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The environmental impact of valorising agricultural by-products

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1 Introduction

Agricultural by-products are an important source of biomass containing nutrients such as nitrogen (N), phosphorus (P) and potassium (K) and carbon (C) and other compounds such as cellulose, hemicellulose and fibres. Agricultural by-products include a varied range of residual biomass such as crop residues (primary residue), residues from processing industry (secondary residue) and manure. Where crop residues and manure are often cycled back to the soil and crop to maintain soil quality and crop growth, residues from processing industries are often used as animal feed or to a lesser extent for human food. By-products can also be used to produce bio-based materials and products. This use, generally called valorisation, is becoming more relevant in light of a circular and bio-based economy that should reduce the need for depletable fossil resources and their related environmental impact (Tuck et al. 2012). Hence, the demand for biomass is deemed to increase up to approximately 24 billion tonnes (bn t) dry matter (DM) in 2050 worldwide (Piotrowski et al. 2015). Agricultural by-products, mostly crop residue, make up approximately 12% of that at 2.8 bn t DM (Lal 2005) and are available in limited quantities as their production is dependent on the production of main products such as grains and oils.

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By-products can be valorised and converted through different treatment pathways such as fermentation, pyrolysis and anaerobic digestion (Fig. 1). These treatment pathways deliver different bio-based materials and products for a range of markets varying in size and economic value. Markets include fuels, chemicals for pharmaceutical and bulk chemical applications, animal feed and materials. Another 'market' is the often original use of by-products for building soil organic matter (SOM) and fertilising crops when returned to the soil either with or without addition of manure (Fig. 1). Although various treatment pathways are possible, conversion to bioenergy such as bioethanol, electricity and heat have been the most considered treatment options in the past decade. Only a few studies have focused on a combination of multiple treatment pathways such as bio-refining and bio-cascading (Fig. 2). Bio-refinery as an integrated treatment pathway is comprised of a combination of technologies and is becoming more relevant in light of providing multiple bio-based materials from the same source of biomass. Bio-cascading is the more general term for valorising different components of biomass.

As valorisation can take place through different treatment pathways and by-products are available in limited quantities, competition for by-products between treatment pathways will easily occur and will affect the environmental consequences. Changing the use of by-products from its original application, such as animal litter or feedstock to another valorisation route will induce the need of a substitute for its original use. For example, when co-digesting beet pulp with animal manure for energy production, the beet pulp cannot be utilised for animal feed (De Vries et al. 2012a). Henceforth, a substitute for the animal feed will need to be introduced, for example barley. Consequently,

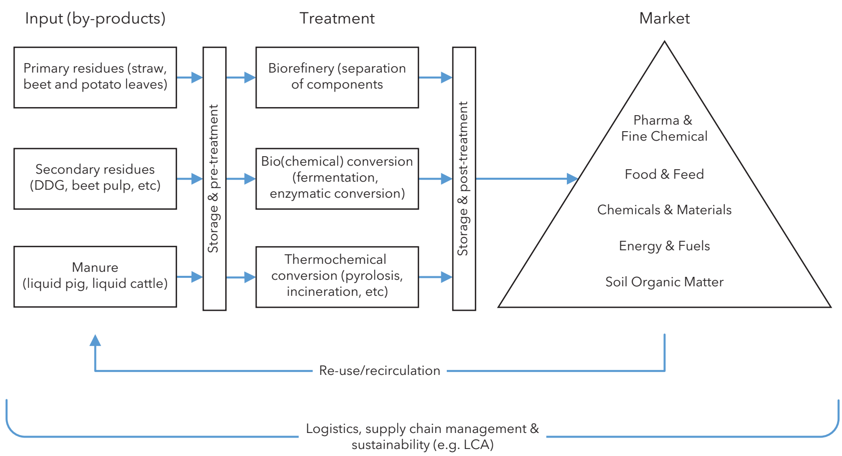

Figure 1 Valorisation of by-products through different treatment pathways, markets, re-use and recirculation, logistics, supply chain management and sustainability.

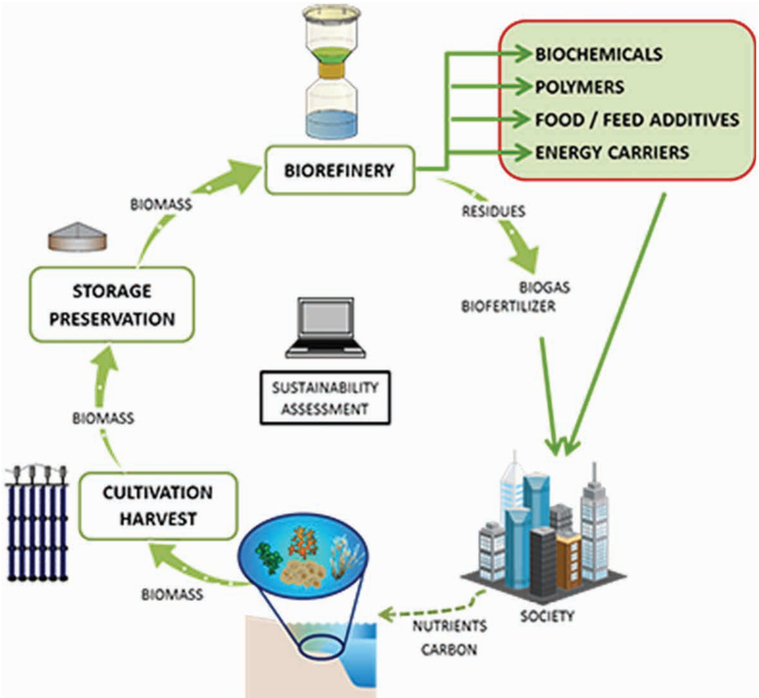

Figure 2 Concept of the bio-refinery in a circular society perspective. Courtesy of Gröndahl (2013).

the environmental impact of producing the barley needs to be included. Consequences related to the production of barley include land use and (indirect) land use changes (LULUC). LULUC can significantly affect the soil carbon (C) balance and sequestration and therewith the greenhouse gas (GHG) balance (Wiloso et al. 2016). Such changes can easily negate any environmental advantages obtained from valorising by-products (Tonini et al. 2015). These environmental trade-offs are also called pollution swapping (De Vries et al. 2015a). Pollution swapping can occur between different stages inside or outside the production system and also between different environmental impacts. To summarise, sustainable valorisation of by-products strongly depends on how competition between the original and aimed uses are included and therewith the related environmental consequences (e.g. consider Plevin et al. (2013) and Hedegaard et al. (2008)).

Important environmental impacts related to the valorisation of agricultural by-products include GHG emissions such as carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O), LULUC and fossil energy depletion (Tilman et al. 2002). Water consumption may also be of importance for industrial processing of by-products (Lynch et al. 2016). Other impacts are eutrophication and acidification potentials which are more related to land use and production of the main crops. Impact on LULUC becomes relevant in light of changing uses of biomass or increasing demands (Parajuli et al. 2015). Where land use change is related to direct changes in cultivation and use of biomass, for example C sequestration differences when using straw for bioenergy, indirect land use change (iLUC) is related to shifting of production to other areas and biomes in the world, for example land expansion causing deforestation (Plevin et al. 2010). C emitted and sequestered in the soil is directly linked to the C and GHG balance and is critical for maintaining soil quality and therewith crop productivity (Wiloso et al. 2016). Soil quality and crop productivity in turn are critical for economic and social stability of the production system (Lal 2004; Pawelzik et al. 2013). Because of this central role, soil as the basic production substrate for all biomass will have to be the central pivotal point around which decisions for valorising biomass and agricultural by-products will have to take place. One question that will have to be answered is: 'What must we feed the soil, or more specifically, the soil biota when biomass is used for other purposes?'

Consequential life cycle assessment (LCA) is one of the most comprehensive tools available to assess the environmental consequences and trade-offs of valorising agricultural by-products. This does imply, however, that system expansion is applied wherever changes in the original use and applications of biomass and its end products occurs.

This chapter serves to provide a literature overview of opportunities to reduce the environmental impact of valorising agricultural by-products through different treatment pathways. In the following headings, we provide an overview and discussion on:

- the main treatment pathways for agricultural by-products,
- as an example the available and collectable by-products in Northwestern Europe (NW EU),
- environmental impacts including GHG reduction potential and soil C sequestration related to the valorisation of by-products through the main treatment pathways,
- future developments for reducing environmental impact of valorising by-products.

2 Main treatment pathways for valorising agricultural by-products

The main treatment technologies for valorising by-products roughly include mechanical treatment, for example pressing and breaking, biochemical treatment, for example fermentation and enzymatic conversion and (thermo)

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Table 1 Bio-refinery platforms based on converting biomass feed stocks to energy and bio-based products (IEA-Bioenergy 2018)

Route	Feed stock	Main treatment technologies	Intermediate and end products
1	Grass silage and food residues	Mechanical and fermentation	Bio-plastics, insulation material, fertiliser and electricity
2	Wood chips	Mechanical and (thermo)chemical	Pulp, paper, turpentine, tall oil, bark, electricity and heat
3	Starch	Mechanical, enzymatic and fermentation	Bioethanol and feed
4	Wood chips	Mechanical, enzymatic, fermentation and thermochemical	Bioethanol, electricity, heat and phenols
5	Oil seed crops	Mechanical and chemical	Bio-diesel, glycerine and feed
6	Oil-based residues	Mechanical and chemical	Bio-diesel, glycerine, bio-oil and fertiliser
7	Wood chips	Mechanical and (thermo)chemical	Biofuels, electricity, heat and waxes
8	Straw	Mechanical and (thermo)chemical	Biofuels and methanol

chemical treatment, for example pyrolysis and incineration (Fig. 1). Bio-refineries use a combination of technologies to convert inputs from biomass, for example the (hemi)cellulose, sugars, and starch and deliver outputs in the forms of for example energy, chemical feed stocks, biofuels and animal feed. Bio-refineries are considered as an alternative to conventional oil refineries (De Jong and Jungmeier 2015). The IEA Bioenergy consortium has defined eight bio-refinery platforms based on converting biomass feed stocks to energy and various intermediate and end products (Table 1). Only one of them (number 8) includes agricultural by-products as feedstock.

Primary residues such as straw contain lignocellulose that can be converted to for example biofuel or chemical feedstock. Secondary residues like beet pulp and distillers dried grains (DDG) are currently used to feed cattle, pigs and poultry. However, they also contain valuable components, such as hemicellulose, pectin and protein that can be used as either food ingredients, for chemicals and biofuels (Panagiotopoulos et al. 2010). Manure contains valuable nutrients and organic matter that can be utilised either as fertiliser or for nutrient extraction and energy or biofuel (De Azevedo et al. 2017).

3 Availability and collectability of by-products

3.1 Availability of agricultural by-products and their original use

Availability of the main agricultural by-products worldwide is estimated at 2.8 bn t DM for cereal crops, 3.1 bn t DM for the main cereal crops and legumes and 3.7 bn t DM for the main food crops (Lal 2005). In NW EU the available by-products are estimated at approximately 111 million tonnes (Mt) DM/year (Table 2). Primary residue from crop production represents approximately 64.7 Mt DM/year (or 59%), secondary residue from processing crops approximately 17.6 Mt DM/year (or 16%) and liquid manure approximately 28.2 Mt DM/year (or 26%). In the EU27 the availability of by-products was estimated at 258 Mt dry primary residue per year (Scarlat et al. 2010) and 597 Mt liquid pig and cattle manure (Foged et al. 2012).

Most of the by-products are produced in France (38%) and Germany (29%) followed by the United Kingdom (14%), Denmark (5%) and the Netherlands (4%). Of the primary residue, 42% is produced in France, 30% in Germany, 14% in the United Kingdom and 6% in Denmark. Approximately two-thirds of the secondary residue and half of the liquid manure are produced in France and Germany. Benelux covers approximately 14% of the liquid manure production.

Most of the primary residue is used for soil conditioning or SOM build-up by leaving it in the field (around 63%). The other part (approximately 37%) is used for litter in animal housing where afterwards it is returned to the soil for crop nutrition and SOM build-up (Helin et al. 2012). Less than 1% of straw is used for combined heat and power production. Secondary residues are often used for either animal feed, for example as a protein source, or for human food consumption. Some of the secondary residues, such as beet pulp, molasses and potato peels are also used for bioenergy production through anaerobic digestion. Manure is mostly used as fertiliser and for SOM build-up. About 11.5% of the manure in the EU is treated, of which 3.1% is treated by separation, 6.4% is anaerobically treated to produce bioenergy and another 2% is further treated by other technologies (Foged et al. 2012).

3.2 Collection of by-products

The collection of by-products includes the harvesting from the field or from the industrial application or animal housing system. The collection of primary residue from the field is dependent on many factors including yield, environmental conditions, available equipment, plant variety and crop rotation (Scarlat et al. 2010). Collection rates of residue vary between 30% and 50% depending on the need for SOM. Collection should occur approximately within a range of 50 km from the processing plant (Monforti et al. 2013) in order to limit the travel range and need for fuel. Collection rates of primary residue in Table 2 were based on the 'collectable' residue rates. Optimal collection rates may be higher depending on the local opportunity (Monforti et al. 2015). Similarly, for primary residues such as leaves, the collectable range will lie in the same order of magnitude but collection is less attractive due to its low availability. Leaves do contain a fair amount of protein (roughly 30% of the dry

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Table 2 Availability of some of the main agricultural by-products in North Western Europe: Belgium (BE), Denmark (DK), Germany (DE), Ireland (IE), France (FR), Luxembourg (LU), the Netherlands (NL), Sweden (SE) and the United Kingdom (UK)

Agricultural by-products (Mt DM/year)	BE	DK	DE	IE	FR	LU	NL	SE	UK	Total	Original use	Source
Primary residue												
Crop residue (wheat, barley, rye, oat, maize, rice, rapeseed, sunflower)	0.97	3.45	18.2	0.73	25.6	0.06	0.61	2.03	8.37	60.0	Litter and SOM	Monforti et al. (2013) Collectable residue
Potato leaves	0.09	0.04	0.27	0.01	0.19	0	0.18	0.03	0.16	0.97	SOM	Eurostat (2018a)[a]
Beet leaves	0.19	0.12	1.20	0	1.39	0	0.24	0.10	0.35	3.61	SOM	Eurostat (2018a)[b]
Other root crops leaves	0.01	0.01	0.01	0.01	0.02	0	0	0	0.05	0.10	SOM	Eurostat (2018a)[c]
Total	1.26	3.62	19.7	0.75	27.2	0.06	1.03	2.16	8.93	**64.7**		
Secondary residue												
Beet pulp	0.96	0.18	0.05	0	0.38	0	0.02	0.16	0.43	2.19	Feed/energy	CEFS (2016)[d]
Molasses	0.11	0.07	0.71	0	0	0	0.18	0.05	0.10	1.22	Food/feed/energy	CEFS (2016)[d]
Distillers dried grains (DDG)	0.37	0	0.76	0.01	1.55	0	0.37	0.16	0.64	3.86	Feed	ePURE (2016)[e]
Soy bean meal	0	0	0.04	0	0.20	0	0	0	0	0.23	Feed/protein	Eurostat (2018a)[f]
Rapeseed meal	0.03	0.41	3.03	0.02	2.95	0.01	0.00	0.19	1.30	7.94	Feed/protein	Eurostat (2018a)[g]
Sunflower meal	0	0	0.03	0	0.95	0	0	0	0	0.98	Feed/protein	Eurostat (2018a)[h]
Brewers grain	0.09	0.03	0.42	0.03	0.09	0	0.11	0.02	0.19	0.98	Feed/protein	the Brewers of Europe (2016)[i]
Potato peels	0.02	0.01	0.07	0	0.05	0	0.04	0.01	0.04	0.24	Feed/protein/energy	Eurostat (2018a)[j]
Total	1.22	0.70	4.33	0.06	4.62	0.01	0.36	0.44	2.05	**17.6**		

(*Continued*)

Table 2 (*Continued*)

Agricultural by-products (Mt DM/year)	BE	DK	DE	IE	FR	LU	NL	SE	UK	Total	Original use	Source
Manure												
Liquid pig	0.50	0.99	2.15	0.12	1.18	0.01	0.97	0.12	0.37	6.39	Fertiliser/SOM	Foged et al. (2012)[k]
Liquid cattle	0.96	0.58	4.88	2.53	7.01	0.07	1.51	0.58	3.73	21.8	Fertiliser/SOM	Foged et al. (2012)[k]
Total	1.45	1.57	7.03	2.65	8.19	0.08	2.47	0.70	4.10	**28.2**		

Values 0 are <0.05. SOM = soil organic matter.
[a] Based on 2.25 t DM/ha (Tijmensen et al. 2002; Neeteson and Greenwood 1987).
[b] Based on 6.75 t DM/ha (Tijmensen et al. 2002).
[c] Assumed similar as potato leaves.
[d] 5-year average 2011–16.
[e] 2016 data converted assuming 0.65 kg DDG/L of ethanol and 90% DM (US Grains Council 2012).
[f] Average of 2013–18. Based on 77% extraction and 89% DM (ingredients101 2018).
[g] Average of 2013–18. Based on 66% extraction (Nova 2007) and 88.5% DM (KW 2018).
[h] Average of 2013–18. Based on 75% extraction and 89% DM (Le Clef and Kemper 2015).
[i] Based on 20 kg brewers grain/hL of beer (Mussatto et al. 2006) and 22% DM (Duynie 2018b).
[j] Average of 2013–18. Based on 38 kg peels/t of potatoes (Ahokas et al. 2014) and 16.5% DM (Duynie 2018a).
[k] Based on 8.23% DM for liquid pig manure and 7.45% DM for liquid cattle manure Menzi (2002) in De Vries et al. (2015a).

weight) that may be interesting for food and feed applications (Kiskini 2017). The collection of secondary residues and manure depends on the original use and storage. Secondary residues and manure are more easily collected than primary residues as they are stored already.

4 Environmental impact of valorising by-products

4.1 Primary residue

Table 3 provides an overview of the environmental impact of the main treatment pathways for agricultural by-products. Studies were often focused on crop residues such as corn stover and wheat or rice straw for biofuel production purposes and manure as they are most abundant, for example De Azevedo et al. (2017), Morales et al. (2015) and Prapaspongsa et al. (2010). Biofuel production from valorising wheat straw and corn stover reduces GHG emissions between 262 and 903 kg CO_2-equivalents (eq)/t DM/year compared to its fossil reference. However, effects of changing soil C sequestration as a result of removing crop residues are not included in all studies (e.g. the 903 kg CO_2-eq/t DM in Table 3). These studies, therefore, overestimate the GHG reduction potentials. In Section 5.2 we further elaborate on this issue and the impact on the GHG reduction estimates. Assuming a maximum reduction potential of 903 kg CO_2-eq/t DM for primary residue, GHGs can be reduced by up to 54.2 Mt CO_2-eq on a NW EU scale (Table 4). This represents about 2.2% of the total GHG emissions and 22% of the GHG emissions from agriculture (for the given countries in Table 4: 2447 Mt CO_2-eq total and 249 Mt CO_2-eq agricultural emissions in 2015, Eurostat (2018b)). When including changes in C sequestration, the estimated GHG reduction potential is 37 Mt CO_2-eq. Fossil fuel depletion (FFD) varies between 1120 and 2219 MJ/t DM/year (input of fossil-based and primary energy for conversion). Fossil fuel savings run up to 18 GJ/t DM/year (output of bioenergy) (Cherubini and Ulgiati 2010). On the contrary, the acidification potential varied between 1.64 and 1.95 kg SO_2-eq/t DM/year and eutrophication potential varied from 0.82 to 1.09 kg PO_4-eq/t DM/year. Acidification potentials were lower compared to the fossil reference (up to 2.43 kg SO_2-eq/t DM/year) and eutrophication potentials are generally higher compared to the fossil reference (up to 0.36 kg PO_4-eq/t DM/year). Higher eutrophication potentials were related to using fertiliser in the production of the main crop.

4.2 Secondary residue

Little information is available about the environmental impact of valorising secondary residues. Some work has been done on the valorisation of glycerol from rapeseed oil and potato juice to propionic acid and also anaerobic co-digestion of residues with manure (dealt with in Section 4.3).

Table 3 Overview of environmental impacts (SOM = soil organic matter, GHG = greenhouse gases, FFD = fossil fuel depletion, Resp inorg = respiratory inorganics, Acidif = acidification potential, Terr Eutr = terrestrial eutrophication potential and Aq Eutr = aquatic eutrophication potential) of the main treatment pathways for valorising agricultural by-products

							Environmental impact							
By-product	Treatment pathway	End product/ Market	Country	Original use	Substitute included (yes/no)?	Unit	GHG[y/n] (kg CO_2-eq)	GHG saved comp to fossil-based reference (kg CO_2-eq)	FFD (MJ)	Resp inorg (kg $PM_{2.5}$-eq)	Acidif (kg SO_2-eq)	Terr Eutr (m^2 UES)	Aq Eutr (kg NO_3-eq)	Source
Primary residue														
Corn stover	Bio-refinery	Bioethanol, heat, electricity and phenols	AU	SOM	Yes	per t DM per year	287[y]	333	1677[a]	n.d.	1.95	n.d.	1.09 kg PO_4-eq	Cherubini and Ulgiati (2010)
Wheat straw	Bio-refinery	Bioethanol, heat, electricity and phenols	AU	SOM	Yes	per t DM per year	273[y]	262	1887[a]	n.d.	1.64	n.d.	0.82 kg PO_4-eq	Cherubini and Ulgiati (2010)
Straw	Bio-refinery	Diesel and methanol		SOM	No	per t DM per year	93.2[n]	903	2219[a]	n.d.	n.d.	n.d.	n.d.	IEA-Bioenergy (2018)
Secondary residue														
Glycerol and potato juice	Bio-refinery (fermentation)	Propionic acid	DK	Heat/ fertiliser	No	per t of rapeseed	~39[n]	~56	~1120[b]	n.d.	n.d.	n.d.	0.17 kg PO_4-eq	Ekman and Börjesson (2011)
Potato peels	Bio-reactor	Hydrogen, feed	NL	Feed	No	per t DM	−25[n]	25	−413	n.d.	n.d.	n.d.	n.d.	Djomo et al. (2008)

Manure													
Liquid pig	Anaerobic mono-digestion	Energy and Fuel	DK, NL	Fertiliser	Yes	per t manure	−16 to -40^{y}	16 to 147	−117	0.13	0.09	144	−0.13 to 0.93	Hamelin et al. (2011) and De Vries et al. (2012a)
Liquid pig	Anaerobic co-digestion	Energy and Fuel	DK, NL	Fertiliser and Feed	Yes	per t substrate	−89 to 105^{y}	89 to −105	−2398 to 1390	−0.13 to 0.36	−0.33 to 1.61	n.d.	−0.34 to 1.12	Hamelin et al. (2011) and De Vries et al. (2012a)
Liquid pig	Incineration	Electricity and Heat	DK	Fertiliser	Yes	per t manure	−11.8^{n}	119	n.d.	0.04	n.d.	64	0.58	Prapaspongsa et al. (2010)
Liquid pig	Gasification	Energy and Fuel	DK	Fertiliser	Yes	per t manure	−8.8^{n}	116	n.d.	0.05	n.d.	64	0.58	Prapaspongsa et al. (2010)
Liquid pig	Separation	SOM	DK, NL	Fertiliser	Yes	per t manure	37 to 88^{y}	16 to −3	348 to 181	0.76 kg PM_{10}-eq	4.8 to 5.9	n.d.	1.02 to 4.21	Ten Hoeve et al. (2014) and De Vries et al. (2012b)
Liquid cattle	Integrated management	Fertiliser and SOM	NW EU	Fertiliser	Yes	per t manure	−78.6^{n}	190	−805	0.05 kg PM_{10}-eq	0.79	n.d.	0.04	De Vries et al. (2015a)
Liquid pig	Integrated management	Fertiliser and SOM	NW EU	Fertiliser	Yes	per t manure	−45.9^{n}	472	−1043	0.27	2.44	n.d.	0.93	De Vries et al. (2015a)

n.d. = no data. Negative numbers under GHG emissions and other impacts represent a net reduction in environmental impact. Negative numbers under GHG saved represent net additions in GHG. GHG$^{y/n}$ means that either soil C sequestration is included or excluded/unknown. Substitute included (Yes/No) means that a substitute is considered for the change of the original use of the by-product.

[a]These figures include only the fossil fuel input for producing bioenergy.

[b]Primary energy consumption.

Table 4 Estimated maximum GHG reduction potential of valorising agricultural by-products through the main treatment pathways in NW EU based on Tables 2 and 3

Potential GHG reduction ($Mt\ CO_2$-eq)	BE	DK	DE	IE	FR	LU	NL	SE	UK	Total
Primary residue										
Crop residue (wheat, barley, rye, oat, maize, rice, rapeseed, sunflower)	0.87	3.11	16.4	0.66	23.1	0.05	0.55	1.84	7.56	**54.2**
Crop residue with changes in C sequestration[a]	0.59	2.11	11.1	0.44	15.6	0.04	0.37	1.24	5.11	36.7
Secondary residue										
Rapeseed meal	0	0.04	0.29	0	0.28	0	0	0.02	0.12	0.76
Potato peels	6.1E-04	2.6E-04	1.7E-03	5.9E-05	1.2E-03	2.8E-06	1.1E-03	1.3E-04	9.1E-04	0.01
Total	0	0.04	0.29	0	0.28	0	0	0.02	0.13	**0.77**
Manure										
Liquid pig	2.85	5.66	12.3	0.67	6.78	0.04	5.54	0.70	2.11	36.6
Liquid cattle	2.43	1.48	12.4	6.44	17.8	0.19	3.83	1.48	9.50	55.6
Total	5.28	7.14	24.7	7.12	24.6	0.23	9.37	2.18	11.6	**92.3**
										147

Belgium (BE), Denmark (DK), Germany (DE), Ireland (IE), France (FR), Luxembourg (LU), the Netherlands (NL), Sweden (SE) and the United Kingdom (UK).
Values 0 are <0.05.
[a] C-sequestration estimates included based on the calculation in Section 4.4.

GHG reduction potential of producing propionic acid was estimated at 56 kg CO_2-eq/t rapeseed. However, a substitute for the original use of the glycerol is not considered and therefore possibly overestimating the reduction potential. Glycerol is currently used for heat and power production because it is produced in excess of market demand. This means that when adding a substitute, additional GHG emissions will be emitted. Assuming natural gas as the substitute with 0.07 kg CO_2-eq/MJ and a lower heating value of glycerol of 16.5 MJ/kg, an additional amount of 45 kg CO_2-eq will be emitted per ton of rapeseed. This additional amount negates 80% of the saved GHGs (45/56 kg CO_2-eq). On a NW EU basis, this means a potential reduction in GHG emission of up to 0.76 Mt CO_2-eq (without a substitute) which represents only 0.03% of the GHG emissions from agriculture. With a substitute for heat production this would be approximately 0.15 Mt CO_2-eq or 0.01% of the GHG emissions from agriculture. Eutrophication potential was 0.17 kg PO_4-eq/t rapeseed.

Another study examined the production of hydrogen from potato steam peels instead of using it as animal feed (Djomo et al. 2008). Results of the study showed a potential to reduce GHGs up to a factor of two to three compared to a fossil reference. GHG emissions were −25 kg CO_2-eq/t DM potato peels and FFD was −303 MJ/t DM potato peels. Again, a substitute for animal feed when using the potato peels was not included. On a NW EU basis, this means a potential reduction in GHG emission of <0.01 Mt CO_2-eq which represents <0.01% of the GHG emissions from agriculture (Table 4).

4.3 Manure

There exist numerous environmental impact studies on manure treatment pathways, from simple separation to the production of energy and strategies for integrated management (Table 3). Strategies for integrated management encompass (technological) changes in all phases of the manure management system from storage through processing and (field) application. This includes acidification of manure, separation and synchronising of manure application with crop demand and also soil treatment (De Vries et al. 2015a). The GHG emission of valorising manure varies between a positive emission of 105 kg CO_2-eq/t manure to a reduction of 472 kg CO_2-eq/t manure (Table 3). On a NW EU basis, this means a potential reduction in GHG emissions of up to 92 Mt CO_2-eq which represents about 4% of the total GHG emissions and 37% of the GHG emissions from agriculture (Table 4). FFD varies between 348 and 1390 MJ/t manure. Fossil fuel savings run up to 1043 MJ/t manure through the production of energy (De Vries et al. 2015a). Acidification potential varied between −0.33 and 5.9 kg SO_2-eq/t manure and eutrophication potential varied from −0.34 to 4.21 kg NO_3-eq/t manure.

4.4 LULUC and soil carbon sequestration

LULUC and consistent removal of by-products pose adverse effects on soil C sequestration and soil quality in the short and long run (Lal 2005). LULUC-induced emission may run up to 4.1 t CO_2-eq per demanded hectare (ha) or 1.2–1.4 t CO_2-eq/t dry biomass (Tonini et al. 2015). As a part of LULUC, soil C sequestration varies between 50 and 1000 kg of C/ha/year corresponding to 183 and 3667 kg CO_2/ha/year (Goglio et al. 2015; Lal 2004). Sequestration is affected by the type of crops that are produced, their rotation sequence and the type of soil management that is applied, for example tillage or no tillage (Lal 2004). These parameters determine the amount of C that is turned into humus and stored in the soil for a longer period of time.

In the current literature, it is estimated that 40–50% of crop residues can be removed without diminishing the stable soil C pool (Monforti et al. 2013, 2015; Scarlat et al. 2010). In no-till farming, residue removal rates could be as high as 82% (Scarlat et al. 2010), but in other circumstances removal rates of 30–40% may already induce hazard to soil quality and the stable soil C pool (Lal 2005).

To estimate soil C sequestration, the C sequestration efficiency can be used. For example, assuming a conservative sequestration efficiency of 17% for wheat straw (Hua et al. 2014), a DM content of 90%, an organic matter content of 94% of DM and 50% of C in the organic matter, approximately 80 kg of C or 292 kg of CO_2/t of dry straw can be sequestered yearly. On a NW EU basis, this approximates to 4.8 Mt of C or 18 Mt of CO_2. This is roughly 32% of the GHG reduction potential of primary residue (18/54.2 Mt CO_2-eq, Table 4) and 12% of the total GHG reduction potential (18/147 Mt CO_2-eq, Table 4).

For pig and cattle manure, assuming a C sequestration efficiency of 11% and 6.5% organic matter, the sequestration is roughly 4 kg of C or 14 kg of CO_2/t wet manure yearly. On a NW EU basis, this approximates 1.4 Mt of C or 5.2 Mt of CO_2 being roughly 1% of the total GHG reduction potential (1.4/147 Mt CO_2-eq in Table 4).

Concluding, soil C sequestration strongly influences the GHG reduction potentials, especially for primary residue, and therefore needs to be assessed when scientific studies lack such inclusion. Using the C sequestration efficiency is a straightforward way to simply estimate the (reduced) C sequestration and its impact on the GHG reduction potential.

5 Future opportunities and perspectives

5.1 Availability and collectability of by-products

The availability and collectability of by-products is critical for valorisation opportunities. Here, the available primary residue was based on the 'collectable estimate' given in Monforti et al. (2015). They also provided an 'optimal estimate' that was 42.9% higher relative to the default collection. Optimal collection rates of primary residue for NW EU would be approximately 86 Mt DM/year with additional potential to reduce GHG emissions up to 77 Mt CO_2-eq from primary residue without lost C sequestration and 52 Mt CO_2-eq with lost C sequestration. Agricultural by-products, however, are not always optimally distributed meaning that collecting residues may not be economically attractive. To be economically feasible for exploitation, valorisation plants will have to be placed at strategic distances from where the residue is located (Golecha and Gan 2016).

Next to collectability, the availability of by-products will depend on their competing uses. When using by-products for different treatment pathways they compete either with the conventional raw materials or with different applications of the same by-product. This may increase their market values. In recent years, prices of by-products used for anaerobic co-digestion, such as energy crops and industrial by-products, increased up to €72/t due to the increased demand for anaerobic digestion (Velghe and Wierinck 2013). Moreover, when by-products were originally used as animal feed, the GHG reduction potential of anaerobic digestion was negated by LULUC, e.g. De Vries et al. (2012a). Since agricultural by-products are dependent on a main product, they remain limited in their availability. In other words, the amount of by-products will generally not increase if market demand rises. Therefore, a good balance between different treatment pathways will be needed to not overexploit the available by-products, resulting in skyrocketing prices and increased consequences for the environment. This will require not only good guidelines on a national and international level but also a level playing field for alternative uses of by-products such as bio-based chemicals and materials (Carus et al. 2011).

Availability of by-products may increase through intensification of agricultural production and using marginal lands. Crop production increased by 2.5% annually since the 1960s up to the late 1990s. Estimates now show that crop production may increase but this depends strongly on the genetic yield potential and environmental factors (Cassman 1999). The environmental consequences of intensifying agriculture has, however, been severely debated and remains controversial.

Asia and the United States will be the main producers of biomass and by-products in the future (Daioglou et al. 2016). Exchange of biomass between countries will not be very likely due to the degradability and bulkiness of the material but will need to be valorised locally as much as possible. Valorisation of using biomass will depend much on the local business opportunities.

5.2 Environmental assessment and reduction potential of valorising by-products

The total maximum GHG reduction potential of 147 Mt CO_2-eq (Table 4) represents about 6% of the total GHG emissions and 59% of GHG emission from agriculture in NW EU. Valorising all wheat straw on a worldwide scale (2.8 bn t DM/year) through bio-refinery with a GHG reduction potential of 611 kg CO_2-eq/t DM (903 kg CO_2-eq/t DM minus 32% C sequestration loss, Table 3 and Section 4.4.), this could be as much as 1.71 bn t CO_2-eq or <4% of the worldwide GHG emissions (approximately 45 bn tons CO_2-eq in 2014). Previous estimates do not yet include valorisation of all by-products as studies on environmental impact reduction are still limited (Tables 3 and 4). The main potential to reduce environmental impact, however, lies in the valorisation of animal manure (63% of the total reduction potential, or 92.3 Mt CO_2-eq of 147 Mt CO_2-eq). Manure management contributes up to 50% of the N_2O and around 20% of the CH_4 emissions in the livestock sector of the EU (Leip et al. 2015). Changes in manure management are often limited to reducing GHGs and affect other emissions in no or limited ways (Table 3). New strategies for integrated manure management show opportunities to further reduce environmental impact but have yet to be implemented in practice.

Few environmental assessments have been done for valorising primary residues such as beet and potato leaves and secondary residues coming from industry after processing. Beet and potato leaves contain protein that can be extracted and used for food products (Tamayo Tenorio et al. 2016). Secondary residues that have been researched for their human food potential, such as brewers' spent grain have shown to be potentially interesting as a protein source as well (Lynch et al. 2016). We estimate that reduction potentials of these pathways will be <1–2% of the total GHG emission in NW EU. However, changing the original use of secondary residues may well introduce trade-offs such as iLUC and therewith negate the GHG reduction potential. Although limited, it is still important to understand and assess the environmental impact of valorisation in a preliminary stage to support decision making.

5.3 Environmental trade-offs and pollution swapping in the chain

Environmental trade-offs and pollution swapping occurs when one environmental impact is reduced and another increases as a direct or indirect consequence of a taken action. Pollution swapping occurs on different scales from process level up to chain level and may include various environmental impacts. In manure management, for example, pollution swapping occurs when covering manure storages or when injecting manure instead of broadcast spreading (De Vries et al. 2015b). When covering manure storages and injecting manure, ammonia emissions are reduced but nitrous oxide emissions are increased. Swapping may also occur on chain level when more N is contained in the manure and is potentially lost during field application.

When valorising agricultural by-products, pollution swapping between environmental impacts also occurs as shown in Table 3. Where GHG emissions were reduced, other environmental impacts such as eutrophication potential or FFD were increased. This is common due to the needed substitute for the original use of the by-product when used for other treatment pathways. The substitute needed may well be an additional amount of crop and consequently an amount of land to produce the needed crop. Next to LULUC, inputs such as fertiliser and manure are needed for production and cause eutrophication and other land-related impacts (Kim and Dale 2005).

Another important environmental trade-off that occurs when by-products are valorised is the shift in C sequestration and the related GHG emissions as demonstrated in Section 4.4. Changes in the C balance as a result of LULUC and C sequestration are essential for estimating GHG reduction potentials when valorising by-products. Wiloso et al. (2016) provide a comprehensive and straightforward overview of how the C balance is related to LULUC and C sequestration. C sequestration, when lacking in environmental assessments, can easily be estimated based on the C sequestration efficiency.

A solution to avoid pollution swapping lies in the (re-)design of the complete valorisation chain. In such a process all requirements of the stakeholders in the chain need to be included. The environment can be included as a 'stakeholder' in order to establish requirements for environmental impact reduction and trade-offs. (Re-)design of a complete system, however, is a complex task and requires strong dedication of stakeholders and a good process manager (De Vries et al. 2015b).

Other environmental impacts not considered here but that are relevant when valorising by-products are water consumption, toxicity effects and human health impacts. When valorising by-products to biofuels, water consumption of processing varies between roughly 800 and 4000 L of water per litre of biofuel produced (Singh et al. 2011). When water availability is limited, the scarcity of water will be relevant. This may require the use of a water stress index (Pfister et al. 2009). The water stress index relates the water use to the local availability or scarcity and thus impact of using water.

Other factors in addition to environmental ones, such as social and economic factors, will have to be included when making decisions on how to valorise agricultural by-products. Examples include increased logistics in small rural areas due to the collection of residues and market prices. In this way a broader scope to sustainability of the treatment pathway will be ensured.

5.4 Economic viability of valorising by-products

The economic viability of valorising agricultural by-products will be determined by the cost of collecting the by-products and their market prices, the investment and operations cost of the treatment pathway and the market value and opportunities for the end products. Market prices of primary residues were estimated at between €16 and €72/t ($19 and $84) (Carriquiry et al. 2011) and for some secondary residues such as brewers' grain at €35/t (Lynch et al. 2016). Typically, manure has a negative value in NW EU meaning the farmers need to pay for removing the manure from their farms. Removal prices vary between €15 and €25/t for liquid pig manure in the Netherlands (NVV 2016).

Bio-refinery pathways are estimated to be viable with revenues between €5.5 and €220 M/year or €394 to €1410 /t of residue per year (IEA-Bioenergy 2018). For straw to diesel and methanol this was estimated to be €160 M/year or €1068/t/year. Other, less economically viable pathways include anaerobic digestion. Anaerobic digestion has shown little economic feasibility if not subsidised or if the digestate could not be sold (Astill and Shumway 2016; De Dobbelaere et al. 2015).

6 Summary and conclusion

Agricultural by-products consist of a wide range of biomass types including crop residue, residues from processing industries (secondary residue) and manure. The main treatment technologies for valorising agricultural by-products include mechanical treatment such as pressing and breaking, biochemical treatment such as fermentation and enzymatic conversion and (thermo)chemical conversion such as pyrolysis and incineration. Bio-refineries are comprised of a combination of technologies using multiple parts of the biomass to produce bio-based end products. Integrated approaches such as bio-refineries and integrated manure management strategies offer opportunities to valorise by-products while reducing environmental impact such as GHG emissions and FFD (Anon 2014).

On a worldwide scale, crop residue is available at approximately 2.8 bn t DM/year (Lal 2005). Assuming a default collection rate for crop residues (Monforti et al. 2015), approximately 111 Mt DM of agricultural by-products are available in NW EU on a yearly basis. Primary residues comprise 59%, secondary residues comprise 16% and liquid pig and cattle manure comprise 26% of the available by-products. Crop residues can be collected at a rate of approximately 40–50% without diminishing soil C stocks and reducing soil health. Optimal collection rates for crop residues could be 43% higher but depend on specific conditions.

The saved GHGs, compared to a fossil reference, when valorising primary residue were between 262 and 903 kg CO_2-eq/t DM (without lost C sequestration). Including a simple estimate of C sequestration, the maximum reduction potential was 32% lower. The maximum estimated GHG reduction potential for primary residue in NW EU was 54 Mt CO_2-eq or 2% of the total GHG emissions without lost C sequestration and 37 Mt CO_2-eq with lost C sequestration. On a worldwide scale this could be as much as 1.71 bn t CO_2-eq or <4% of the worldwide GHG emissions. GHG reductions were often accompanied by higher eutrophication potentials. The saved GHGs of valorising secondary residue were between 25 and 56 kg CO_2-eq/t DM. GHG reductions were <0.1% on NW EU scale. Valorisation of secondary residue was also highly susceptible to LULUC. The saved GHGs when valorising manure were between 105 and 472 kg CO_2-eq/t DM. The maximum GHG reduction potential for manure in NW EU was 92 Mt CO_2-eq or 4% of the total GHG emissions.

In total, the saved GHG emissions of valorising agricultural by-products in NW EU ran up to approximately 147 Mt CO_2-eq or 6% of the total and 59% of the agricultural GHG emissions. Altering manure management had the greatest potential to reduce GHG emissions (63% of the total estimated reduction potential). The opportunity to reduce GHG emissions is, however, easily diminished if LULUC is involved or soil C sequestration is lost when removing crop residue. A simple approach based on the C sequestration efficiency showed that a C sequestration potential of about 262 kg CO_2/t dry straw is lost when residue is removed. This represented about 32% of the GHG reduction potential when valorising primary residue. Removing crop residue from the field requires estimates of soil C sequestration. This is relevant especially when answering the question: 'What must we feed the soil, or more specifically, the soil biota when biomass is used for valorisation?' Not answering this question properly will lead to reduced sequestration and SOM and subsequently to reduced soil health and long-term productivity.

Other trade-offs besides LULUC and C sequestration include increased eutrophication potentials. Eutrophication is related to producing a substitute to replace the original use of the by-product. Trade-offs are important to take into account when establishing new valorisation chains and treatment pathways.

Future opportunities for reducing environmental impact of valorising agricultural by-products lie in increased availability of by-products through improved production systems such as conservation agriculture and no-till farming (Kim et al. 2009). The availability of crop residue in NW EU can run up to 86 Mt DM in optimal collection circumstances further reducing GHG emissions up to 77 Mt CO_2-eq (instead of 52 Mt CO_2-eq under collectable residue rate) without lost C sequestration or 54 Mt CO_2-eq with lost C sequestration. Information was limited on the environmental impact of valorising primary residue such as beet leaves and secondary residues from industries. The GHG reduction potential was estimated at <0.1% of the total GHG emission in NW EU but may easily be negated when LULUC is induced.

Finally, next to availability and environmental aspects, the viability of valorising agricultural by-products depends on the economics and revenues that can be generated from the end products. A level playing field for stimulating other bio-based products than fuel or energy is deemed necessary for best environmental results when valorising agricultural by-products.

7 Where to look for further information

The cited literature provides a good overview of the current assessments done and the information on the availability of agricultural by-products. The IEA-Bioenergy recently published information on valorisation of biomass to bioenergy. This includes information on the environmental impacts and value chains of different biorefinery treatment pathways (IEA-Bioenergy 2018), for example https://www.iea-bioenergy.task42-biorefineries.com/en/ieabiorefinery/Factsheets.htm.

Knowledge institutes such as the Nova Institute in Germany, Wageningen University and Research in the Netherlands and the Joint Research Centre of the European Union regularly publish data and information on the bioeconomy. They communicate through reports and newsletters available on their websites (Nova 2018a,b; WUR 2018; JRC 2018), for example:

- http://news.bio-based.eu/biomass-cascading-use-equals-best-lca/
- https://www.wur.nl/en/Research-Results/Research-Institutes/food-biobased-research/about/Biobased-Products-2.htm
- https://ec.europa.eu/jrc/en

General information on bioeconomy networks and research in the European Union can be found at (BBE 2018; EU 2018):

- https://www.biobasedeconomy.nl/
- https://ec.europa.eu/research/bioeconomy/index.cfm

Data on the production of crops, biofuels and industrial products can be found at different locations (Eurostat 2018a; FAO 2018; Oil World 2018), for example:

- https://ec.europa.eu/eurostat/data/database
- http://www.fao.org/statistics/en/
- https://www.oilworld.biz/

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