exergy	TRACI 2.1	x	x	x	x			x	SFP, resp. effects, HT, ET	Variety of GWP results. *e.g.* 6.9 g CO_2-eq. MJ^{-1} electricity for Napier grass bale (sc2)
111	2020	Spain	Evaluate organic livestock farming of seven (extensive) farming systems	kg of live weight or kg of FPCM, ha	mass	IPCC 2014, Recipe H Midpoint	x								Variety of GWP results. Lowest CO_2-emissions from montanera pig and semi-extensive dairy goat farms.

Table A7: Results on BECCS. s.e. system expansion or substitution, n.a. not applicable or (clearly) specified. GWP global warming potential, OP ozone depletion, HT human toxicity, AP acidification, EP eutrophication, ET ecotoxicity, LU land use, WD water depletion, EC energy consumption, ME marine eutrophication, FD fossil depletion, PMF particulate matter formation, POF photochemical oxidant formation, POCP photochemical ozone creation potential, SFP smog formation potential, IR ionizing radiation, AD abiotic depletion, RD resource depletion, FET freshwater ecotoxicity, MET marine ecotoxicity.

Ref.	Year	Country	Goal	Functional unit	Alloc.	LCIA method	Impact Cat.								GWP result
							GWP	OP	AP	EP	LU	WD	EC	Others	
28	2017	Brazil, China, Netherlands, India, US	Consider BECCS implications of biomass feedstock derived from different climates, regions and land types	Varies, *e.g.* ha land (for GWP)	Spec. assump-tions	n.a., different sources	x					x	x		for case studies (mean) between 31 to -1124 t CO_2-eq. ha^{-1}
112	2017	Australia, Global	A sustainability framework for BECCS technologies	kg of wet MSW	s.e.	ALCAS, IPCC 2013	x	x		x		x		AD, HT, PMF, FET	-0.7 and 0.59 kg CO_2-eq. kg^{-1} of wet MSW for MSW-CCS and LFG-CCS (global), respectively.
113	2018	China	Determine GHG emissions from co-firing of biomass in coal fired power plants	kWh of generated electricity delivered to the grid	energy (agr. process)	IPCC 2013 values	x								-651 kg CO_2-eq. MWh^{-1} generated electricity for the BECCS scenario (CBECCS-CrB4)
114	2018	UK	Evaluation of performance of pulverized coal-biomass fueled power plant	MWh electricity	n.a.	n.a. only CO_2	x						x		only mentioned that BECCS result in net negative emissions, no numbers given in text
29	2018	Global	Determine feasibility for using MSW as biomass resource	kg of wet waste delivered to landfill	s.e.	ALCAS IPCC 2013, Recipe Midpoint H	x	x	x	x		x		AD, HT, PMF, FET	Between 0.6 and -0.7 kg CO_2-eq. for LFG-CCS and MSW-CCS, respectively.
115	2018	Australia	Economic and environmental potential and impacts of BECCS in the Australian power sector	kWh electri-city generated	n.a.	ALCAS, IPCC 2013	x	x	x	x		x		AD, HT, PMF, FET	Between -0.66 to -1.81 kg CO_2-eq. kWh^{-1} electricity for LFG-CCS and BG-CCS, respectively
30	2018	USA	Integration of algae as feedstock in BECCS systems	t of algae produced	s.e.	IPCC 2013, Recipe Midpoint H	x					x			-5,210 kg CO_2-eq. t^{-1} of algae
116	2019	n.a.	Compare two transition pathways for power plants: a fuel change from coal to gas, and the implementation of BECCS	MWh electri-city produced	n.a.	n.a.	x								External biomass configurations remove between '0.5 to 1 t CO_2-eq. MWh^{-1} produced
117	2019	China	Environmental assessment of coal power plants co-fired with biomass with CCS	one MWh of net power produced	n.a.	CML 2001	x	x	x	x				AD, HT, MET, FET, POCP, ME	-877 kg CO_2-eq. MWh^{-1}, for BECCS assuming carbon neutral biocmass cycle
118	2020	n.a.	Feasibility of Hydrothermal Treatment (HTT)-CCS , compared to BECCS	1 Mt biomass on DW	n.a.	GREET, ecoinvent	x						x		Derived from figure, around -1.5 t CO_2-eq. Mt^{-1} biomass on DW, depending on feedstock and plant
119	2020	Central Europe	Techno-environmental assessment of hydrogen production from natural gas and/or biomethane	'Production of 1 MJ of compressed ga-seous hydrogen... '	Allocated to other sectors, Econ. (in background)	ILCD 2.0, Recipe, CED	x	x	x		x		x	many others	Biomethane as feedstock resulted in -125 g CO_2-eq. MJ^{-1} H_2 for scenario 'ATR BM, HT +LT; VPSA 98'

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