APPENDIX

12 Glossary

AOX	Adsorbable Organic halides - a measure of the organic halogen load of a material
BSE	Bovine Spongiform Encephalopathy, commonly known as mad cow disease - a transmissible spongiform encephalopathy and fatal neurodegenerative disease in cattle that causes a spongiform degeneration of the brain and spinal cord
BTEX + S	Benzene, Toluene, Ethylbenzene, the ortho-, para- & meta-Xylenes and Styrene - the most abundant volatile organic compounds that can occur in petroleum-derived and biomass ash as a result of incomplete combustion
ChemP	The chemical precipitation of phosphorus with metal salts in a waste water treatment configuration
CMC	Component Material Category in the proposed Revised Fertiliser Regulation[27]. A CE marked fertilising product shall consist solely of component materials complying with the requirements for one or more of the CMCs. This project evaluates techno-scientific evidence in view of a possible inclusion of STRUBIAS as CMC in the Revised EC Fertiliser Regulation.
DAP	Di-Ammonium Phosphate, a water-soluble mineral fertiliser that contains nitrogen and phosphorus
DG GROW	The Directorate-General for Internal Market, Industry, Entrepreneurship and SMEs is the European Commission service that is leading the process of laying down rules on the making available on the market of CE marked fertilising products
DG SANTE	The Directorate-General for Health and Food Safety is a Directorate-General of the European Commission, responsible for the implementation of European Union laws on the safety of food and other products, on consumers' rights and on the protection of people's health
EBC	European Biochar Certificate - a voluntary European industrial standard for pyrolysis & gasification materials
EBPR	Enhanced Biological Phosphorus Removal - a waste water treatment configuration applied to activated sludge systems for the removal of phosphate based on the action of polyphosphate-accumulating organisms.
EC	European Commission
EU	European Union
FAO	Food and Agricultural Organisation of the United Nations
IBI	International Biochar Initiative – an international platform that groups stakeholders that have an interest in using pyrolysis & gasification materials as fertilising products
IED	Industrial Emissions Directive (Directive 2010/75/EU on industrial emissions (integrated pollution prevention and control).

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[27] More information on: http://europa.eu/rapid/press-release_IP-16-827_en.htm

JRC	Joint Research Centre of the European Commission
MAP	Mono-Ammonium Phosphate - a water-soluble mineral fertiliser that contains nitrogen and phosphorus
MBM	Meat and bone meal
NAC	Neutral Ammonium Citrate - a chemical extractant used as a proxy for plant-available phosphorus
NDA	Nutrient deficient areas; areas characterised by low nutrient levels in soils
NPK fertilisers	Mineral fertilisers that contains nitrogen (N), phosphorus (P) and potassium (K)
NSA	Nutrient surplus areas; areas characterised by high nutrient levels in soils
OECD	Organisation for Economic Co-operation and Development - an intergovernmental economic organisation founded to stimulate economic progress and world trade
P_2O_5	Phosphorus pentoxide (see section 13 for chemical conversion factor to phosphorus pentoxide)
PAH	Polycyclic Aromatic Hydrocarbons (also polyaromatic hydrocarbons or polynuclear aromatic hydrocarbons)
PCB	PolyChlorinated Biphenyl - an organic chlorine compound with the formula $C_{12}H_{10-x}Cl_x$
PCDD/F	PolyChlorinated DibenzoDioxins (PCDDs) and PolyChlorinated DibenzoFurans (PCDFs)
PFC	Product Function Category to which CE marked fertilising products shall belong in the proposed Revised Fertiliser Regulation[28] in line with their intended function (i.e. fertiliser, liming material, soil improver, growing medium, agronomic additive, plant biostimulant, fertilising product blend).
POP	Persistent Organic Pollutants - organic compounds that are resistant to environmental degradation through chemical, biological, and photolytic processes
RAE	Relative agronomic efficiency; the term expresses the agronomic efficiency in terms of plant yields and/or plant nutrient uptake for fertilisers derived from STRUBIAS materials relative to mined rock phosphate and processed P-fertilisers
REACH	Registration, Evaluation, Authorisation and restriction of Chemicals, Regulation (EC) No 1907/2006 of the European Parliament and of the Council. The Regulation was adopted to improve the protection of human health and the environment from the risks that can be posed by chemicals.
SSP	Single Super Phosphate - a water-soluble mineral phosphorus fertiliser that contains calcium dihydrogen phosphate and gypsum
STRUBIAS	STRUvite, BIochar and ASh-based materials. The acronym STRUBIAS, has been chosen as working title and does not necessarily reflect the final

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[28] More information on: http://europa.eu/rapid/press-release_IP-16-827_en.htm

materials	scope of any possible proposals for CMC categories
STRUBIAS sub-group	A technical working group that constitutes a sub-group of the Commission expert group on Fertilisers. The STRUBIAS sub-group participates in the process of sharing knowledge and provides non-binding expert advice to the European Commission on possible recovery rules for nutrients from eligible input materials into STRUBIAS materials.
TRL	Technology readiness level
TSP	Triple Super Phosphate - a water-soluble mineral phosphorus fertiliser, also known as calcium dihydrogen phosphate with the chemical formula $Ca(H_2PO_4)_2{\bullet}H_2O$
WHO	World Health Organization - a specialised agency of the United Nations that is concerned with international public health.

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13 Chemical conversion factors

The table below provides a number of conversion factors to quickly convert a fertiliser's nutrient content expressed as one chemical form into the content expressed as another chemical form.

For example, a material with a phosphorus content of 30%, expressed as P_2O_5, has a phosphorus content of 30% x 0.44 = 13.2% expressed as elemental phosphorus P.

P_2O_5	x	0.44	=	P
K_2O	x	0.83	=	K
Na_2O	x	0.74	=	Na
CaO	x	0.71	=	Ca
MgO	x	0.6	=	Mg
NH_3	x	0.82	=	N
SO_3	x	0.4	=	S
CaO	x	1.78	=	$CaCO_3$
P	x	2.29	=	P_2O_5
K	x	1.2	=	K_2O
Na	x	1.35	=	Na_2O
Ca	x	1.4	=	CaO
Mg	x	1.66	=	MgO
N	x	1.23	=	NH_3
S	x	2.5	=	SO_3
$CaCO_3$	x	0.56	=	CaO

ANNEXES

14 Potential input materials for fertilising products containing STRUBIAS materials

14.1 Introduction

Potential input materials for the production of STRUBIAS materials include waste and by-products within the meaning of Directive 2008/98/EC, animal by-products within the meaning of Regulation (EC) No 1069/2009, and biological materials. Each of these input materials have **specific properties,** including nutrient and contaminant contents, that impact upon their suitability to be used as input materials for a specific STRUBIAS production stream. This section aims at providing an overview of the characteristics for input materials that have a high potential for nutrient recovery with a specific emphasis on P-recovery.

The works of van Dijk et al. (2016) and Buckwell and Nadeau (2016) provide an excellent overview of the P-flows within the food and non-food production–consumption–waste chain for the EU-27 (Figure 28). Although the work provides an overview for streams within the year 2005, it provides a starting point for **designating material streams and core sectors that are dominantly responsible for the dissipation of P into the environment**. In section 7.3, best estimates on the potential for P-recovery from the streams and sectors of interest for the year 2030 through STRUBIAS will be provided.

It is indicated that the P-dissipation into the environment mainly takes place through losses from crop production, food processing, and human consumption (Figure 28). Animal production and non-food production are associated to lower losses (Figure 28) (van Dijk et al., 2016).

Losses from **crop production** mainly occur due to P accumulation in soils (924 kt P yr^{-1}), run-off and erosion (45 kt P yr^{-1}), and leaching and drainage to water bodies (40 kt P yr^{-1}). Additional losses from the agricultural sector are observed due to diffuse losses from stables (63 kt P yr^{-1}) (van Dijk et al., 2016).

Losses from **food processing** mainly originate from the slaughtering of animals and the subsequent removal of P-rich rest materials (e.g. animal bones) from the biogeochemical P cycles. This loss flow equals 294 kt P yr^{-1} (van Dijk et al., 2016). Other loss streams for the sector indicated by van Dijk et al. (2016) include food processing solid wastes (36 kt P yr^{-1}) and wastewaters (9 kt P yr^{-1}).

Losses from **human consumption** are dominated by materials that are lost from waste water treatment plants (van Dijk et al., 2016). About 227 kt P yr^{-1} ends up in communal sewage sludge, and an additional 74 kt P yr^{-1} is lost as effluents from urban and decentralised waste water treatment plants. Untreated and uncollected waste waters sum up a total of about 59 kt P yr^{-1}. Other significant sources of P-dissipation from human consumption include food waste from households, retail and food service (175 kt P yr^{-1}), pet excreta (69 kt P yr^{-1}) and paper and wood waste (30 kt P yr^{-1}) (van Dijk et al., 2016).

The losses from **non-food production**, as designated by van Dijk et al. (2016), relate to losses from forest-based industries (woodworking, furniture industry, pulp and paper industry). The total losses equal 77 kt P yr^{-1}, with wood industry waste being the dominant fraction (65 kt P yr^{-1}).

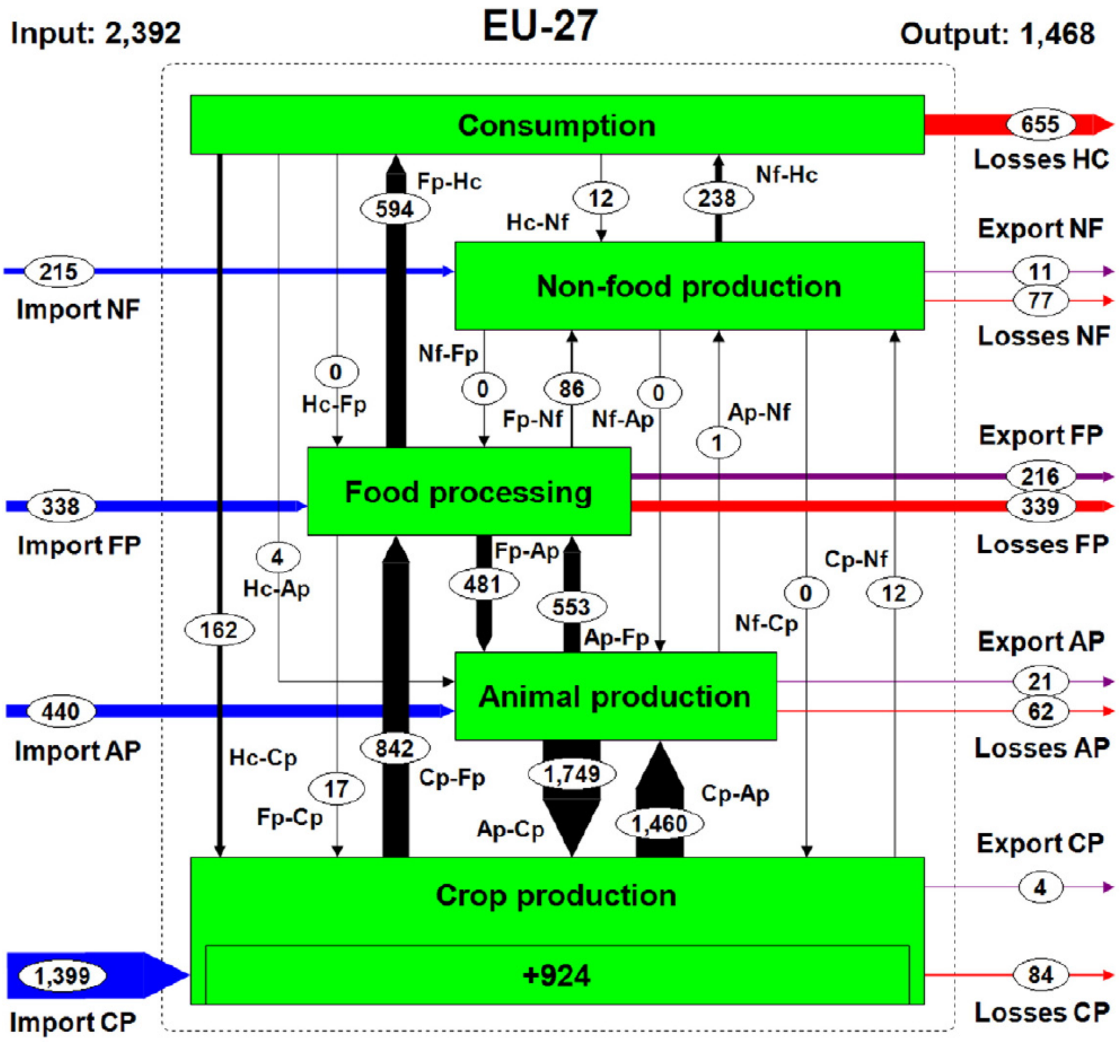


Figure 28. Phosphorus (P) use for the EU-27 in 2005 [kt P yr^{-1}] aggregated at the food and non-food production–consumption–waste chain; showing the imports (blue), exports (purple), losses (red) and internal upward/downward flows (black) for crop production (CP), animal production (AP), food processing (FP), non-food production (NF) and human consumption (HC) sectors (indicated with square blocks); the arrow thickness shows the relative flow sizes; the positive balance of +924 in CP represents annual net accumulation of P in agricultural soils in 2005 (adopted from van Dijk et al., 2016).

In the following sections, the characteristics of the designated potential input material streams by van Dijk (2016) and Buckwell and Nadeu (2016) will be reviewed in view of their potential to be used as an eligible input material for STRUBIAS production processes. Furthermore, the STRUBIAS sub-group pointed towards additional input materials that are suitable and already used for STRUBIAS production processes. Often, these input materials are used for a process aimed at the production of a different primary product (e.g. energy, clean water, steel) and STRUBIAS materials are produced or can be manufactured from residues from the process.

14.2 Crop residues

Cereals (283 Mt yr^{-1} harvested wet material, averaged over the period 2005-2012, expressed as wet matter) and **root crops** (173 Mt yr^{-1} harvested, averaged over the same period) are the most important types of crops in the EU-27, both **in terms of area cultivated (data not shown) and production amounts** (Table 20) (Eurostat, 2016). Crop harvest of oil seeds (26 Mt yr^{-1}) and rice (3 Mt yr^{-1}) make up a smaller contribution to the overall total of 350 Mt yr^{-1} for the EU (Table 20). Other crop types (e.g. other vegetables, nuts and non-food crops) were not considered in this analysis as they make up a very small contribution to the EU overall total crop production (Eurostat, 2016). Large differences exist between EU Member States, due to the climate conditions, specific soil condition and farming practices (data not shown; Eurostat, 2016).

The **residue-to-harvest ratio** varies widely across crop types, with the highest values observed for oil seeds (1.5 – 2.1), followed by cereals (1.0 – 1.6) (Table 20) (Energy Information Administration of the United States, 2001; Scarlat et al., 2010; Kremer, 2013). Root crops generally produce minor amounts of residues relative to the harvested crop biomass as the harvested tubers make up the dominant weight fraction of the plant biomass (residue-harvest ratio varying from 0.2-0.4) (Table 20). **Root crops and oil seeds have significantly higher N and P concentrations in their residues than cereals**, although large differences in nutrient concentrations were observed among the different cereal crops (Table 20) (Kremer, 2013; Plants Database, 2016). Grain maize has, for instance, a nutrient content that is about 3 to 4 times higher than that of wheat straw.

Regarding nutrient content, **cereals produce about half of the crop residue nutrients** present (1353 kt N yr^{-1}, 279 kt P yr^{-1}), with the **other half split roughly equally between root crops** (676 kt N yr^{-1}, 126 kt P yr^{-1}) and **oil seeds** (743 kt N yr^{-1}, 150 kt P yr^{-1}). Five crop types (wheat and spelt, grain maize and corn-cob mixtures, sugar beet, sunflower seeds and rape and turnip rape seeds), contribute for 71%-73% of the nutrients generated as crop residues (Table 20).

Table 20: Production, characteristics and nutrient contents of agricultural crop residues in the EU-27, averaged for the period 2005-2012 (Sources: Energy Information Administration of the United States (2001); Eurostat (2016); Kremer (2013); Plants Database (2016); Scarlat et al. (2010))

crop	harvest	residue-to-harvest ratio	residue dry matter content	residue nutrient concentration		residue nutrient content	
				N	P	N	P
	(Mt yr-1)	(-)	(Mt yr-1)[1]	(%)	(%)	(kt yr-1)	(kt yr-1)
cereal	283		292			1353	279
wheat and spelt	136	1.3	150	0.28	0.05	419	73
barley	57	1.2	58	0.43	0.09	248	50
grain maize and corn-cob-mix	61	1.0	47	0.81	0.20	384	96
oats and spring cereal mixtures	12	1.3	14	0.70	0.14	101	20
rye and winter cereal mixtures	8	1.6	12	0.48	0.11	57	13
triticale	10	1.3	11	1.30	0.24	144	27
rice	3	1.4	3	0.67	0.13	23	5
root crops	173		40			676	126
potatoes	59	0.4	20	1.10	0.18	224	36
sugar beet	115	0.2	20	2.28	0.45	452	89
oil seeds	26		38			743	150
sunflower seeds	7	2.1	12	2.84	0.58	351	71
rape and turnip rape seeds	19	1.5	26	1.53	0.31	392	78
overall total	350		374			2795	559

[1] calculated based on residue-to-harvest ratio and residue dry matter content

A distinction, however, has to be made between residues remaining in the field and those generated after harvesting. The majority of the crop residue is not collected and removed, but **ploughed back into soil as its collection is too expensive to be profitable.** Additionally, agricultural residues play an important role in **maintaining or improving soil characteristics**, protecting the soil from erosion, maintaining or increasing soil organic matter, maintaining nutrients in the soil and improving water retention (Nelson, 2002 in Scarlat, 2010). Therefore, it is recommended to leave a significant share of the crop residues on the field for agricultural sustainability, dependent on crop type, farming practices, site conditions, and climate. The current best practice of incorporation for the EU is of one-third of total residues (Joint Research Centre, 2009), although other studies have reported even higher values of 40%-50% (Scarlat et al., 2010).

Crop residues are only collected if there is an economic potential, for instance, for their use as animal feed, bedding material or biofuel. The economic viability increases if fertilising materials from crop residues are produced as part of a cascading approach that relies on the residues of the primary process of recovery of valuable organic substances from crop residues (e.g. lactic acid, citric acid, ethanol, caffeine, yeast production, biogas production, etc.).

At present, harvested residues are used for many often **onsite-specific purposes**: food, fodder, feedstock, fibre, and further use such as compost production. Some amount of crop residues is also collected for mushroom cultivation and various horticultural uses. Scarlat et al. (2010) estimated that about 26 Mt yr^{-1} of residues are used in animal husbandry and another 1.6 Mt yr^{-1} of residue for mushroom production over the whole of the EU. The uptake of crop residues by the animal husbandry sector largely depends on the farm ratio of livestock to crop, for which higher values were documented for countries with high relative livestock proportions (e.g. for the UK, Searle and Malins, 2013). Much of the residue consumption for livestock occurs on site, i.e., the same farmer who harvests cereals and collects the straw feeds it to his or her livestock. Wheat and barley straw are also traded in Europe for use in the livestock sector. Some residues are sold to other parties, and this amount is easier to track. Studies have estimated off-farm residue use to amount to 5% - 6% (Kim and Dale, 2004).

Crop residues are often cited and explored for **their energy recovery potential**, but at present, crop residues are only used to a very small extent throughout the EU. Available EU-level data indicate that dedicated energy cropping for biofuels and electricity and heat generation covered approximately 5.5 million hectares of agricultural land in 2008 (ETC/SIA, 2013), or about 3% of the EU utilised agricultural area. Practically all of this land was used for dedicated biofuel cropping (bioethanol and biodiesel). The realistic potential derived from the technical-sustainable potential for agricultural crop residues to contribute to renewable energy production has been estimated at 75 million tonnes per year in the EU-28, with a dominant contribution of the cereals wheat (26%), maize (21%) and barley (16%) (Iqbal et al., 2016). Based on the P content as given in Table 20 and taking into consideration the recommended crop extraction rates, **the total P content in the crop residues that could be used in nutrient recovery processes is estimated at around 110 kt P yr^{-1}.** Nutrient recovery from crop residues used for renewable energy production could potentially be a value-adding life stage or end-of-life solution for materials subject to anaerobic digestion for biogas production, or other thermochemical energy recovery processes. Such pathways are compatible with the production of **precipitated phosphate salts & derivates** (after anaerobic digestion, although concerns exist due to the low phosphate content of anaerobically digested crop residues), **thermal oxidation materials & derivates** (Insam and Knapp, 2011), and **pyrolysis & gasification materials** (EBC, 2012).

14.3 Manure

The **EU farm livestock population** consists of 153 million pigs, 88 million cattle, 1.3 billion poultry, 83 million sheep and 10 million goats (FAOSTAT-Commodity Balances Livestock, 2012). Together, the **livestock excretes around 1400 Mt of manure** (Table 21) (Gendebien et al., 2001; Foget et al., 2011; Agrotechnology Atlas, 2016; Buckwell and Nadeu, 2016). Cattle manure (1092 Mt) represents the dominant manure fraction, with amounts that are about one order of magnitude greater than for pig (177 Mt) and poultry (112 Mt) (Foget et al., 2011) (Table 21). The manure produced from other livestock groups is mostly deposited directly on land by grazing animals. Production and characteristics of manure on a farm are largely **affected by species and growth stages of animals, feed ratios, manure collection and handling methods, and the amount of water added into the manure collection systems** (Agrotechnology Atlas, 2016).

Animal manure contains complex organic compounds originated from the undigested and wasted feed and veterinary products as well as simple organic and inorganic compounds produced in the gastrointestinal tract of animals. Hence, **manure slurry** is a mix of faeces and urine from livestock, bedding material with small structure like sawdust or chopped straw, washing water, water spill, etc. and originating from stables with whole or partly slotted floors (Bicudo, 2009). **Solid-liquid separation**, possibly after anaerobic digestion, is often used as a treatment method from improving manure handling properties and producing manure solids for energy generation, compost production and animal feeding. Another goal for solid liquid separation is to produce fertiliser products with different ratios of nitrogen to P and N to potassium (K) serving a better tuning with crops requirements for nutrients. Manure is removed from the livestock stables on a frequent (for instance daily) basis, and placed in a manure pad with drains, enabling to collect liquid fractions such as urine, silage effluents, process water and alike, in separate stores, and vice versa with the solid fractions. A broad variety of solid-liquid separation techniques is, however, available (Hjorth et al., 2010). Effective solid-liquid separation that is capable of removing a substantial amount of organic solids from fresh liquid or slurries can potentially offer the benefits of production of nutrient-rich organic solids, odour reduction in the subsequent liquid manure storages, and improvement in the economics of **subsequent liquid manure treatment processes**. The separated **manure solids** can be utilized on farms near animal operations or can be exported to other areas at significant economic and environmental costs as fertiliser and soil conditioning products (Agrotechnology Atlas, 2016). Alternatively, livestock is kept in stables where on a bed of long straw or similar material, up to 1 metre thick. The bed, also referred to as **deep litter**, is only removed with intervals of up to one year, when the livestock is removed from the stable for slaughter or grazing (Agrotechnology Atlas, 2016).

In spite of the vast nutrient amounts present in manure, these are present in highly diluted form as manure has an **average dry matter content of only 19%** (Table 21). The dry matter content is lowest for pig manure, with values as low as 5% for the largest share of the pig manure (pig slurry, Table 21). The **nutrient content of manure stocks varies broadly with the origin of the manure (cattle, pig, and poultry) and the type and extent of separation** (Table 21). Across the different origins of manure, poultry and pig slurries have the highest N and P concentrations with values of 5.8% - 8.3% and 2.2% - 2.3%, for N and P, respectively (Table 21). The N/P ratios of most manure types vary between 2 and 5, with the exception of liquid cattle manure that has an N/P ratio of about 12 (Table 21). All manure types have generally high organic matter contents (range 57% - 82%) (Table 21).

Cattle is the dominant manure fraction when data are expressed on a nutrient content basis. **Cattle manure contributes for 61% and 54% of the N and P present in the total livestock manure**, respectively (Table 21). **Poultry** manure is another significant nutrient stream, especially for P (36% of the total manure, up to 883 kt P yr^{-1}) and to a slightly lesser extent for N (30% of the total manure) (Table 21). **Pig** manure contributes for 9% (for N) and 10% (for P) to the total nutrients presents in manure (Table 21). Poultry and pig are often kept inside year-round, for which reasons their manure can be easily collected.

Table 21: Amount of manure in the EU-27 and its composition (sources: Agrotechnology Atlas (2016); Foget et al. (2011); Gendebien et al. (2001))

	Amount	Dry matter	Organic matter	N/P	K			N			P		
					concen-tration	content	relative contribution	concen-tration	content	relative contribution	concen-tration	content	relative contribution
	(Mt yr^{-1})	(%)	(% of dry matter)	(-)	(% of DM)	(kt K yr^{-1})	(%)	(% of DM)	(kt N yr^{-1})	(%)	(% of DM)	(kt P yr^{-1})	(%)
Pig slurry	148.6	5	69	3.7	4.6	373	4.7	8.3	673	7.2	2.2	181	7.3
Source separated pig manure							0.0			0.0			0.0
Solid	14.2	24	80	2.4	2.0	71	0.9	3.2	113	1.2	1.4	48	1.9
Liquid	8.8	2	n.a.	5.4	9.1	14	0.2	17.1	27	0.3	3.2	5	0.2
Pig deep litter	5.3	28	75	2.5	4.0	59	0.7	2.3	35	0.4	0.9	14	0.6
Total pig	176.9	7				517	6.5		847	9.1		248	10.0
Cattle slurry	447.8	9	66	4.6	4.7	1899	23.9	4.5	1818	19.6	1.0	399	16.1
Source separated cattle manure													
Solid	294.9	22	64	3.3	2.1	1371	17.2	2.4	1557	16.8	0.7	475	19.2
Liquid	54.6	3	57	12.5	29.4	447	5.6	12.2	185	2.0	1.0	15	0.6
Cattle deep litter	294.9	25	77	4.7	3.2	2397	30.2	2.9	2135	23.0	0.6	454	18.4
Total cattle	1092.1	17				6114	76.9		5694	61.3		1342	54.3
Poultry slurry	3.4	14	82	3.0	2.6	13	0.2	6.8	33	0.4	2.3	11	0.5
Poultry deep litter	109.5	57	74	3.1	2.1	1307	16.4	4.4	2713	29.2	1.4	872	35.3
Total poultry	112.9	55				1319	16.6		2746	29.6		883	35.7
Overall total	1381.9	19	69	4.4		7950			9287			2473	

It should be noted that the numbers for P presented in Table 2 differ from the ones presented in van Dijk et al. (2016). In their study, a total manure flux of 1.75 Mt P yr^{-1} was observed for the reference year 2005, in contrast to the number of 2.47 Mt P yr^{-1} as presented in Table 21. The reasons for this discrepancy relate to the different reference years that are used in the studies, and the uncertainty in average values for the nutrient concentrations and dry matter contents of the diverse manure fractions. Hence, estimating manure nutrient inputs is associated to a significant degree of uncertainty, and the absolute numbers given should be interpreted with the necessary caution.

The main fate of manure in the EU-28 is currently land application. At present, more than 90% of the manure produced in the EU is returned to agricultural land either through the spreading of the collected manure or directly by outside grazing activities (Buckwell and Nadeu, 2016). Only 7.8% of the produced manure, or 139 kt P, was processed in the year 2010 (Foget et al., 2011).

Landspreading of manure involves a risk for the accumulation of potentially toxic elements, pathogen recycling, and odour nuisance (Gendebien et al., 2001). Manure may contain significant amounts of metals/metalloids due to the use of mineral supplements and veterinary products. This is particularly true for pig slurry, which can contain up to 600 mg kg^{-1} dry matter of Cu, and up to 900 mg kg^{-1} dry matter of Zn. Also, cattle slurries contain high amounts of Zn (up to 750 mg kg^{-1} dry matter) (Gendebien et al., 2001). Considering the large volumes of manure applied, metals/metalloids may accumulate as a result of long-term agricultural use. Accumulation of metals/metalloids could not only affect the soil fertility, but also promote metal migration through leaching and runoff. Additionally, there is also growing concern regarding the environmental fate and potential impacts of the veterinary pharmaceuticals on human and ecosystem health as animal feeding pharmaceuticals are excreted into manure either as parent compounds or as bioactive metabolites (Song et al., 2010). Finally, animal manures contain pathogenic elements in variable quantitites depending on the animal health. Manures applied without treatment and restriction on the application to land represent a risk for possible re-infection of resident animal population and the contamination of meat, dairy products, vegetables and water resources. There have been reports on cases of drinking water supplies contaminated by cattle slurry resulting in outbreaks of human diseases (Gendebien et al., 2001; Spellman and Whiting, 2007). The sources of ammonia emissions and odour nuisance from livestock production are from animal housing, waste handling, storage and landspreading. The storage of pig and poultry manure may cause odour problems if wastes turn anaerobic and give rise to strong odour when the crust is broken.

While nutrient recovery has **an enormous, positive role in well-adjusted agricultural systems**, much of the recycled manure is often **applied according to non-sustainable practices** due to spatial constraints associated to high transport costs of the large-volume material and unbalanced nutrient supply stoichiometry. This may lead to P accumulation in soil and water eutrophication after the leaching of nutrients from the soil.

In the legal sense, the animal by-products regulation (EC/1069/2009) classifies manure as an animal by-product of category 2. According to this regulation, **manure can be applied on land without further processing.** Possible manure **use restrictions** are, however, defined in other EU legislation.

The **Water Framework Directive** (WFD, 2000/60/EC) establishes an integrated and co-ordinated framework for the sustainable management of water, including prevention of deterioration of water bodies (lakes, rivers, coastal and transitional, groundwater), promotion of sustainable water use, and ensuring "enhanced protection and improvement of the aquatic environment". The **Nitrates Directives** (91/676/EEC) indirectly limits the return of P to land by limiting the amount of N in manure that can be applied on land. However, the stoichiometric N/P ratios documented for soil microbes and plants (around 6 - 8; Cleveland and Liptzin, 2007) are higher than the N/P ratios of most types of manure (on average 4.4, Table 21). This indicates that **even manure applied to land in line with the Nitrates Directive contributes significantly to the observed P accumulations** in agricultural ecosystems that receive high manure loads (Figure 1; van Dijk et al., 2016). Member states (Belgium (Flanders, Wallonia), Denmark, the Netherlands, UK, Northern Ireland, Ireland) who have or had (Germany, Austria) an derogation for the use of N from animal manures (Nitrates directive) have to meet requirements of sound fertilisation practices which already leads to a more balanced nutrient management. In the Netherlands, for instance, manure processing is obligatory in case a farm has insufficient land to apply animal manure according the regulatory application standard. It is estimated that in 2015 about 8.9 kT P yr^{-1} from animal manure was processed in the Netherlands (BMA, 2015), with an additional 4.9 kT P yr^{-1} being planned. **Integrated pollution control**, as a principle of environmental protection and management, aims to minimize the overall environmental impact of human activities by taking into account pollution of air, water, land and the human environment, and identifying the action that causes on balance the least damage. As a legal system, integrated pollution control has been adopted by the European Union and, with an extended remit, been put in force as integrated pollution prevention and control (IPPC). IPPC covers intensive animal rearing for farms with a capacity of greater than 40,000 animal places for poultry, 2000 for fattening pigs and 750 for sows. New farms and those with extensive modifications have to comply immediately whereas existing farms had to do so by 2007. Following inter-institutional negotiations, the Directive on industrial emissions 2010/75/EU (IED) was adopted on 24 November 2010. The IED repeals the IPPC Directive and the sectoral directives as of 7 January 2014. The IED requires **'best available techniques'** to be applied in the operation of an installation. This will include measures for the management of manure/slurry within the installation (e.g. its removal from animal houses) and methods for its storage so as to prevent or minimise environmental impacts.

Eurostat monitors the risk of P pollution from agriculture based on the indicator "**gross P balance**", indicative for the potential surplus of P. The P balance lists all inputs and outputs into and out of the soil and calculates the gross P surplus as the difference between total inputs and total outputs. The gross P surplus per ha is derived by dividing the total gross phosphorus surplus by the reference area.

The inputs of the phosphorus balance are:

- Fertilisers, which consist of:
    - inorganic fertilisers;
    - organic fertilisers (excluding manure).
- Gross manure input, which is calculated from:
    - manure production (phosphorus excretion);
    - manure withdrawals (manure export, manure processed as industrial waste, non-agricultural use of manure, other withdrawals);
    - change in manure stocks;
    - manure import.
- Other phosphorus inputs, which consist of:
    - seeds and planting material;
    - atmospheric deposition.

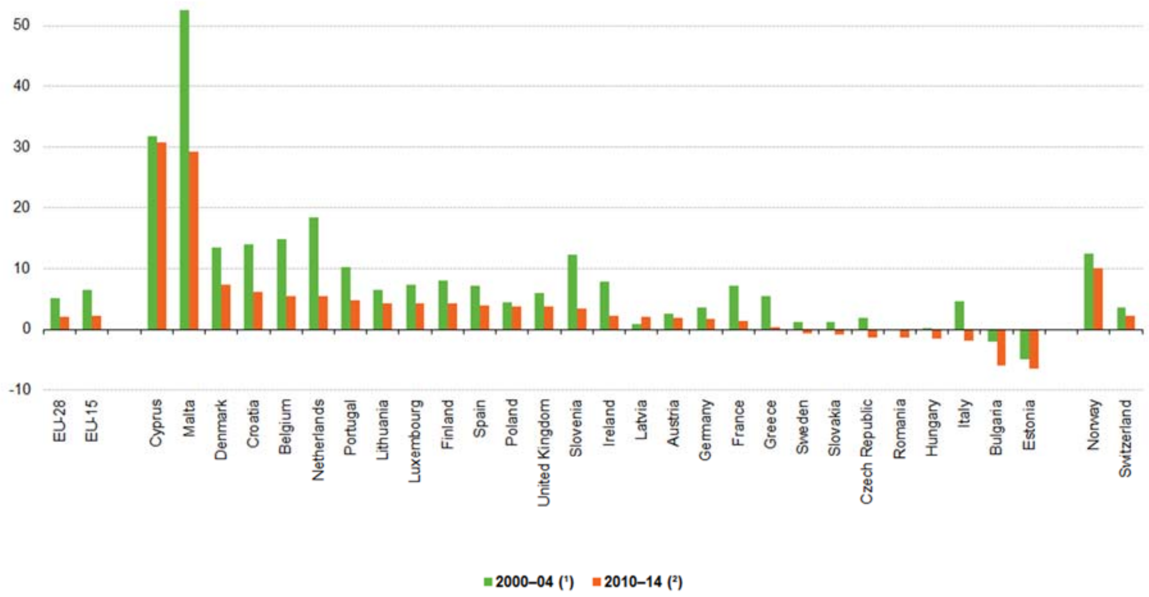
The outputs of the gross phosphorus balance are:

- Total removal of P with the harvest of crops (cereals, dried pulses, root crops, industrial crops, vegetables, fruit, ornamental plants, other harvested crops).
- Total removal of P with the harvest and grazing of fodder (fodder from arable land, permanent and temporary pasture consumption).
- Crop residues and soil of (root) crops removed from the field.

The indicator provides an indication of the potential surplus P on agricultural land (kg P per ha per year). The quality and accuracy of the estimated gross P surplus per ha depends on the quality and accuracy of underlying data and coefficients used. As methodologies (especially with regards to the coefficients) and data sources used in countries vary, the balances are only consistent within a country across time. The gross P balances are not consistent across countries implying that data cannot be compared between countries. It should be noted that data on **manure withdrawals, manure stocks and imports** were not **available in most countries**. Available data on manure withdrawals for non-agricultural use show that this input is significant (more than 5% of total manure input) in some countries (Belgium, the Czech Republic, the Netherlands), while non-significant in other countries (the United Kingdom, Switzerland).

The gross P balance can only indicate the **potential risk** to the environment while the **actual risk for water eutrophication depends on many factors including climate conditions, soil type and soil characteristics, management practices such as drainage, tillage, irrigation,** etc. The risk of P pollution is only partially determined by the P balance of a particular year. It is often more strongly determined by the cumulative P balance of the past.

The **potential P surplus for the EU-28 decreased from an estimated average of 5.2 kg P per ha per year in the period 2000-2004 to 1.9 kg P per ha per year in the period 2010-2014**. For the EU-15 the potential P surplus dropped from on average 6.4 kg P per ha per year in 2000-2004 to an average of 2.3 kg P per ha per year in 2010-2014. The average potential P surplus per ha in 2010-2014 was highest in the Mediterranean islands Cyprus and Malta, above the EU average in Norway, Denmark, Croatia, Belgium, the Netherlands, Portugal, Lithuania, Luxembourg, Finland, Spain, Poland, the United Kingdom, Slovenia, Switzerland, Ireland and Latvia while the balance was negative for Sweden, Slovakia, the Czech Republic, Romania, Hungary, Italy, Bulgaria and Estonia (Figure 29). In all Member States, with the exception of Latvia, the potential P surplus between 2010 and 2014 was lower than between 2000 and 2004.



Note: Eurostat estimates for EU-28, EU-15, Belgium, Bulgaria, Denmark, Greece, Spain, Croatia, Italy, Cyprus, Latvia, Lithuania, Luxembourg, Malta, Austria, Romania and Slovakia.
(1) Estonia: 2004
(2) Germany, Ireland, Sweden and Switzerland: 2010-13

Figure 29: Gross phosphorus balance, averages 2000–04 and 2010–14 (Source: Eurostat - Agri-environmental indicator - risk of pollution by phosphorus)

Nonetheless, **there is still a clear scope to increase the efficiency of nutrient recycling in the agricultural sector. One option is through manure treatment options** that aim at producing a safer, lower volume, and more targeted fertiliser that better matches crop needs. For the year 2010, **7.8% of the manure (108 Mt, 556 kt N, 139 kt P) was being collected for treatment** in the EU (Foget et al., 2011; Flotats et al., 2013). Distributed on livestock manure treatments, pig slurry is a major focal area for treatment due to the high transport costs for the highly diluted manure and the small surface area to dispose the slurry produced (Foget et al., 2011). Two major routes are applied individually or combined to process manure with the objective to change the physical and chemical manure properties, to recover energy or to prepare for nutrient removal from the stream (Foget et al., 2011).

Separation via mechanical, chemical or other technologies is applied to treat 49 Mt of livestock manure, while anaerobic digestion is applied for 88 Mt of material (Buckwell and Nadeu, 2016). In almost all types of separation, organic and inorganic coagulants and flocculants are applied to achieve a good separation between the solid and liquid phase, resulting in the precipitation of suspended solids and the concentration of the phosphates in the solid fraction. The use of coagulants and flocculants results in an increase of the costs of the mechanical separation process, and some flocculants might have adverse environmental effects, e.g. on soil and aquatic organisms (Buczek et al., 2017). Their use also has a strong effect on the further treatment potential of the solid and liquid fraction (Schoumans et al., 2010), although new technologies are emerging. Usual coagulants and flocculants are polyelectrolytes, aluminum and iron sulphate, aluminum and iron chloride, calcium oxides and calcium hydroxides, and also magnesium oxide and magnesium hydroxides. In case of targeting P-recovery in agriculture, the use of metal-containing salts might potentially impact upon the adeptness for P-recycling.

Recovery of biogas during **anaerobic digestion** is advantageous as it can be used for electric energy generation or for the heating and drying during the further processing of manure. Additionally, anaerobic digestion increases the dewatering properties of the slurry as more components end up in the solid fraction following separation of the slurry, and the drying of the solid fraction proceeds more rapidly (Schoumans et al., 2010). The solid fraction can then be dried before pelletising or following incineration, or alternatively, composting is used. For 0.7% of the manure production in the EU, the liquid fraction is further processed, mostly through biological treatment via conventional nitrification – denitrification systems and concentration through evaporation or filtration methods to produce a mineral concentrate (Foget et al., 2011; Buckwell and Nadeu, 2016). In the Netherlands, reverse osmosis techniques are used to produce NK fertilisers from liquid fractions of separation of animal manure or digestates. Substantial variations in the extent of manure processing and the different manure processing techniques are observed across EU member states (Foget et al., 2011).

Manure and stable livestock slurries are used as inputs for operational STRUBIAS processes. Stichting Mestverwerking Gelderland produces **K-struvite** from veal manure in the Netherlands. Also the EU funded BioEcoSim project aims at producing a mixture of precipitated phosphate salts & derivates (from pig manure). Different companies in the Netherlands and the UK (e.g. BMC Moerdijk, Fibrophos, etc.) produce (mono- or co-incineration) **poultry litter ashes** from a combination of slaughtered animals and poultry beds. Finally, Hitachi-Zosen is evaluating the possibility of establishing **pig and poultry manure pyrolysis facilities** in Europe based on experience obtained from their pilot plant in Japan. Also, the BioEcoSim project and their partners are evaluating suchlike facilities in Western Europe.

14.4 Animal by-products other than manure

14.4.1 Food waste from households, retail and food services

Food waste in the household sector involves **waste associated to meal preparation, leftovers, and purchased food not used in time**. Causes for this waste are diverse and relate to a lack of awareness and cooking skills, personal preferences, improper planning, labelling issues, storage and inappropriate portion sizes. In **wholesale/retail**, waste accumulates as a result of damage and expiry of products or surplus due to supply chain inefficiencies, improper stock management and deficient storage. Finally, additional aspects that cause waste from the **food services** include the single portion size, overstocking and meeting specific customer wishes (e.g. school children, etc.) (Bio Intelligence Service - Umweltbundesamt - AEA, 2010). Legally, waste from households, retail and food service waste containing products of animal origin is **category 3 animal by-product material**.

The amount of food waste generated from households, retail and food services is estimated at **62 Mt fresh material yr^{-1}, of which about three quarters (73%) is household waste**, **17% is food services and 8% is retail** (Stenmark et al., 2016). The **nutrient content** of food waste is largely dependent on the exact composition (fraction animal and plant origin), but generally ranges from **1.6% - 2.3% for N and 0.26 - 0.54% P**. The dry matter content varies in the range 15% - 29% of the fresh weight (Brink, 1993; Widen, 1993). Hence, it is estimated that food waste from households, retail and food service could contain around **25-100 kt P per year**.

The current fate of the separately collected food waste is highly Member State specific. Currently, there is **no direct obligation at the EU level to recycle biodegradable waste** resulting in great differences across Member States. The Landfill Directive (1999/31/EC) obliges Member States, however, to reduce the amount of biodegradable municipal waste that they landfill to 35% of 1995 levels by 2016 (for some countries by 2020). On 2 July 2014, the European Commission adopted a legislative proposal to review waste-related targets in the Landfill Directive as well as recycling and other waste-related targets in Directive 2008/98/EC on waste and Directive 94/62/EC on Packaging and Packaging Waste. The proposal aims at phasing out landfilling by 2025 for recyclable waste (including plastics, paper, metals, glass and bio-waste) in non-hazardous waste landfills, corresponding to a maximum landfilling rate of 25%.

Biodegradable waste **is a suitable input material for nutrient recycling options, such as composting and anaerobic digestion,** thanks to the low heavy metal content and high bio-availability of the nutrients. A critical aspect is, however, to what extent the separate collection and processing of food waste can be achieved.

In the **legal sense**, food waste is treated in the same way as normal waste that is non-hazardous if and only if it does not exhibit any properties that may render it hazardous. **Products of animal origin, or foodstuffs containing products of animal origin, and catering waste are, however, classified as category 3 Animal By-products**. Stringent controls are applied to its transport, handling and storage, treatment and disposal through Animal By-Products Regulation (EC) No 1069/2009**. Therefore, a share of the food waste is processed together with slaughter residues by the rendering industry** (see paragraph 14.4.4).

14.4.2 Materials from the fish industry

In 2012 the fish processing sector in the EU comprised approximately 3,500 firms with fish processing as their main activity (Doring and Borrello, 2014). Italy possessed the largest fish processing industry in 2012 in terms of number of firms (16% of the total) and the United Kingdom in terms of people employed (16% of the total), followed by Spain and France (Doring and Borrello, 2014). This sector includes activities such as the processing of white or pelagic fish; fatty fish; shellfish, i.e. crustaceans and molluscs, and fresh water fish. Fish processing is very widespread and varied. Many species of fish are mass processed, including cod, tuna, herring, mackerel, pollock, hake, haddock, salmon, anchovy and pilchards (European Commission, 2006b). Post-harvest loss occurs during pre-processing, processing, storage and transportation of fishery and aquaculture products. Post-harvest fish losses are, for instance, caused by fish scrap generation during fileting, curing and smoking processes and discarding of bycatch at sea because fish is too small or not valuable enough to bring to land for sale. At the global scale, up to **70% of total processed raw fish (on weight basis) ends up as solid waste in processing plants** (UNEP, 2000).

In the EU-28, about 5000 kt of fresh fish is processed on a yearly basis (Eurostat, 2017). Moreover, the EU-28 had a stable output of aquaculture products during the period 2004–2014, with a production quantity fluctuating around 1200 – 1300 kt live weight. Assuming a P content of 0.43% (Hjerne and Hansson, 2002), **the P content of fish residues from catches and aquaculture for the EU-28 could amount of up to 27 kt P yr^{-1}.** Moreover, fish excreta and non-digested feed from land-based aquaculture also form a P-source that can potentially be recovered as STRUBIAS materials. Fish residues generated during the processing of raw fish is a great source of minerals, proteins and fat, but if discarded, they can represent a major P loss. Treated fish waste has found many applications among which the most important are animal feed, biodiesel/biogas, dietetic products (chitosan), natural pigments (after extraction) and cosmetics (collagen). Residues from the sector are thus majorly used for the production of value-added products, and off-shore P losses from the sector are estimated relatively small (Hamilton et al., 2016).

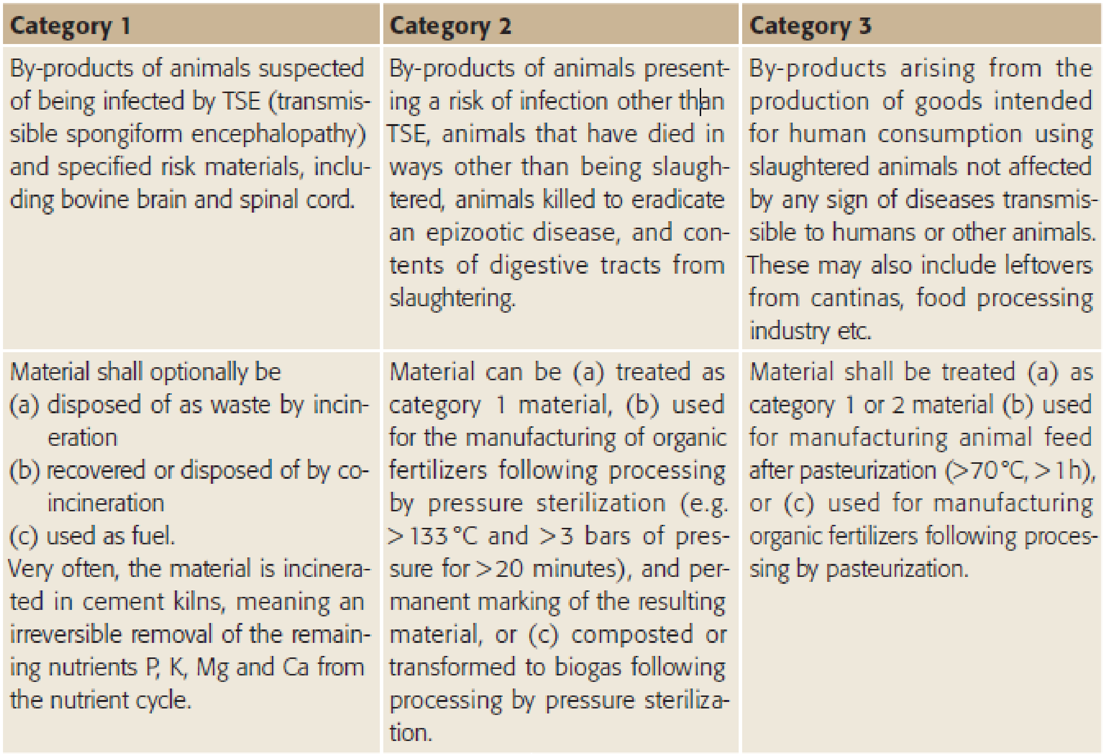
Legally, residues from the fish industry are classified **as animal by-products and derived products**. The materials show similarity to animal by-products from the meat and dairy industry, for which reason P-recovery in the form of recovered phosphates, ashes and pyrolysis & gasification materials is in theory possible. No industrial STRUBIAS processes of high TRL level have so far been documented by the STRUBIAS sub-group, but similar recovery processes as documented for materials from the meat industry could potentially apply (see paragraph 14.4.3 and 14.4.4).

14.4.3 Materials from the meat industries

The cattle and poultry industries are the largest source of animal food industry waste (Jayathilakan et al., 2012; EFPRA, 2017), with numbers largely exceeding those for fish scrap (Hamilton et al., 2016). In recent years there has been a change in the terminology used to describe outputs from slaughterhouses. The term “by-product” is being used increasingly; in this document, the term "**slaughter residues**" will be used in order to make a clear distinction with other animal by-products, such as manure. Slaughter residues produced in abattoirs consist of the portion of **slaughtered animals that cannot be sold as meat or used in meat products**. The consumer has a preference for lean meat, and only limited amounts of organ meats such as brain, kidney, sweetbread, tongue, etc. are consumed. As a result, basically the following residues become available in the slaughter process: (1) edible products such as blood and liver; (2) inedible products such as hair, bones, feathers; (3) manure, contents of rumen and intestines, (4) wastewater, and (5) fat (recovered from the wastewater by means of fat-separators). A complete overview is given in the Reference Document on Best Available Techniques in the Slaughterhouses and Animal By-products Industries (European Commission, 2005).

Slaughter residues are classified with other **animal by-products according to Regulation (EC) 1069/2009**. Slaughter residues are animal derived, and can contain different bacterial, viral, prion and parasitic **pathogens** and cannot be discharged into the environment without proper treatment. Therefore, the use of unprocessed slaughter residues for animal feed has been banned in the EU since 2000 due to fear of Bovine Spongiform Encephalopathy (BSE). Determinate category-specific product **treatment options** should now be undertaken prior to further utilization (Table 22). The rendering industry handles slaughter residues, fallen stock taken from farms, catering waste and unsold animal products that have the potential to become a health risk (EFPRA, 2017). Through applying the prescribed procedures, rendering makes the material safe and **suitable for reuse** as outlined in the animal by-products Regulation (EC No 1069/2009). The material as obtained after the application of heat and pressure to sterilise and stabilise animal material is rich in fat and protein, for which reason it is suitable for a number of application as outlined in Figure 30. At present, specific animal by-products of category 2 and 3 can be processed and put to further use as feed, organic fertilisers and soil improvers (Article 32 of EC Regulation (EC) 1069/2009 on animal by-products).

Table 22: Classification of substrates according to Regulation (EC) 1069/2009, and their respective main treatment routes (adopted from Moller, 2015)



Category 1	Category 2	Category 3
By-products of animals suspected of being infected by TSE (transmissible spongiform encephalopathy) and specified risk materials, including bovine brain and spinal cord.	By-products of animals presenting a risk of infection other than TSE, animals that have died in ways other than being slaughtered, animals killed to eradicate an epizootic disease, and contents of digestive tracts from slaughtering.	By-products arising from the production of goods intended for human consumption using slaughtered animals not affected by any sign of diseases transmissible to humans or other animals. These may also include leftovers from cantinas, food processing industry etc.
Material shall optionally be (a) disposed of as waste by incineration (b) recovered or disposed of by co-incineration (c) used as fuel. Very often, the material is incinerated in cement kilns, meaning an irreversible removal of the remaining nutrients P, K, Mg and Ca from the nutrient cycle.	Material can be (a) treated as category 1 material, (b) used for the manufacturing of organic fertilizers following processing by pressure sterilization (e.g. >133 °C and >3 bars of pressure for >20 minutes), and permanent marking of the resulting material, or (c) composted or transformed to biogas following processing by pressure sterilization.	Material shall be treated (a) as category 1 or 2 material (b) used for manufacturing animal feed after pasteurization (>70 °C, >1 h), or (c) used for manufacturing organic fertilizers following processing by pasteurization.

Poultry (23 Mt yr^{-1}) and **pig** (21 Mt yr^{-1}) constitute the greatest fractions of animal carcasses expressed on a weight basis, followed by bovine meat (8 Mt yr^{-1}) (Table 23) (Eurostat, 2016). Sheep and goat meat do not make up a substantial amount of the slaughtered animals. The waste fraction is highest for cattle (0.42), than for pig (0.34) and poultry (0.25) (Table 23). **A total amount of >17 Mt of slaughter residues** is generated with poultry meat (55%) having the highest relative contribution, followed by pig (25%) and bovine meat (20%). The data of Table 5 are generally in agreement with the data as presented by Van Dijk et al. (2016), and indicate a total P flow of about 281 kT P yr^{-1} for the sector. Abattoir wastes are characterized by **very high P contents**. The mean P content of bone for bovine and poultry bone is about 10.5% on a dry weight basis (Beighle et al., 1994; Hemme et al., 2005), and P contents for blood and offal may reach up to 4.6% and 1.5%, respectively (Gendebien et al., 2001).

Table 23: Overview of the slaughtered animals and the amounts of slaughter residues generated in EU-28 for the year 2005

	absolute weight of slaughtered animals[a,b] (kt yr^{-1})	Inedible fraction[c,d] (-)	slaughter refuse (kt yr^{-1})	relative contribution (%)
bovine meat	8136	0.42	3417	25
calve	946			
adult cattle	6819			
pigmeat	21781	0.34	7405	53
sheep and goat meat	809	0.48	388	3
poultry meat	10797	0.25	2699	19
overall sum	49289		13910	

[a] based on Eurostat values for slaughtering in slaughterhouses for the year 2005 (cattle, pig and poultry) and 2015 (sheep and goat)

[b] data for poultry meat are underestimated because data for specific EU Member States are referred to as confidential in the Eurostat database, and could thus not be included

[c] includes the sum of animal fractions that are not transferred to the consumer and food-processing industry

[d] based on EFPRA (2017)

Eleven percent of pork carcasses, 15% of beef carcasses and 16% of lamb carcasses are **bone**. With a P content of about 10% (Beighle et al., 1994; Hemme et al., 2005), **this material represents a dominant proportion based of the P present in the non-edible animal by-products**. As a matter of fact, the majority of P (85-88%) exists as bone P in the body of vertebrates (Hua et al., 2005).

Also, significant volumes of **waste waters** are produced at the slaughterhouse. This category contains dung and urine from animal holding areas, washings from distribution vehicles, and waste water generated during the process of meat and bone meal production (European Commission, 2005). These waste waters are either treated on-site or discharged to municipal waste water treatment (European Commission, 2005). The on-site generated waste waters may have been in contact with category 1 animal by-products. Therefore, the P-recovery from these waste waters may be legally impeded as the category 1 materials may not be used for the production of fertilisers according to Regulation (EC) 1069/2009.

14.4.4 Rendering industry

Rendering transforms the non-edible materials discarded by the meat and other industries into stable, value-added materials. Rendering can be carried out on an industrial, farm, or kitchen scale. The majority of tissue processed comes from slaughterhouses, but also includes restaurant grease and butcher shop trimmings and expired meat from grocery stores. This material can include the fatty tissue, bones, and offal, as well as entire carcasses of animals condemned at slaughterhouses, and those that have died on farms, in transit, etc. The most common animal sources are beef, pork, sheep, and poultry. The rendering process simultaneously dries the material and separates the fat from the bone and protein. A rendering process yields a fat commodity (yellow grease, choice white grease, bleachable fancy tallow, etc.) and a protein meal (meat and bone meal, poultry byproduct meal, etc.). Rendering plants often also handle other materials, such as slaughterhouse blood, feathers and hair, but do so using processes distinct from true rendering. The mono feed animal bone processing industry is a specific rendering industrial operation, which is processing food grade category 3 animal bones only or category 3 and 2, and having specific output products, such as food grade gelatine, China Bone for porcelain industry, processed bone grist (bone meal) and organic ingredients including hydrolysate for pet food.

The work of van Dijk et al. (2016) indicates significant P losses of 294 kt P yr^{-1} from the slaughter residues for the reference year. It should be noted that **these numbers might be outdated at present because the entry into force of the EC Animal By-Products Regulation** (Regulation (EC) No 1069/2009). Following the crises related to the outbreak of foot-and-mouth disease, the spread of transmissible spongiform encephalopathies such as bovine spongiform encephalopathy (BSE), the further use of animal by-products was largely restricted. The disposal of all animal by-products was, however, not a realistic option as it would lead to unsustainable costs and risks for the environment, and contradicts with the sustainable management of these materials. The clear interest of EU Member States to use animal by-products for a wide range of applications in sustainable manner was reflected in EC Regulation No 1069/2009 as outlined in Figure 30.

Altogether, the European Fat Processors and Renderers Association (EFPRA) process annually about 17 Mt of material in Europe. **From this 17 Mt, about 12 Mt is classified as category 3 materials, about 0.8 Mt as category 2 materials and 4.6 Mt as category 1 material** (EFPRA, 2017). Given the limited treatment disposal options for category 1 animal by-products, these materials are almost exclusively used for renewable energy production (Dobbelare, 2017) (Figure 30). Category 2 material is mainly used for the production of biodiesel (fat fraction), and fertilisers (protein fraction) (Dobbelare, 2017) (Figure 30). About 180 kt of category 2 protein material is used for the production of (organic) fertilisers in the year 2016 (Dobbelare, 2017). The fat fraction of Category 3 animal by-products are typically processed to produce Processed Animal Proteins (PAP), that can be used for pet food, animal feed including and fish feed, oleochemicals, edible fats and biodiesel (EFPRA, 2017). The protein fraction for category 3 material equalled 2.7 Mt of material in 2016, and was dominantly used for the production of pet food (~70%), (organic) fertilisers (18%), and to a smaller extent for fish feed, human food (gelatin) and fur feed (EFPRA, 2017) (Figure 30). About 510 kt of (organic) fertilisers were produced from category 3 material in the year 2016. **Assuming a P content of 5.3% for the protein fraction (Moller, 2015), the current fertiliser volumes of category 2 and 3 category animal by-products would equal about 27 kt P yr^{-1}.**

Time series of the fate of animal by-products for France confirm that the fraction of category 3 animal by-products that is incinerated for energy recovery has significantly decreased over the last decade (2006 – 2015), and that the relative share used for pet food production significantly increased (SIFCO, 2017). **It is concluded that the entry into force of Regulation (EC) No 1069/2009 effective increased the re-utilisation of animal by-products of category 3.** Therefore, the pet food sector is a significant competing industry for the fertiliser industry that produces plant nutrition products from category 3 animal by-products.

Therefore, the current P fraction that is dissipated is largely restricted to P present in category 1 (4.6 Mt material yr^{-1}) that is incinerated for energy recovery or used as biodiesel after prior processing steps.

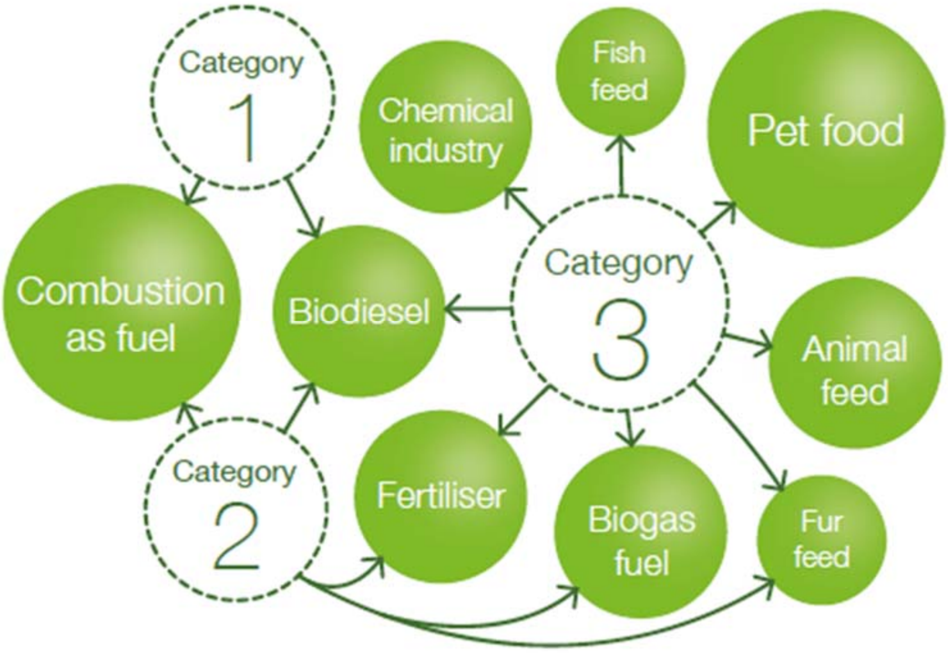


Figure 30: Potential application of processed animal by-products of category 1, 2 and 3 (source: EFPRA (2017))

Animal by-products other than manure and derived products not intended for human consumption are currently already used for the manufacturing of STRUBIAS materials. The mono-incineration of 1 tonne of animal derived meal and grist generates about 100-300 kg of **ash** (Coutand et al., 2008). During the incineration process, all organic matter in the material, including proteins, is transformed to CO_2, H_2O and nitrous and sulphur oxides, etc. Minerals like Ca, Mg and P are relatively stable in response to heating (Deydier et al., 2005a; Zheng et al., 2013). As a result, meat and bone meal ashes have high P (average 14.0%, range 6.1% - 18.9%) and Ca contents (20.9%), but low N contents (average 0.17%) (Deydier et al., 2005a; Wopenka and Pasteris, 2005; Czaja and Hermann, 2011). The combustion induces a wide range of structural modifications, such as crystalization of calcium phosphate, substitution reactions, etc. These processes reduce the P-solubility and therefore the value of the meat and bone meal ashes as P fertiliser in comparison to the original substrate (Moller, 2015). At present, however, co-incineration is the dominant thermochemical pathway due to the ease of operation and increased energy revenues. Mono-incineration is only applied at specific facilities (e.g. Kalfos – SARIA) that produce Kalfos fertiliser based on meat and bone meal ashes. Bone grist is also the input material for the **production of Animal Bone biochar** (3R

AgroCarbon, 2016), a fertiliser material with a P content of 13%. **Calcium phosphates can be precipitated** when degreasing animal residues (bone) during the demineralisation of the liquor during **gelatine manufacturing**.

14.5 Effluents and residues from municipal waste water treatment plants

Waste water treatment plants process grey/black water from households and industrial waste water produced. The dominant share of P from **industrial waste waters** present in **industrial sewage** is found in the sludge from the meat industry, potato industry, dairy industries, vegetable industry, and pulp and paper industry (Geertjes et al., 2016), and will be covered in the respective sections.

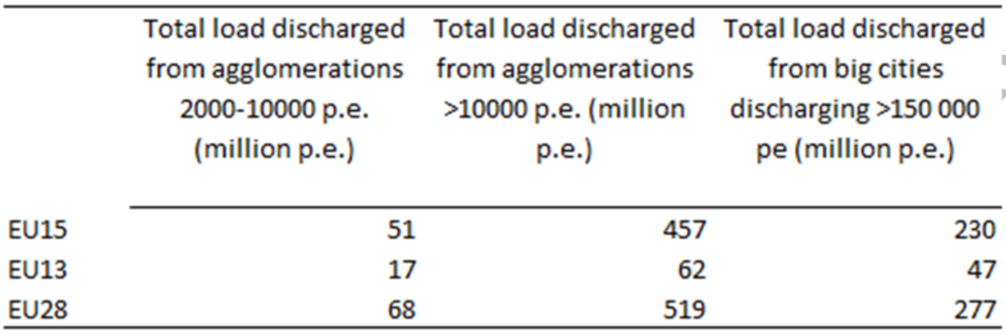
Municipal sewage is used water mainly coming from domestic activities and sometimes combined with used water from industrial activities and/or with surface run-off. The Urban Waste Water Treatment Directive (91/271/EEC) defines an **agglomeration** as an area where the population and/or economic activities are sufficiently concentrated for urban waste water to be collected and conducted to an urban waste water treatment plant or to a final discharge point. The size of an agglomeration in terms of generated pollution load is measured in **"population equivalent"** (p.e.). This is the organic biodegradable load that has a five-day biochemical oxygen demand (BOD5) of 60 g of oxygen per day, or in more popular terms – the organic biodegradable load generated by one person per day.

Sewage treatment is the process of removing contaminants from wastewater. It includes physical, chemical, and biological processes to remove these contaminants and produce environmentally safe treated wastewater (or treated effluent). **Primary** treatment removes part of the suspended solids, while **secondary** (biological) treatment uses aerobic or anaerobic micro-organisms to decompose most of the organic matter and nutrients (mostly N). **Tertiary** (advanced) treatment removes the organic matter and nutrients even more efficiently. It generally includes P retention and in some cases enhanced N removal. Nitrogen removal is regularly achieved through biological N removal through N reducing pathways (denitrification, possibly Anammox) that **remove the N from the system** as gaseous compounds, although specific recovery options are available (e.g. ammonia stripping).

The main objective of the **Urban Waste Water Treatment Directive** (91/271/EEC) defines in which conditions the waste water should be collected and treated before being released in the environment according to the quality of the receiving waters. The directive includes **requirements for collection and treatment of wastewater** in all settlements (agglomerations) and areas of economic activity with a population equivalent (p.e.) larger than 2000. The connection rate in Central European countries is even higher, and exceeds 90%. About 80% of the population is connected to waste water treatment in Northern and Southern European countries. On the basis of data reported in 2010, about 67% of the total population is connected to wastewater treatment in the countries of Eastern Europe. **Advanced (secondary or tertiary) treatments for nutrient and organic matter removal are required for populations larger than 10 000 p.e**. The current population connected to plants with tertiary treatment is in the order of 70% in Northern and Central Europe, and about 50% for Southern and Eastern Europe (European Environment Agency, 2013b). Average connection rate in South-East Europe (Turkey, Bulgaria and Romania) is about 40%. About a quarter of the population in South Eastern countries is connected to collecting systems without treatment (European Environment Agency, 2013b).

Table 24 shows the generated organic pollution load that agglomerations discharge as a function of size class (European Commission, 2017a).

Table 24: The generated organic pollution load of urban waste water treatment agglomerations as a function of size class (European Commission, 2017a)



	Total load discharged from agglomerations 2000-10000 p.e. (million p.e.)	Total load discharged from agglomerations >10000 p.e. (million p.e.)	Total load discharged from big cities discharging >150 000 pe (million p.e.)
EU15	51	457	230
EU13	17	62	47
EU28	68	519	277

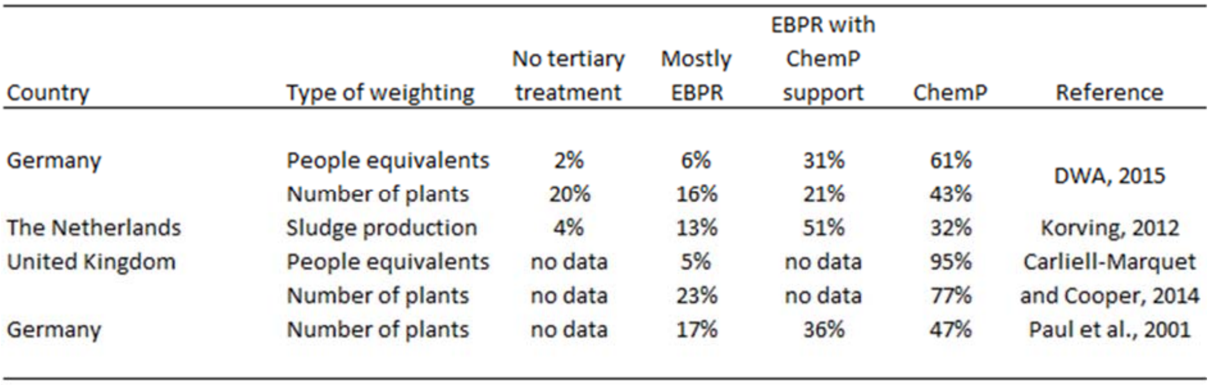
Different configurations exist for P removal in waste water treatment plants, with the most popular techniques being **enhanced biological phosphorus removal (EBPR)** and **chemical phosphorus removal (Chem-P)** using Fe or Al salts.

The major feature of the **EBPR process** is that organic matter uptake and P release take place under anaerobic condition and P uptake takes place under subsequent aerobic zone (Zhang et al., 2013). The enhanced phosphorus removal is attributed mainly to a group of selectively enriched heterotrophic bacteria, i.e., **P-accumulating organisms (PAOs)**. PAOs store carbon sources as intracellular polymers [i.e., poly-β-hydroxyalkanoates] using the energy of polyphosphate (polyP) and glycogen degradation and then release orthophosphate (orthoP) into the outside under anaerobic conditions. In the subsequent aerobic phase, PAOs use the stored poly-β-hydroxyalkanoates as an energy source, transforming orthoP to polyP, replenishing glycogen and self-growth. Phosphorus is accumulated in the sludge and is removed by sedimentation and discharging the P-rich excess sludge.

Chemical treatment for phosphorus removal involves **the addition of metal salts** to react with soluble phosphate to form solid precipitates that are removed by solids separation processes including clarification and filtration. The most common metal salts used are in the form of alum (aluminum sulfate), sodium aluminate, ferric chloride, ferric sulfate, ferrous sulfate, and ferrous chloride. The chemicals can be added before the primary settling, during secondary treatment or as part of a tertiary treatment process. Iron salts are usually preferred over Al salts due to their lower cost (Wilfert et al., 2015). Apart from P removal, Fe plays an important role to prevent hydrogen sulfide emissions during anaerobic digestion and acts as a coagulant to improve sludge dewatering (Charles et al., 2006; Ge et al., 2013).

Chem-P is more prevalent than EBPR in municipal waste water treatment plants in Europe (Paul et al., 2001; DWA, 2005; Korving, 2012; Carliell-Marquet and Cooper, 2014; Wilfert et al., 2015).

Table 25: Configurations used in enhanced phosphorus removal methods used in selected EU Member States (adopted from Wilfert et al., 2015)



Country	Type of weighting	No tertiary treatment	Mostly EBPR	EBPR with ChemP support	ChemP	Reference
Germany	People equivalents	2%	6%	31%	61%	DWA, 2015
	Number of plants	20%	16%	21%	43%	
The Netherlands	Sludge production	4%	13%	51%	32%	Korving, 2012
United Kingdom	People equivalents	no data	5%	no data	95%	Carliell-Marquet and Cooper, 2014
	Number of plants	no data	23%	no data	77%	
Germany	Number of plants	no data	17%	36%	47%	Paul et al., 2001

Although removal of phosphorus from waste water is a requirement under Article 5 of the Urban Waste Water Treatment Directive (91/271/EEC), **this directive does not require extraction of the P in a useable form**. One particular feature of the Directive is that it allows the flocculation of the phosphorus using aluminium and iron, which produces a strongly bound compound from which the P is not easily commercially recoverable and which may not be fully available to plants (Wilfert et al., 2015).

Sewage sludge is the semi-solid material or slurry that remains after the treatment. Sewage sludge is an organic substrate relatively **rich in nutrients, but also might contain a substantial amount of organic and inorganic contaminants**. The material has to undergo further treatment before being suitable for disposal or land application. According to the CEN (European Committee for Standardization), sludge is defined as "*a mixture of water and solids separated from various types of water as a result of natural or artificial processes*". In sewage treatment works, **sludge is further treated** to reduce its water content (thickening, dewatering, drying, or a combination thereof) and to increase its stability (anaerobic digestion, aerobic stabilization, lime stabilization, composting, or a combination thereof).

The current **fate of treated sludge** includes disposal in landfills, application to land, incineration, or composting (Figure 31). Existing national and EU regulation also set progressive limits on disposing sludge in landfill and its direct application to land, especially in densely populated EU regions (Buckwell and Nadeu, 2016). This has caused an increase and interest for **the importance of alternative disposal and recycling routes for growing amounts of sewage sludge**.

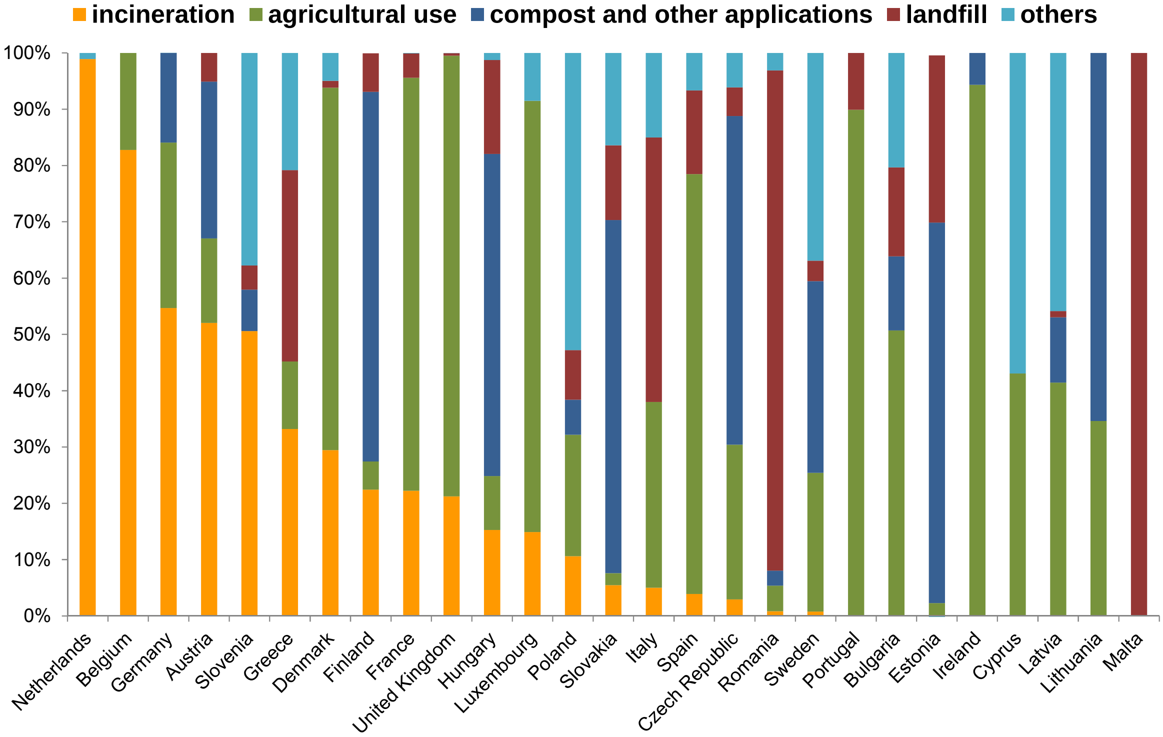


Figure 31: Routes for sewage sludge disposal in the year 2012 in EU-27 (year 2010 data for Italy) (data available from Eurostat).

In 2012, about **23% of the sludge is incinerated** in Europe (2.3 Mt dry sludge yr^{-1}), meanwhile **49%** (5.0 Mt dry sludge yr^{-1}) of the sludge is directly **returned to land for agricultural use** (Eurostat, 2012). Nevertheless, large differences in the proportional contribution of sewage sludge disposal routes exist among Member States for the fate of sewage sludge (Figure 31). Countries with a high population and animal density, and strict restrictions on landfilling, incinerate high proportions of their sewage sludge (e.g. Netherlands, Belgium, Germany and Austria) (Figure 31). Other Member States (Denmark, France, United Kingdom, Luxembourg, Spain, and Portugal) apply large amounts of the sewage sludge directly on land. Finally, Member States as Greece, Italy, Romania, Estonia and Malta landfill significant amounts of sludge (Figure 31).

Although the Sewage Sludge Directive (91/271/EEC) has established the conditions for safe use of sludge on agricultural land, but concerns exist, notably as regards the maximum limit values for cadmium and other contaminants are considered to be too high (European Commission, 2013a). Sixteen Member States have adopted more stringent standards than those set out in the Directive. **Therefore, direct sewage sludge application on agricultural land is progressively decreasing in Europe.** Harmonisation of higher quality standards would encourage greater confidence amongst farmers and consumers on the safe use of sludge in the EU. In order to encourage more efficient resource use in the future, these issues will need to be addressed so that standards for sewage sludge-derived products inspire confidence right across the chain of end users: namely farmers, retailers and ultimately consumers.

A **detailed description of the sewage sludge mono-incineration process** is given in Donatello and Cheeseman (2013). Sludge and hot compressed air (ca. 500–600°C) are fed to the combustion chamber. The sand bed temperature is typically 750°C and the overhead freeboard zone at 800–900°C. Temperatures can be finely controlled by the injection of water or liquefied gas oil. The sand bed acts as a ‘‘thermal fly wheel’’ and helps stabilise temperature fluctuations in the incinerator. Particle residence times in the combustion chamber are typically only 1–2 s and during this time water is evaporated, volatile metals vapourise and organic compounds are combusted completely to gases, either directly or via the formation of an intermediate char. The remaining inorganic material is carried out of the chamber as fine particulates with the exhaust gases. During incineration, **most of the N is released into the atmosphere while much of the P and K, and heavy metals are retained in the fly ash** (Deydier et al., 2005a; Zheng et al., 2013). Sulphur (S) is retained in the air pollution system, and can possibly be recycled as Na_2SO_4 from the alkaline scrubber. The fly ash is generally removed by bag filters, electrostatic precipitators or cyclones after passing through a heat exchanger. The flue gas is then treated using a wet scrubber with acid, alkali and possibly activated carbon dosing to comply with emission limits, as required by Industrial Emissions Directive (2010/75/EU). The scrubbing process produces an additional waste sludge, which is dewatered and normally disposed of in hazardous waste landfill. **Mono-incineration** produces **fly ash with high P contents (2-12%)**. The average P content in sewage sludge ashes is 10.8 % in the Netherlands (CBS Statistics Netherlands, 2015) and 8.9% in Germany (Krüger and Adam, 2015). In Germany, about half of the generated municipal sewage sludge is currently already processed within mono-incineration plants (Adam et al., 2015).

Sludge can also be **co-incinerated with municipal solid waste or industrial waste** in existing general purpose incineration plants that produce energy. The ash produced has a lower P concentration (e.g. on average 4.9% in Germany), and potentially a higher amount of impurities and contaminants.

In the EU-28, it is estimated that annually generated municipal waste waters contain **2.3-3.1 Mt of N and around 0.50 Mt of P** (Sutton et al., 2011; Leip et al., 2014; van Dijk et al., 2016). About 227 kt P yr^{-1} ends up in communal sewage sludge, and an additional 74 kt P yr^{-1} is lost as effluents from urban and decentralised waste water treatment plants. Untreated and uncollected waste waters sum up a total of about 59 kt P yr^{-1}. The remainder 140 kt P is contained in sewage sludge that is directly applied on agricultural land.

STRUBIAS materials can be produced from (processed) wastewaters and sludges at municipal waste water treatments plants, as well from the incinerated sewage sludges (see section 15).

14.6 Food processing industry residues other than animal by-products

Three different food processing sectors, other than the meat and fish industry, have been identified that show a discernible potential for P-recovery. Losses from food processing industries correspond **to 44 kt P and 157 kt N per year** (Sutton et al., 2011; van Dijk et al., 2016). The share of P can be split up in losses in solid forms (e.g. sludges: 36 kt P) and losses as wastewaters (9.2 kt P). Currently, **P is recovered in the form of precipitated phosphate salts & derivates from the wastewaters from the potato and dairy industry**. Given that the sludges are characterised by high moisture but low P contents, supplementary STRUBIAS recovery from the sludges will most likely be in the form of precipitated phosphate salts & derivates, eventually after anaerobic digestion of the residues.

14.6.1 Potato crisps and chips industry

Two of the main potato-based products are crisps and chips. The manufacturing of both essentially consists of peeling the raw material, slicing to an appropriate size and blanching, followed by frying to achieve the desired sensory properties. To prevent colourisation of the potato, a substance **called pyrophosphate** ($Na_2H_2P_2O_7$) is used to complex iron (Fe^{2+}). In this way sodium acid pyrophosphate prevents that iron in the potato reacts with chlorogenic acid during the heating processes (Rossell, 2001). The oxidation of the Fe^{2+}-chlorogenic acid complex by oxygen from the air would otherwise result into a grayish-colored substance that causes the after-cooking gray discoloration (Rossell, 2001). This is a very significant P-source in the waste water from potato processing installations. The waste water have a typical PO_4^{3-}–P concentration of about 200 mg L^{-1} (European Commission, 2017b). **Average P-recovery efficiencies of 80–90% have been reported**. The cost of recovery is lower compared to phosphorus removal by chemical precipitation using, for example, $FeCl_3$ (European Commission, 2017b). At present, more than 4 t of struvite per year is produced by the potato-processing in Italy, Belgium and the Netherlands (STRUBIAS sub-group comments; Dewaele, 2015).

14.6.2 Waste waters from the dairy, brewery, grain, fruit and vegetable industry

Plant-derived waste arises from cultivated grains, fruits, and vegetables. A determined fraction of the plant materials are not edible and are thus treated as agrowaste. Spillage, spoilage and storage loss or outgrading, pest infestation, and loss of quality during storage can be the main reasons for loss of agricultural produce after harvesting. Additional waste is generated during processing stages such as peeling, washing, boiling, and slicing. Finally, byproducts such as pomace and spent grain are formed, and wastes from plant shutdowns or washing occur (de las Fuentes et al., 2003).

Water consumption is one of the key environmental issues for the food processing sector. Dairy and brewery industries are major water consumers, producing waste waters that are generally not dangerous but are heavily loaded with organic matter (Gendebien et al., 2001). The composition of the effluents is quite variable in composition. Compared to effluents from the chips and crisp industry, dairy, brewery and starch manufacturing industries have – generally speaking - less P in their waste waters.

A significant proportion of the waste waters are originating from the washing of installations. Typical **cleaning agents** used in the food-processing industry sector are (European commission, 2006a):

- alkalis, e.g. sodium and potassium hydroxide, metasilicate, sodium carbonate;
- acids, e.g. nitric acid, phosphoric acid, citric acid, gluconic acid;
- pre-prepared cleaning agents containing chelating agents such as EDTA, NTA, phosphates, polyphosphates, phosphonates or surface-active agents;
- oxidising and non-oxidising biocides.

The use of chelating agents and biocides may hamper nutrient recovery as the contaminants may be transferred to the recovered material.

Many **dairies** use large amounts of water, mainly for cleaning. The PO_4^{3-}–P concentration in the waste water varies between 20 and 200 mg L^{-1} (European commission, 2006a). Many dairies have built their own effluent treatment plant and produce large amounts of sludges rich in P and organic matter. Humana Milchunion E.G. has installed a struvite reactor to recover P from dairy wastewater effluents (PO_4^{3-}–P concentration in the waste water: 60 – 65 mg P L^{-1}; P-recovery efficiency of about 75%).

In the **brewery**, waste water from the anaerobic reactor is driven to a reservoir where it is mixed with aerobic sludge (from the second sedimentation) and with untreated neutralised waste water (taken before anaerobic digestion). The PO_4^{3-}–P concentration in the waste water varies between 0 and 200 mg L^{-1} (Gendebien et al., 2001). All these flows are recirculating and, in these conditions, the aerobic sludge encourages the growth of P-assimilating bacteria. Phosphorus could then be recovered after the bacterial release of orthophosphates.

Sugar mills produce wastewater, emissions and solid waste from plant matter and sludge washed from the sugar beet (Hess et al., 2014). Sugar beet is 75% water, and the extraction process, by definition, aims to release a high proportion of water contained in the beets. The technique applied for sugar extraction from plant tissues has an impact on the volumes of water used (consumed and polluted) to produce sugar (Bio Intelligence Service - Umweltbundesamt - AEA, 2010). Considering the high nutrient contents of the sugar beet, the waste generated during the sugar beet processing is also rich in N and P (Buckwell and Nadeu, 2016). Gendebien et al. (2001) indicated, for instance, effluent P concentrations of > 100 mg P L^{-1}.

Moreover, **materials from other food industry sectors can be used for the production of STRUBIAS materials, mostly thermal oxidation materials and pyrolysis materials of lower P-content.**

14.7 Chemical industry waste waters

The EU chemical industry sector provides a significant contribution to the EU economy. It is one of its most international and competitive industries, connected to a wide field of processing and manufacturing activities (European commission, 2014). **Specific chemical industry subsectors are responsible for the most significant emissions of macronutrients to water, especially P** (E-PRTR, 2013; European commission, 2014). Emissions of P to water by the chemical industry sector equal a total of **10.3 kt P** for the year 2010. The dominant contribution originates from **manufacture and formulation of pharmaceutical products** (9.41 kt P), with only a minor share from other chemical industries such as basic organic chemicals (0.45 kt P), basic inorganic chemical (0.32 kt P) and fertilisers (0.12 kt P) (E-PRTR, 2013; European commission, 2014).

Pharmaceuticals are produced using synthesis or fermentation. Organic wastes produced in the pharmaceutical industry are mainly **biomass** (cells from the fermentation process), **synthesis residues, alcohol and organic solvents** from the cleaning process, product residues and dust from reprocessing (Gendebien et al., 2001). Care has to be taken where residues originate from the pharmaceutical industry as it is very **difficult to fully remove traces of the pharmaceutical end product and hazardous solvents from the waste waters**. Aqueous wastes from the manufacture, formulation, supply and use (MFSU) of pharmaceuticals is classified as hazardous waste according to the European List of Waste pursuant to Directive 2008/98/EC.

At present, Genzyme bvba makes use of a struvite reactor for P-recovery in the form of precipitated phosphate salts & derivates from their pharmaceuticals production plant in Geel, Belgium. The P-rich wastewaters (55 mg PO_4^{3-}-P) are used for the production of 220 kg of struvite (Dewaele, 2015).

14.8 Iron and steel industry residues

Blast furnace slag is a by-product of the manufacture of iron by thermochemical reduction in a blast furnace. It is formed in a continuous process by the fusion of limestone (and/or dolomite) and other fluxes with the residues from the carbon source and non-metallic components of the iron ore. Oil, tar, natural gas, powdered coal and oxygen can also be injected into the furnace to combine with the coke to release additional energy which is necessary to increase productivity. Silicate and aluminate impurities in the ore and coke are chemically bound to lime (CaO), and then removed as a molten slag with a complex Ca-Mg-Al-silicate composition. Blast furnace slag is generated at temperatures above 1500°C. Dependent on the way of cooling of the liquid slag it can be distinguished between crystalline air-cooled blast furnace slag and glassy granulated blast furnace slag.

Steelmaking slags are residues of processing molten iron into a specific type or grade of steel (Reijonen, 2017). Today there are two major commercial processes for making steel, namely basic oxygen steelmaking, which has liquid pig-iron from the blast furnace and lower amount of scrap steel as the main feed materials, and electric arc furnace steelmaking, which uses scrap steel or direct reduced iron as the main feed materials.

Basic oxygen furnace slag is formed during the conversion of liquid iron (hot metal) into crude steel during a batch process in a basic oxygen furnace. The slag is generated by the addition of fluxes, such as limestone and/or dolomite, during blowing oxygen into the melt. Due to the oxidising conditions, some elements (like Fe and Mn) are partly oxidised and contribute to the formation of the slag. Furthermore some components are either oxidised to gas (like carbon) or are chemically bound in the slag (like silicon or phosphorus). The liquid slag which has tapping temperatures of around 1600°C is air-cooled under controlled conditions in pits forming crystalline slag. The main components of a basic oxygen furnace slag are complex Ca-Mg-Fe-Mn-silicates.

Electric arc furnace slag is produced when scrap metal and fluxes are oxidized by the use of an electric current. Chemical energy is supplied via several sources including oxy-fuel burners and oxygen injections. Oxy-fuel burners combust natural gas using oxygen or a blend of oxygen and air. In some operations, oxygen is injected via a consumable pipe lance to "cut" the scrap. The oxygen reacts with the hot scrap and burns iron to produce intense heat for cutting the scrap. Once a molten pool of steel is generated in the furnace, oxygen can be lanced directly into the bath. This oxygen will react with several components in the bath including, aluminum, silicon, manganese, phosphorus, carbon and iron. All of these reactions are exothermic and thus supply additional energy to aid in the melting of the scrap. The metallic oxides that are formed will end up in the slag. Electric arc furnace slags are crystalline materials with a complex Ca-Mg-Fe-Al-Mn-silicate composition.

Nowadays, basic oxygen steelmaking and electric arc furnaces account for virtually all steel production (Jewell and Kimball, 2014). On average the production of one tonne of steel results in 200 kg (via electric arc furnace) to 400 kg (via blast furnace and basic oxygen furnace) of residues. These include slags, dusts, sludges and other materials.

Blast oxygen furnace slag is formed in the basic oxygen converter during the conversion of pig iron into crude steel. In this process, molten metal from blast furnace is treated with oxygen to remove impurities via oxidation at 1400–1650 ºC (Yildirim and Prezzi, 2011). Oxidation is followed by slag formation with burned lime. The principal components of both slags are silicates, aluminates and oxides of Ca, or to lesser extent of Mg (Waligora et al., 2010). **Granulated blast furnace slag** is formed in the smelting process of iron ore/pellets with coke and flux (limestone, burned lime or dolomite). Silicate and aluminate impurities in the ore and coke are chemically bound to lime (CaO), and then removed as a molten slag. Oil, tar, natural gas, powdered coal and oxygen can also be injected into the furnace to combine with the coke to release additional energy which is necessary to increase productivity. **Electric arc furnace slag** is produced when scrap metal and fluxes are oxidized by the use of an electric current. Chemical energy is supplied via several sources including oxy-fuel burners and oxygen injections. Oxy-fuel burners combust natural gas using oxygen or a blend of oxygen and air. In some operations, oxygen is injected via a consumable pipe lance to "cut" the scrap. The oxygen reacts with the hot scrap and burns iron to produce intense heat for cutting the scrap. Once a molten pool of steel is generated in the furnace, oxygen can be lanced directly into the bath. This oxygen will react with several components in the bath including, aluminum, silicon, manganese, phosphorus, carbon and iron. All of these reactions are exothermic and thus supply additional energy to aid in the melting of the scrap. The metallic oxides that are formed will end up in the slag.

Slags that have undergone **a thermal oxidation at high temperatures** could be considered **as thermal oxidation materials & derivates in the STRUBIAS project**. The sTRUBIAS sub-group indicated that at present, approximately 800 000 tonnes of these slags are used as **fertilising products within the EU, mostly in the form of liming materials and P-rich slags that are used as fertilisers.**

14.9 Forest-based industry residues and green waste

14.9.1 Woody residues

This category includes woody residues that originates from (1) **sidestreams produced by the woodworking industry** for instance harvest slash, sawmill sidestreams and shavings from timber yards, materials from chipboard and other timber processing, reclaimed timber from buildings, pallets and packing crates, (2) currently **uncollected forest residues** and (3) the processing of plant materials other than residues **from households waste** classified as food waste (including parks and garden waste).

The **total amounts of wood residues** that are available within the EU for posterior use are **estimated at about 120 Mt per year**, with future predictions remaining more or less at the same level (Searle and Malins, 2013). This number is the sum of following fractions:

- According to Manteau (2012), **26 million tonnes of post-consumer wood** (i.e. wood products such as furniture that are discarded) was generated in 2010. Of this, 7.8 million tonnes was recycled into other materials and 10.3 million tonnes was burned for energy in power plants or households. About 8 million tonnes was permanently disposed of or combusted without energy recovery.
- Forests and other wooded land occupy over 44% of the EU's surface and represent 5% of the world's forests. In the last 50 years, both their area and the standing timber volume (growing stock) have continued to grow. Nowadays, they gain almost 700 000 ha annually. According to Searle and Malins (2013), it was estimated that the total production of **forestry residues in the EU was 80.7 million tonnes** in 2011. Some forestry residues are currently collected, but according to ECF (European Climate Foundation, 2013), the current usage of forestry residues in the EU is only about 3%, with activities mainly occurring in Scandinavia. Similar to crop residues, a share of the forestry residues should remain on land to protect soil carbon and sustainable ecosystem functioning. To be conservative and to avoid other unintended consequences, Searle and Malins (2013) assumed that **50% removal of forestry residues may be sustainable** if combined with good management practices.

Assuming these values, a total of about 40 million tonnes of uncollected forestry residues might potentially be available for nutrient recovery.

- Presumably much of the 19.7 million tonnes of **household vegetal waste are garden clippings and other wood residues** (Searle and Malins, 2013).

Wood **treated** with preservative chemicals such as pentachlorophenol, lindane or copper chrome arsenate **may hinder its posterior use**, including recycling and energy recovery. **Untreated wood waste is a material with high organic matter content, but with a relatively low nutrient content,** both in terms of N (often < 1%) and P (~0.1%). Gendebien et al. (2001) indicated an average P content of 0.09%, but no numbers were given for N. Wood N/P contents vary between 10 and 28 (Mooshammer et al., 2014; Sardans and Peñuelas, 2015), for which we estimate assume an N content of 1.5%. The high C/N ratio makes it an unsuitable material for direct fertiliser applications as it will promote microbial N immobilization and thus reduce the N availability in the soil. It may, nevertheless, be used as a mulch to discourage weed growth and conserve moisture or as an aggregate for compost. The total nutrient content of forest residues can then be calculated by multiplying abovementioned numbers on forest residue availability with the assumed nutrient contents. Assuming an estimated moisture content of 25-50%, 900 – 1350 kt N and 54-81 kt P yr^{-1}. These numbers are generally in line with the estimated P-content of **76 kt P as estimated** by Van Dijk et al. (2016).

Bark and wood residues from wood handling can be combusted **for energy recovery**. Wood ash from bark boilers contains nutrients taken from the forest with the wood raw material and this ash can be suitable as a fertiliser as long as the wood originated metals like Cd meet harmonised requirements for fertilisers. Wood biomass is used for energy production is many EU Member States, especially in northern Europe. Wood combustion in Denmark, Finland and Sweden generate >290 kT of biomass ashes, whereas the combined wood ashes of Austria, Germany, Ireland, Italy and the Netherlands add another 300 kT of wood ashes (van Eijk et al., 2012). Assuming a P content of 0.1%, the **wood bottom and fly ashes in these European countries thus contain only 0.3 kT of P**. Moreover, competing uses (concrete industry, fill and ground remediation) exist for these biomass ashes (van Eijk et al., 2012).

Given their high carbon content, wood residues are mainly used for energy production, but the combustion residues are often applied on (forested) land in north European countries (Insam and Knapp, 2011). Wood material is currently also the **dominant input materials for the production of pyrolysis & gasification materials** that are used as soil improver (EBC, 2012).

14.9.2 Pulp and paper industry

For a complete overview of the processing of woody material and recovered materials, it is referred to the Best Available Techniques (BAT) Reference Document for the Production of Pulp, Paper and Board (European Commission, 2015a).

The production process used in papermaking depends on the stock used to generate the fibre (Gendebien et al., 2001). When virgin wood fibre is used to produce **paper**, the pulp creates liquid effluent and the **sludge** mainly contains lignin and cellulose. When waste paper is used in the process, de-inking and bleaching is required, and the de-inking sludge will contain chemical residues. The process of reusing fibre from recycled paper produces large amounts of sludge (1 tonne of sludge for every tonne of paper produced) (Gendebien et al., 2001). De-inking sludge will also contain high levels of carbon, calcium carbonate and, generally, aluminium silicate (Gendebien et al., 2001). Within the paper industry, the most economic choice for sludge disposal can determine the process used (Gendebien et al., 2001).

Pulp and paper primary sludge comprises cellulose fibre (40 to 60% of dry solids). De-inking primary sludges also contain printing inks and mineral components (40 to 60% dry solids: kaolin, talc, and **calcium carbonate**). The abundance of metals in the sludges has significantly decreased over the last decades due to stricter legislation on metal/metalloid contents of the ink. The P content of the pulp and paper industry sludges is however relatively low (0.3% P on average; Gendebien et al., 2001).

The **incineration of the sludges** from the dissimilar paper and cardboard producing and pulp processes is commonly applied in the sector. The use of auxiliary fuel may be necessary to maintain good burning conditions unless the sludge is mixed with bark and other wood waste material. Burning reduces the volume of waste and the **inorganic content remains as ash**. It is noted that specific effluents cannot be sent for incineration as a consequence of the use of certain chemicals (e.g. chlorine dioxide).

The **sector BAT conclusions** indicate that the recovery of energy by incinerating wastes and residues from the production of pulp and paper that have high organic content and calorific value is permitted on condition that the recycling or reuse of wastes and residues from the production of pulp and paper is not possible (European Commission, 2015a).

14.9.3 Emissions to water

Liquid effluents contain material in colloidal or suspended forms and dissolved substances. With few exceptions (eucalyptus pulping), **the original levels of N and P are low and are added to biological waste water treatment plants to feed the biomass**.

14.9.4 Conclusion

From an economic point of view**, the cost of harvest, transport, and processing is disproportional in relation to the P quantities that can be recovered**. Therefore, it is more likely that any P-recovery from **wood will be formed as part of cascades where synergies exist between the manufacturing of other products (energy, paper) and nutrient recovery.**

14.10 Municipal solid waste

About **258 Mt of municipal solid waste** (MSW) is produced yearly in the EU-27 (Eurostat, 2016). The N and P of the biodegradable waste fractions mainly originate **food waste and woody residues**. Also some N could be present in textile materials. The biodegradable fraction represents on average 37% of all municipal solid waste, although the fraction varies widely between EU countries (European Commission, 2010b).

Municipal solid waste raises problems since it is a **mixture of materials that are heterogeneous in nature and not segregated**. The composition of MSW varies regionally, but usually contains a mixture of organic waste, paper and cardboard, textile waste, plastics, metals, glass and potentially some biomedical waste and hazardous (battery, nail polish bottles, insecticides) compounds (Sokka et al., 2004; Chandrappa and Das, 2012).

A MSW can undergo a **mechanical sorting** of the waste into a biodegradable material containing fraction and a non-biodegradable material containing fraction. In the latter case, nutrient recovery from the biodegradable fraction is possible through composting and anaerobic digestion.

In the alternative scenarios, the MSW is not separated and may either be **landfilled** (resulting in a complete loss of the material, including its nutrients) **or incinerated.**

The ashes from MSW generally contain relatively **low amounts of P**, with values of approximately 0.4% P (Kalmykova and Fedje, 2013).

14.11 Others

Abovementioned input material represent the overall share of the nutrients present in waste and have therefore the greatest potential for nutrient recovery options. **The STRUBIAS sub-group did not identify waste materials, industrial residues or biological materials other than those mentioned in sections 14.2 - 14.9.**

Nevertheless, other streams originating from secondary raw materials are not de facto excluded as STRUBIAS input materials. Based on detailed information provided from EU Member States, Gendebien et al. (2001) provides an excellent, though somewhat out-dated, overview of the mass amounts, nutrient contents and environmental concerns and health issues for numerous other waste streams that are currently spread on agricultural land**: tannery sludge, decarbonation sludge, inorganic waste from chemical industry, textile waste, wool scourers waste, waste lime from cement manufacture or gas processing, waste gypsum, waste from energy production and dredgings.**

It should be noted that some of these streams (e.g dredgings, waste lime, and waste gypsum) may **contain large amounts of biological contaminants and high amounts of heavy metals** that may potentially hinder nutrient recovery processes. Therefore, treatment may require a potentially large chemical and electrical demand in order to recover a relatively small amount of nutrients from the waste stream. Others streams may be suitable for use as such, in case they are low in contaminants identified.

14.12 Conclusion

Understanding the **complex flow of nutrients throughout the food and non-food production and consumption chains** in Europe is needed **to assess the feasibility and impact potential of different streams as input material** for fertilisers derived from secondary raw materials. The overview presented in this document builds further upon the information presented by the scientific community, multi-stakeholder research platforms, think tanks and international and non-governmental organizations (Sutton et al., 2011; Leip et al., 2014; Buckwell and Nadeu, 2016; van Dijk et al., 2016), complemented by own estimates as described in each of the input material sections.

STRUBIAS materials are currently produced from all listed input materials, but the deliberateness of nutrient recovery varies between the different materials. A nutrient recovery system can specifically be designed for nutrient recovery, with the aim:

- to produce **plant nutrition materials from secondary raw materials:** all processes;
- **to avoid the loss or spillage of nutrients into the environment**: manure, effluents and residues from municipal waste waters, food processing residues, residues from gelatin production process;
- to **remove nutrients for improved functioning of biological waste water treatment plants**;

Note that some STRUBIAS materials can be listed under different items because the production process can serve different goals and benefits at a time.

STRUBIAS materials can also be produced – with or without process adaptation - as a primary product or residue of a production process **aimed at the production of a different primary output:** energy, **clean water effluents,** poultry litter ashes, forest-based industry residues, iron and steel industry residues. In many occasions, STRUBIAS materials form part of a cascade that transforms secondary raw materials in a set of added value materials, and to reduce as such the burden on primary raw materials.

15 Production processes and techniques

The following section aims to give a general overview of different production processes and techniques of each of the three STRUBIAS material groups, as well as on the general principles applied in production processes. Note, however, that STRUBIAS materials are only recently becoming available on the market, and that there are many ongoing technical developments in the progressively emerging market sector. Therefore, the production processes are, by no means, exhaustive or complete.

15.1 Precipitated phosphate salts & derivates

15.1.1 Waste water treatment plants

In **conventional wastewater treatment** plants, **P** is mainly **eliminated** by **enhanced biological phosphorus removal (EBPR)** or by **chemical precipitation with metal salts (ChemP)** or a combination of both. With EBPR, microorganisms (P accumulating organisms, PAOs) incorporate P in a cell biomass compound called polyphosphate and the P is removed from the process by sludge wasting. Chemical precipitation with metal salts can remove the P to low levels in the effluent. The commonly used chemicals are aluminum (Al(III)), ferric (Fe(III)), ferrous (Fe(II)), and calcium (Ca(II)) ions. The direct use of P-rich sludge as a fertiliser is associated to an increasing number of concerns due to concerns related to pathogens, and uncertainties related to P bioavailability (Cox et al., 1997; Vaneeckhaute et al., 2016).

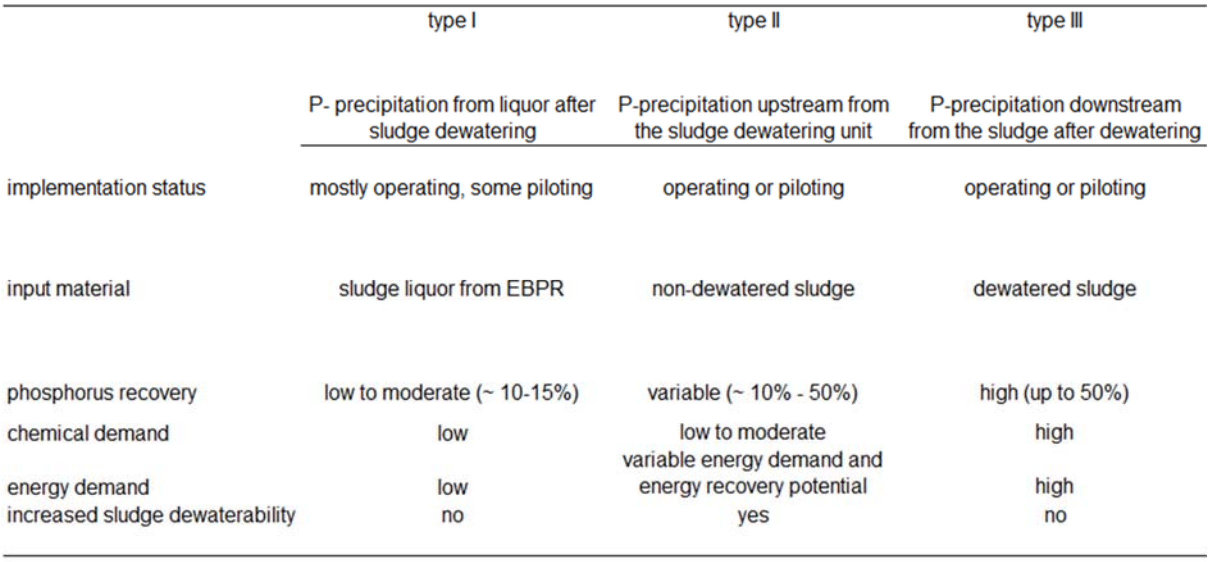
Phosphate salts can be recovered from **sludge liquor** and from **digested sludge** when the PO_4^{3-} precipitates together with Mg^{2+} or Ca^{2+}, possibly also trapping NH_4^+ and/or K^+ in the molecular structure. Struvite, the most commonly recovered phosphate salt, forms from equimolar quantities of Mg^{2+}, PO_4^{3-} and NH_4^+ implying that the efficiency of NH_4^+ removal is relatively low and the excess N remains in soluble form. In most sewage treatment applications **Mg is the limiting element**, for which it is added to the process as $MgCl_2$ or MgO. The formation of precipitates is strongly influenced by pH, hence if the feed stream does not have sufficient alkalinity, NaOH is added and/or CO_2 is stripped from the solution. More detailed information on the crystallization dynamics and kinetics for the struvite crystallization process is given in Le Corre et al. (2009).

Phosphorus recovery from sludge liquor and from digested sludge is limited to the amount of **soluble PO_4^{3-}**. For most waste water treatment plants, the latter is in the range of **5-20% of total P** load of the sludge under normal pH conditions (Jossa and Remy, 2015). P content of the solid phase can be substantially mobilized into the liquid phase (e.g through the **addition of volatile fatty acids**, **acidification**, or **thermal hydrolysis)** so that total P recovery rates of **up to 50%** are achievable.

Precipitation in the form of phosphate salts without those pre-treatments is essentially **applicable to those wastewater treatment plants where EBPR is used**. Here, the polyphosphates stored in the bacterial cells are partly released again under anaerobic conditions, thereby significantly increasing the PO_4^{3-}-P content in the sludge system to levels that support P recovery. The P content in wastewater treatment plants with EBPR and anaerobic digestion can be 75–300 mg L^{-1} PO_4^{3-}-P after the anaerobic digester (García et al., 2012). **P-recovery processes that include** pre-treatment step are in principal able to deal with sludge that had been subjected to chemical removal and coagulation with Al and Fe salts. Nevertheless, additional chemical or energy demand may be required when sludges rich in chemical coagulants are applied (Kabbe et al., 2015).

Four types of recovered phosphate forming processes are considered: (I) from the sludge liquor, (II) from the digested sludge, (III) downstream from the digested sludge after a pH regulating treatment, (IV) upstream from the digested sludge with thermal hydrolysis. The 4 types differ in their P-recovery rate, sludge input materials that be used, and energy and chemical demand (Table 26). For phosphate salt formation from the **liquor** (type I), the **concentrated side streams** after the anaerobic treatment or the dewatering unit after anaerobic digestion are the best options for P recovery. The implementation of a P-recovery system **before the anaerobic digester** (type II and type IV) reduces uncontrolled P precipitation in the anaerobic digester or post-digestion processes and enhances P recovery (Marti et al., 2008; Martí et al., 2010). Downstream P recovery from the **sludge phase** can include recovery from both the digester sludge before and after the dewatering unit.

Table 26: Overview of the principles and properties of production process for precipitated phosphate salts at waste water treatment plants or downstream sludge processing plants.



	type I	type II	type III
	P- precipitation from liquor after sludge dewatering	P-precipitation upstream from the sludge dewatering unit	P-precipitation downstream from the sludge after dewatering
implementation status	mostly operating, some piloting	operating or piloting	operating or piloting
input material	sludge liquor from EBPR	non-dewatered sludge	dewatered sludge
phosphorus recovery	low to moderate (~ 10-15%)	variable (~ 10% - 50%)	high (up to 50%)
chemical demand	low	low to moderate	high
energy demand	low	variable energy demand and energy recovery potential	high
increased sludge dewaterability	no	yes	no

- <u>Type I: P- precipitation from liquor after sludge dewatering.</u> Recovered phosphates, mostly struvites, can be formed **from the sludge liquor (i.e. reject water) in mixed stirred tanks**. The *PHOSPAQ*® and *ANPHOS*® processes operate in a single and two separate stirred tank reactors. An increase in pH (CO_2 stripping) and mixing are obtained via aeration, and MgO is added to the wastewater. The *NuReSys*® process differs from the *ANPHOS*® process since it is operated in continuous mode instead of batch, at a lower residence time. Another difference is the use of a different Mg source ($MgCl_2$) and the addition of a 29% NaOH solution to the crystallization reactor. The *Struvia*® process relies on the use of a continuous stirred tank reactor with integrated solid/liquid separation by calming zone and lamellar packing or with additional lamella settler. Also the *Phorwater*® and *Prisa*® technologies rely on the struvite crystallization in a continuous liquid flow system. In the PHORWater® process the elutriation of the mixed sludge (primary and EBPR sludge) allows reducing the P load entering the anaerobic digester and achieving a high P concentration in the supernatant of the sludge thickener (Martí et al., 2010; Bouzas et al., 2016). In the *Ekobalans*® (pilot) plant, struvite precipitation is a simple, low-cost process which produces microcrystals that are separated out using hydrocyclones. The struvite microcystals are then formulated into dry, regular granules in combination with $(NH_4)_2SO_4$ and K salts, to give a NPK fertiliser adapted to agricultural use. Some processes (*PhosphoGreen®, Naskeo®, Crystalactor®, Rephos®, and Ostara Pearl*®) apply a controlled chemical crystallization in a **fluidized bed reactor to form struvite from the sludge liquor**. Fluidised bed reactors contain a bed of granulated struvite or fine sand, which acts as a seed material for crystal growth to facilitate the nucleation and separate crystals from the liquid phase. The process has the advantage of allowing large phosphate salt pellets to be kept in suspension in the bottom of the reactor without washing out fine crystal nuclei from the top of the reactor.

- Type II: P-precipitation upstream from the sludge dewatering unit. In this configuration, phosphate salts are precipitated upstream to the dewatering unit, providing the benefit that the sludge volume and dry matter content is reduced, thus decreasing the costs for dewatering. The *AirPrex*® precipitates salts **directly from the outflow stream from the anaerobic digester**. In the process, the digested sludge is led through a cylindrical reactor, with an inner cylindrical zone mixed by air upflow and a settling zone between this inner cylinder and the outer cylinder. Internal sludge recycling allows the crystals to grow, until they reach a size at which they can escape from the recycle flow and settle (Desmidt et al., 2015). The recovered phosphate is crystallised within the wet sludge and can therefore show some organic and inorganic impurities. Washing and gentle drying of the mineral crystals improves the quality and provides a marketable fertiliser product (Ewert et al., 2014). The typical P-recovery from such processes is currently around 10-20%. Hydrothermal hydrolysis at typical temperatures between 150-200°C or other pre-treatments can be applied on the raw sludges to increase the fraction of phosphates in solution and the methane production potential of the sludges, and thus further increasing the potential for P and energy recovery from the sludges (e.g. Pondus, Eliquo-Stulz, TerraNova Ultra). Also the Ostara's Pearl process can be combined with the Waste Activated Sludge STRIPping process WASSTRIP (Baur, 2009). Here, the raw sludges (primary, secondary and/or tertiary) are sent to the WASSTRIP reactor where P and Mg are released (stripped) by the micro-organisms as a consequence of endogenous respiration and fermentation. A subsequent solid-liquid separation process separates a Mg and P-rich stream that is sent directly to the struvite reactor, thus by-passing the anaerobic digester and dewatering unit. Hence, only the C-rich solid fraction from the material leaving the WASSTRIP reactor is further processed in the anaerobic digester and dewatered. The advantage of this process that those units have to process a smaller sludge volume, significantly reducing the capital and operating costs for anaerobic digestion and sludge dewatering. Finally, the CalPrex™ process includes an acid phase digester to provide a low-oxygen and low pH environment to facilitate the rapid release of orthophosphates in EBPR sludge. Also, the Struvex® process relies on pre-treatment techniques installed prior to the dewatering unit, possibly in combination with hydrolysis to recover P from EBPR and ChemP sludges. If such pre-treatment units are placed upstream from the dewatering unit, phosphate salts can be precipitated as struvites or calcium phosphates from the digested sludge and significant reductions in sludge volumes and P-recovery rates (up to 50%) can be achieved.

- Type III: P-precipitation downstream from the sludge after dewatering. A **wet-chemical extraction process** to process **digested sludge from waste water treatment plants.** These processes can use sludges produced in biological (EBPR) and chemical (precipitation with metal salts) waste water treatment processes, although the chemical and energy demand varies for both types of sludges. In contrast to the processes of type II, these processes enable higher P recovery efficiencies recovery by transferring (dissolving) P fixed in the solid sludge phase into the aqueous phase.
  - In the *Seaborne*® process (or Gifhorn process), nutrients are separated from the sewage sludge using a **wet-chemical process** and processed into a marketed fertiliser containing acceptable levels of heavy metals or organic pollutants (Muller et al., 2005; Desmidt et al., 2015). In the first process step, an **acidification** of the sludge occurs by the addition of H_2SO_4 in order to dissolve the solids and to release heavy metals and nutrients. In case the sludge input material originates from a plant that uses chemical precipitation, an additional precipitation step between extraction and dewatering by addition of Na_2S is introduced in order to avoid that Fe phosphate compounds with low plant availability are transferred to the nutrient product (Muller et al., 2005; Desmidt et al., 2015). The remaining solids are separated from the flow by using a centrifuge and filter system, and are then dried and directed to the sludge incineration. In the next treatment step, sulphuric digester gas is used to precipitate heavy metals from the effluent liquor. In the following process step the nutrients are recycled. Phosphate is precipitated majorly as struvite by the addition of NaOH, to obtain an alkaline pH-value, and MgO as precipitant, but significant amounts of calcium phosphates can be formed likewise. Finally, the surplus N is recovered by air stripping of ammonia. Around

90% of the nutrients (P, N) could be recovered by the Seaborne process, the P as struvite, the N for just under a third in struvite and the remainder in $(NH_4)_2SO_4$ (Günther et al., 2007).

- The ***Stuttgarter®* process** relies on the same principle of wet chemical treatment but differs from the Seaborne process by the fact that it uses a **chamber-filter-press for solid-liquid separation**, and that complexation of heavy metal ions to avoid co-precipitation is achieved by dosing of citric acid (Ewert et al., 2014). Here, the recovery product is mainly struvite (ca. 95%) (Ewert et al., 2014).
- In the **ExtraPhos** (**Budenheim®) process** the sewage sludge/water suspension is aerated with carbon dioxide under pressure of approx. 10 bar. With this treatment, carbon dioxide becomes carbonic acid in the sewage liquor, the pH decreases to a value of between 4.5 and 5.5 and a part of the phosphates bound to the sewage sludge matrix is dissolved. In the following solid/liquid separation, the sewage sludge particles are separated from the liquid phase using Ca-based coagulants. The end material recovered is dicalcium phosphate.
- The **TerraNova® process** applies a thermal hydrolysis treatment on downstream EBPR or ChemP sludges in order to achieve the leaching of the phosphates from the sludge, after which these are precipitated using Ca ions. The process is at present in a pilot stage.

o <u>Type IV: upstream sludge hydrolysis.</u> Instead of releasing phosphorus by acidification from digested sludge by applying chemicals, **thermal hydrolysis** at temperatures between 150-200°C could be applied on secondary sludge from waste water treatment plants applying **EBPR.** This releases P to a soluble form so that higher recovery rates can be achieved by phosphate precipitation, and also improves CH_4 production. The digested sludge firstly is thickened and then subjected to hydrolysis and digestion. Digested sludge has a very high water absorbing capacity, greatly reducing the degree of dewatering in proportion to its share. Through thermal hydrolysis in a pre-treatment step, poorly degradable substrates such as proteins and polysaccharides are modified such that micro-organisms can easily degrade them. Hence, by deploying this procedure prior to primary sedimentation, the good degradability and dewaterability rates of the sludge liquor is increased (Ewert et al., 2014). Processes include thermal hydrolysis (e.g. *Cambi*, *Eliquo Stulz (LysoGest)*, *Exelys*) or thermo-chemical hydrolysis (e.g. *Pondus*). Thermal hydrolysis of the digested sludge upstream of primary sedimentation is then followed by P recovery from the fully digested sludge (e.g. AirPrex® procedure) (Ewert et al., 2014).

There are specific processes (PASCH®, P-bac (INOCRE)®) that produce **struvite from ashes as input material**. These products will be described in section 15.2 (thermal oxidation materials & derivates).

15.1.2 Precipitation from other input materials

Most of the **techniques** that recover P in the form of phosphate salts (struvite, dicalcium phosphates, or a mixture of Ca- and Mg-salts) are developed for **municipal wastewater** (Desmidt et al., 2015). P-recovery techniques based on precipitation techniques can apply in principle to all phosphate rich liquids or slurries. Therefore, the techniques can also be applied on phosphate-rich industrial waste water (e.g. potato industry, dairy industry, type I processes) and anaerobically digested biowaste and manure fractions (mostly type I applications). Also other organic C-rich materials such as manure and sludges from the food industry could be subject to P-recovery techniques that increase the recovery efficiency (type III and IV processes).

15.1.3 Deliberateness of the nutrient recovery

The production processes can be **specifically developed** for the P-removal through the precipitation of Ca- or Mg-phosphate from phosphate rich waste water streams (often from the food processing industry). Mostly, the P-recovery installation is **an integral part of a larger installation** as often pre-treatment is required (e.g. EBPR, anaerobic digestion).

The P-precipitation process may provide important **benefits for the simplicity of operation of biological waste water treatment plants** and associated economic returns, even without retailing the recovered phosphate salt as a fertiliser.

- Phosphate salt producing processes of Type II and IV may **increase the dewaterability** of the sludge, and thus the associated costs of sludge disposal and chemical demand associated to traditional sludge dewatering options (e.g. addition of flocculation agents, acid and alkaline, etc.). At present, costs for sludge treatment and sludge disposal account for around 29% of the total expenses of the whole wastewater treatment processes, of which sludge dewatering accounts for 16% (STRUBIAS sub-group comments). The divalent cation bridging theory states that flocculation, which is strongly linked to dewaterability, is driven by the ratio of divalent cation concentrations (Ca^{2+}, Mg^{2+}) over monovalent cations (Na^+, K^+, NH_4^+, etc.). Divalent cation creates bridges between particles whereas monovalent cations tend to deteriorate flock structures. Therefore, an improved dewaterability can be expected if the addition of magnesium divalent cations surpasses the effect of sodium hydroxide dosing. Marchi et al. (2015) indicated the importance of a proper tuning of chemical additions in order to achieve progressive dewatering.
- Waste water treatment costs are also reduced by the lower maintenance costs due to the **avoided pipe clogging and abrasion of centrifuges**.
- The **reduction of the P and N load of the sludge liquor** has a direct effect on the treatment capacity of the whole waste water treatment plant as well as a cost

factor, since the removal of nutrients from the wastewater requires energy, chemicals and tank volume (Ewert et al., 2014).

The presence of some other species present in the stream or the purposeful addition of specific chemicals may cause the (co-)formation of materials other than struvite (K-struvite, calcium hydroxyl apatite, vivianite, etc.).

15.2 Thermal oxidation materials & derivates

Whether ashes as obtained after the thermal oxidation processes can be suitable for direct use as a fertilising material is dependent on (1) the elemental composition of the ashes, (2) the presence of metals and metalloids in the input materials, and (3) the availability of the plant nutrients present in the ashes.

15.2.1 Raw ashes and melting/sintering materials

15.2.1.1 Thermal oxidation technology

Ashes obtained from the combustion of organic materials (e.g. wood residues, poultry manure, meat and bone meal, animal bones, sewage sludge) are used directly as a multinutrient fertiliser and/or liming material in many different EU Member States. Available technologies for the combustion of such organic biomass include (van Eijk et al., 2012):

- **Bubbling fluidized bed boilers** (BFB) are often preferred in small-scale applications, with fuels having low heat value and high moisture content. The bed is fluidised by means of an arrangement of nozzles at the bottom of the furnace which create turbulence that enhance the mixing of the fuel, increasing the boiler's efficiency by converting unburned C remaining to usable energy. The bed is usually formed by sand and with a small amount of fuel. Solids fluidization occurs when a gaseous stream (primary air) passes through a bed of solid particles at enough velocity (above the minimum fluidization velocity) to overcome the particles gravity force. Limestone might be added to the bed to eliminate sulphur and/or chlorine. BFB operation range is between the minimum fluidisation velocity and the entrainment velocity on which the bed particles would be dragged by the passing gas, being usually 1.2 m/s at full load. Combustion temperature is typically between 800 and 950ºC, being 850ºC a usual bed temperature.
- **Circulating Fluidized Bed** (CFB) technology boilers are normally used in larger applications, being similar in basic concept to the BFB. CFB has enhanced flexibility over BFBs for firing multi-fuels with high moisture content and significantly higher efficiency up to 95%. CFB configuration includes solid separators that separate the entrained particles from the flue gas stream and recycles them to the lower furnace. The collected particles are returned to the furnace via the loop seal. The addition of the solid separators allows CFB technology to reach the higher values regarding efficiency and availability and provides fuel flexibility. The entrainment velocity is the limit point that defines the transition from a BFB to a CFB. The CFB operation range is fixed over that entrainment velocity. Beyond this velocity the bed material becomes entrained and the

solids are distributed throughout the furnace with a gradually decreasing density from the bottom to the top of the furnace. Fluidizing velocity is higher than in a BFB and can be between 4.5-6.7 m/s.

- Similar to BFB, **grate boilers** are used in units below 100 MWe and normally for industrial uses. Grate technology can burn a range of fuels wider than a BFB, but worse emissions and efficiency as BFB. Grate boiler provides very good performance burning low moisture and high alkalis content fuels. Grate can burn difficult fuels as straw, poultry litter, high alkaline agro crops that are more challenging to combust using BFB/CFB due to high agglomeration tendency.
- Organic residues can also be heated to temperatures between 800ºC and 1500ºC to achieve a transformation of solid materials through **melting** (e.g. in a rotary kiln or cupola furnace). Melting occurs in a non-oxygen limiting environment, resulting in the formation of ashes and P-slags. Due to the addition of carbonates, soda (Na_2CO_3) and quartz sand, it is possible to separate P from many other elements and to influence the crystal structure of the P containing slags (e.g. isomorphic substitution of PO_4^{3-} ionic group by $SiO2^{2-}$ or $CO3^{2-}$) affecting the reactivity of the final product and therefore the plant P availability. Metals/metalloids are partially volatilised (Zn, Cd, Hg, F), partially remain in the metal fraction (e.g. Fe, Cu, Cr, Ni) or remain in the slags (see post-processing). Therefore, this process can be applied on non-combusted organic materials or as a post-processing step on combustion ashes to improve the material quality (see section 15.2.1).

Also steelmaking processes make use of thermal oxidation melting processes in basic furnaces (converters) or electric arc furnace. Steel slag is produced as molten rock at around 1650°C during the conversion of hot metal, sponge iron or steel scrap into crude steel. It consists of the oxidised accessory elements from hot metal, steel scrap and the other metallic substances, and of the slag-forming additives such as limestone, burnt lime or dolomite. Depending on how the crude steel is produced, a distinction is made between basic oxygen furnace slag from the basic oxygen furnace process, and electric arc furnace slag from the electric arc furnace process. A **blast furnace** is a type of metallurgical furnace that relies on thermal reduction for smelting to produce industrial metals. In a blast furnace, fuel, ores, and flux (limestone) are continuously supplied through the top of the furnace, while a hot blast of air (sometimes with oxygen enrichment) is blown into the lower section of the furnace through a series of pipes called tuyeres, so that the chemical reactions take place throughout the furnace as the material moves downward. An **electric arc furnace** is a furnace that heats charged material by means of an electric arc. Modern furnaces mount, however, oxygen-fuel burners in the sidewall and use them to provide chemical energy to the cold-spots, making the heating of the steel more uniform. Additional chemical energy is provided by injecting oxygen and carbon into the furnace. In specific metallurgic treatments under development (Bartsch et al., 2014), organic residues or their ashes are heated together with slags from basic oxygen furnaces to achieve a reconfiguration of the solid materials with the intention to improve the quality (e.g. increased plant availability through the formation of silicophosphates, reduced metal content) of the resulting P-rich fertilising material (see section 15.2.2).

15.2.1.2 Flue-gas treatment systems

Flue-gas treatment (FGT) systems are constructed from a combination of individual process units that together provide an overall treatment system for the flue-gases (European Commission, 2006c). The individual components of a FGT system are combined to provide **an effective overall system for the treatment of the pollutants that are found in the flue-gases**. There are many individual components and designs, and they may be combined in many ways. The diagram below shows an example of the options and their possible combination (European Commission, 2006c). The FGT technology impacts upon the quality of the combustion residues.

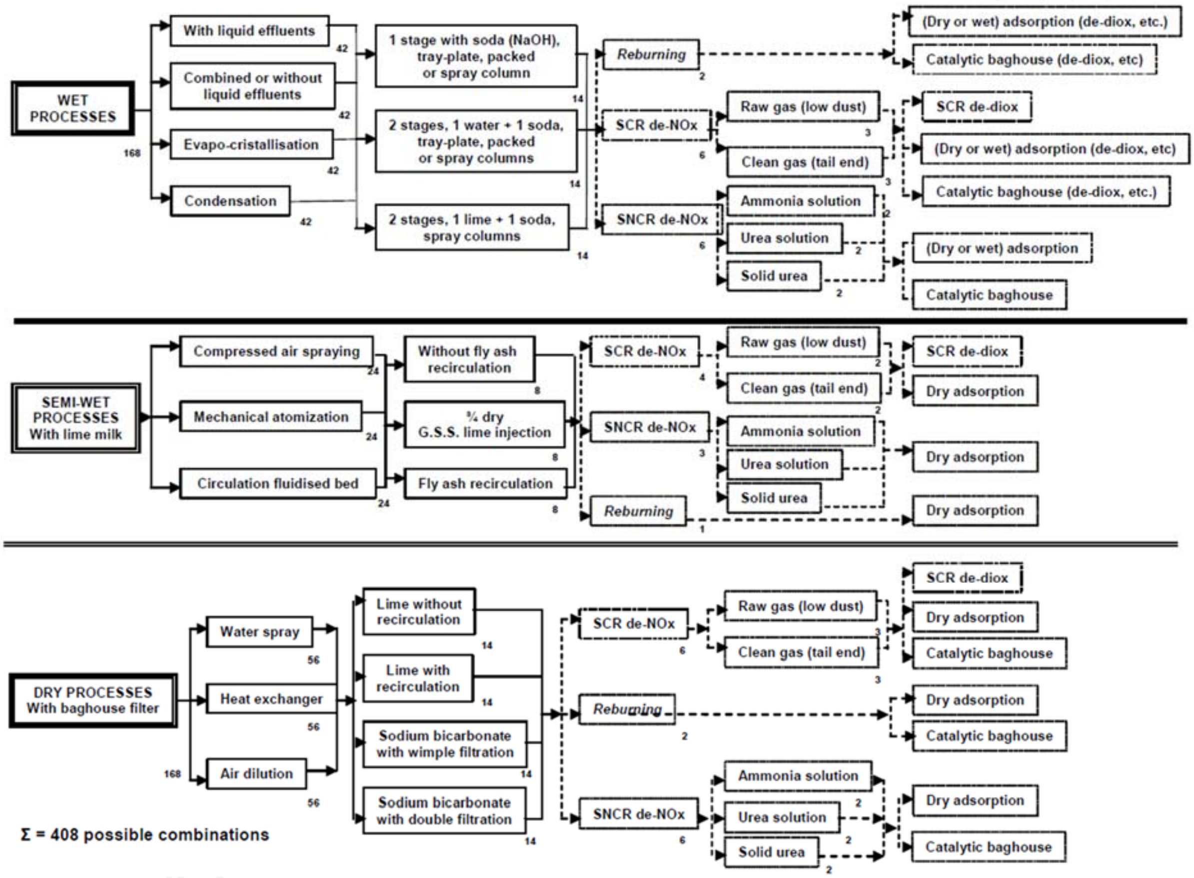


Figure 32: Overview of potential combinations of fluegas treatment systems (European Commission, 2006c)

15.2.2 Ash and slag derivates

The use of **raw ashes** in agriculture can be associated to two major issues (Chandrajith and Dissanayake, 2009; Herzel et al., 2016). At first, the conditions during thermal oxidation induce a wide range of structural modifications that can **reduce the P-solubility and plant availability**. Secondly, ashes produced of specific input materials (e.g. sewage sludge) contain **high contents of metals/metalloids**. Hence, direct fertiliser use will return these potentially toxic elements into soil, water, air, food crops, and ultimately the human body tissues via the food chain.

Ashes not suitable for direct recycling can be treated through two different routes that aim at P-recovery: (1) wet-chemical processes and (2) thermal processes (Table 27). Ashes from mono-incineration (i.e. not mixed with low-P wastes like industrial sludges, municipal solid refuse) are relevant for enhanced P-recovery strategies because of the high P content. These processes are especially relevant for P-rich ashes generated from sewage sludge, meat and bone meal, manure, etc. Phosphorus **recovery rates** from mono-incinerated ashes can reach **up to 90%** (Cornel and Schaum, 2009). For sewage sludge ashes, specific thermal as well as wet-chemical processes are able to process **ashes originating from EBPR as ChemP plants** (Kabbe et al., 2015).

- Type I: wet-chemical processes
    - An almost complete **acidic dissolution of P at pH-values below 2** through the addition of chemicals is the principle of action to transform P into a bio-available form via wet-chemical extraction techniques. This process is unavoidably accompanied by a partial dissolution of metals or their compounds. The amount of dissolved metals depends on the composition of the raw input material (Fe- or Al-rich) as well as on the type and amount of the added acid (H_2SO_4 or HCl). Some processes effectively separate and remove toxic inorganic contaminants (e.g. Pb, Cd, Hg, etc.) in order to increase the quality of the P-recovery product. Additionally, it is desirable to separate especially Al and Fe as well, as these elements can reduce the quality and plant bio-availability of the recovery product. **For the removal of cations from the acidic leachate** different approaches are technically feasible to obtain satisfactory P-removal: sequential precipitation, liquid-liquid extraction, and ion exchange (Table 27).
        - P-rich ashes of specific characteristics can replace ground phosphate rock in the **acidulation process applied by the fertiliser industry**. The addition of sulphuric acid or nitric acid will result in the production of traditional P-fertilisers (e.g. DAP, MAP, TSP, nitrophosphate, etc.). The P-rich ashes should be consistent and the Fe/Al content should be relatively low in order to enable the partial substitution of phosphate rock by ashes in the process; typically ashes make up only 10-20% of the P in the mixture with the remaining P being added as phosphate rock (Langeveld and Ten Wolde, 2013). The metal removal rates are low as the P and other compounds in the ashes are dissolved in process, and end up together in the fertilising material.
        - The basis of the **SEPHOS process is** the **sequential precipitation** of P complexes with an alkaline treatment (Takahashi et al., 2001; Schaum, 2007). The separation of dissolved P from heavy metals is achieved by raising the pH-value in the acidic leachate to induce the precipitation of Al-P while most heavy metals remain in solution (Takahashi et al., 2001).

The heavy metal content of the Al-P product is then further decreased by precipitating heavy metals with sulphide (Schaum, 2007). Since the entire P has to be precipitated as Al-P, this process is especially suitable for Al-rich ashes coming from waste water treatment plants that employ chemical P-removal by addition of Al-salts. Since Al-P cannot be directly reused as fertiliser, the precipitated Al-P may be dissolved by alkaline treatment followed by precipitation as Ca-P. Altogether, this type of wet chemical P-recovery process results in a total chemical demand (at least 600 g H_2SO_4 /kg ash and 300 g NaOH/kg ash) (Schaum, 2007). A P-recovery rate of 90% is documented for the Sephos process. The **SESAL-Phos process** (Petzet et al., 2012) applies a softer acidification treatment (to a pH value of around 3 through HCl addition), followed by direct alkaline dissolution of P. In this case, only the low amounts of P dissolve, while most (heavy) metals remain in the ash. In a following process step, the dissolved P can be precipitated from the alkaline solution (pH > 13) as Ca-P with a very low impurity level, via the addition $CaCl_2$. The amount of Al-P directly leachable via alkaline treatment depends on both the Al content and the Ca content of the ashes (Schaum, 2007). In case of sewage sludge ash with very low Ca contents, a significant amount of P can be dissolved with low chemical demand, for which the process is more suitable for soft waters. Consequently, the SESAL-Phos process leads to a significantly reduced specific chemical demand, but the recovery rate of 74-78% is lower for the SEPHOS process (Petzet et al., 2012).

The **Leachphos®** process is another **sequential process** with a leaching and a precipitation step to treat fly ash, amongst other from municipal solid waste incineration (Adam et al., 2015). The first step is leaching of sewage sludge ash with dilute H_2SO_4 that dissolves about 70-90 % of the P in the ashes, depending on acid concentration and reaction time. The leaching is followed by a solid/liquid separation step carried out on a vacuum belt filter or in a filter press. The leached sewage sludge ash filter cake is withdrawn from the process and must be disposed. The P containing liquid is pumped into a second stirred reactor, where dissolved P is precipitated by dosing of lime (CaO) or caustic soda (NaOH). A product with relatively high P content (13% P), considerable metal depletion and sufficient dewaterability is thereby produced (Adam et al., 2015). Depending on the precipitation agent, P is present in different mineral phases. If precipitated mainly with lime, P is present in the form of **calcium phosphate next to aluminium phosphate**. After precipitation and separation of the phosphorus product, the liquid waste stream requires additional treatment. Treatment consists of pH elevation to a pH of 9 by dosing of additional lime and of sulphidic precipitation of metals by an organosulphide precipitation agent (Adam et al., 2015). This is carried out in a third reactor followed by an additional solid/liquid separation step by a filter press. Thus the metals in the wastewater are removed almost completely. Thereafter, the pH in the waste water is adjusted to a pH of 7 and is discharged either to a waste water treatment plant or directly to a receiving water body (Adam et al., 2015).

- The **PASCH®** (Phosphorus recovery from Ash, developed at Aachen University) process utilizes **liquid-liquid extraction** for heavy metal and iron separation (Nieminen, 2010; Pinnekamp et al., 2010). Different acids were tested for P-dissolution by Montag and Pinnekamp (2009) with results of 25%, 50%, 80% and 90%, for NaOH, H_3PO_3, H_2SO_4 and HCl respectively. After the acid leaching, a lamella separator and filter separate the residue. The filtrate, containing phosphorus, calcium, and metal compounds, is treated in the extraction step with Alamine 336 and tributylphosphate (TBP). Reduction in the heavy metal concentrations is over 95% and iron over 99%. The final step precipitates the phosphate as calcium phosphate or struvite depending on precipitation chemical (i.e. lime or magnesium compounds).

- The **BioCon® process** recovers P as H_3PO_4 from sewage sludge ashes (Balmér et al., 2002; Nieminen, 2010). The entire process consists of three phases: sludge drying, sludge incineration, and recovery unit with **ion exchangers**. The first step of the recovery process dissolves the P and heavy metal contents with H_2SO_4 at a pH value of 1 (Berg and Schaum, 2005; Herrman, 2009). The solution passes through a series of ion exchangers. The first exchanger is cationic, separating Fe^{3+} ions. It is regenerated with HCl producing $FeCl_3$. The following exchanger is anionic, collecting K^+ ions, and after regeneration with H_2SO_4 produces $KHSO_4$. In this step, NaOH is used for both pH adjustment and regeneration (Hultman et al., 2001). The final exchanger collects phosphates. Regeneration with HCl produces a stream of H_3PO_4 (Lundin et al., 2004). It should be emphasized that H_3PO_4 has no soil fertilising properties on its own; it is an intermediate in the production process of mineral P-fertilisers.

- Additionally, there are different patented multi-modular approaches of which the process detail are kept confidential. The **EcoPhos®** process is already implemented at full-scale and relies on a multi-step approach to valorise low grade phosphate rock and also P-rich ashes to high quality market products as H_3PO_4 or dicalcium phosphate (DCP) (EcoPhos, 2016). First step is the leaching of the ash with HCl. Undergoing different modules (including ion exchange resins) which are kept confidential, a purified H_3PO_4 for fertiliser or food and feed industry is produced. At the same time most of the produced residues are sellable products as $CaCl_2$, gypsum, silicate as well as iron- and aluminum chlorides. With the **TetraPhos®** process, Remondis developed and implemented a similar approach in pilot scale (Hamburg) using H_3PO_4 instead of HCl to the leach the ash, and ending up with H_3PO_4 as a final product (Remondis Aqua, 2016). A full-scale operational TetraPhos plant at the Hamburg waste water treatment plant will be opened in 2019. Also in the acid leaching **RecoPhos®** process (Weigand et al., 2013; RecoPhos, 2016), the plant-available phosphate fraction is increased by reacting the sewage sludge ash with H_3PO_4. Thereby, the primary minerals are transformed into soluble calcium and magnesium dihydrogen phosphate, the primary nutrient components of the RecoPhos P 38 fertiliser. The piloting **Edask process** relies on semi-permeable membranes, under the influence of an electric potential, to separate phosphate ions (electrodialysis) (Thornberg, 2015). The P-recovery end product is H_3PO_4. The **EasyMining Ash2Phos** relies on similar principles, including the acidulation of the ashes, followed by the separation of elements and removal of metals. Also here, the process produces various added value materials, including P-fertilisers, ferric and aluminium hydroxide. The process consists of several successive chemical reactions undertaken in room temperature, and can also use sludge ashes with lower P content than from mono-incineration. The heavy metals are separated as sulfide precipitate for disposal which will be disposed. **Phos4life** is a process for P recovery process from sewage sludge incineration ash, for which development is led by Zurich Kanton (AWEL) and ZAR (Zentrum für nachhaltige Abfall- und Ressourcennutzung. The process developed together with Técnicas Reunidas and successfully tested under micro-pilot plant operation in Madrid uses sulphuric acid (at 96%) to solubilise P and other elements in the ash, then hydrochloric acid and solvent extraction to separate phosphorus acid from iron chloride solution and heavy metals. The full scale process is planned to treat 30 000 t/year of sewage sludge incineration ashes, to produce 11 000 t/year of 74% phosphoric acid (after concentration using steam), 34 000 t/y of 40% iron chloride solution for recycling as coagulant agent in waste water treatment plants and 42 000 t/y of a residue which can be used by the cement industry. Heavy metal contaminants are nearly completely (>85%) transferred to a metal concentrate for metal recycling. The initial test results show the following recovering rates of the total potential in the SSIA: Phosphorus >95% (as H_3PO_4); iron: >90% (as iron-chloride solution).

- The **P-bac process®** offers selective recovery of P from sludge ash, bed ash and contaminated soils via a biotechnological route (Inocre Biotech, 2016). The P-bac process combines efficient and selective phosphate recovery with **bioleaching** and therefore is especially suitable for solids with high content of heavy metals. The selective recovery of phosphate with the P-bac process is realised in two phases. The first step is based on the “bioleaching principle”, which is applied worldwide for the exploitation of metals (e.g. Cu, Zn, U, etc.) in the mining industry. By

microbial generation of H_2SO_4, phosphate derivates and metals/metalloids are dissolved within few hours. The remaining solid matter is separated from the liquid matter and can be disposed for reduced costs. The phosphate-enriched biomass subsequently is separated from the liquid phase and can be precipitated as struvite after anaerobic dissolution. Up to 90% of the original phosphate can be recovered with the P-bac process.

- Type II: Thermal processes
  - Nutrients can be recovered from ashes by high temperature treatments (Table 27). Processes were developed that transfer P into a metallurgical slag by reductive smelting at very high temperature temperatures in a shaft furnace (Scheidig, 2009) or that reduce P to elemental P that is separated via the gas phase in an inductively heated shaft furnace (Schönberg et al., 2014). The general principle is that volatile heavy metals such as Zn, Pb, Cd and Hg are separated from the product via the gas phase and further collected in the flue dust, and heavy metals with high boiling points such as Fe, Cu, Ni and Cr are separated in the form of a liquid alloy.
    - The **Mephrec** (Metallurgical Phosphorus Recovery) process was developed by the German company Ingitec. The process recovers P and energy from sludge and many other input materials of high calorific value such as meat and bone meal and/or wood ash. **Dried sludge** is briquetted with slag forming substances and coke. The mixture is treated in 2000°C transferring P into the mineral slag and heavy metals to liquid metal phase (Fe, Cu, Cr, Ni) or to gaseous phase (Hg, Cd, Pb, Zn). The silico phosphates containing slag is separated from metal phase after being tapped at 1450°C (Adam 2009). The final product contains, depending on the input materials used, 5-10% P with over 90% citric acid solubility. The P content can be varied by mixing sewage sludge with animal meal. The energy recovery from high-calorific raw off-gas can be realized either by directly combusting and using the heat in an Organic Rankine Cycle (ORC) process, or multi-stage gas cleaning and use in a combined heat and power (CHP) plant (Adam et al., 2015). With **sewage sludge ash**, the P content can reach up to 9%, but energy recovery is not possible.
    - The **FEhS/Salzgitter process** is a process to increase the P-content of liquid steel slag with phosphorus by blending it with ashes from the incineration of P-containing materials like sewage sludge and/or meat and bone meal. For the process, the slag is separated from the metal bath and transferred to an external slag pot, in which the cold ash is blown into the melt from the top together with oxygen or air. In the liquid slag the ash's phosphates are dissolved. After cooling and solidification, P_2O_5 is converted to plant-available Ca-Si-phosphate, similar to the phosphate in Thomas ground basic slag. The oxidation of residual metallic iron and bivalent iron in the steel slag produces the energy to maintain the

    necessary process temperatures of around 1500°C. The process has been tested in laboratory and in industrial scale.

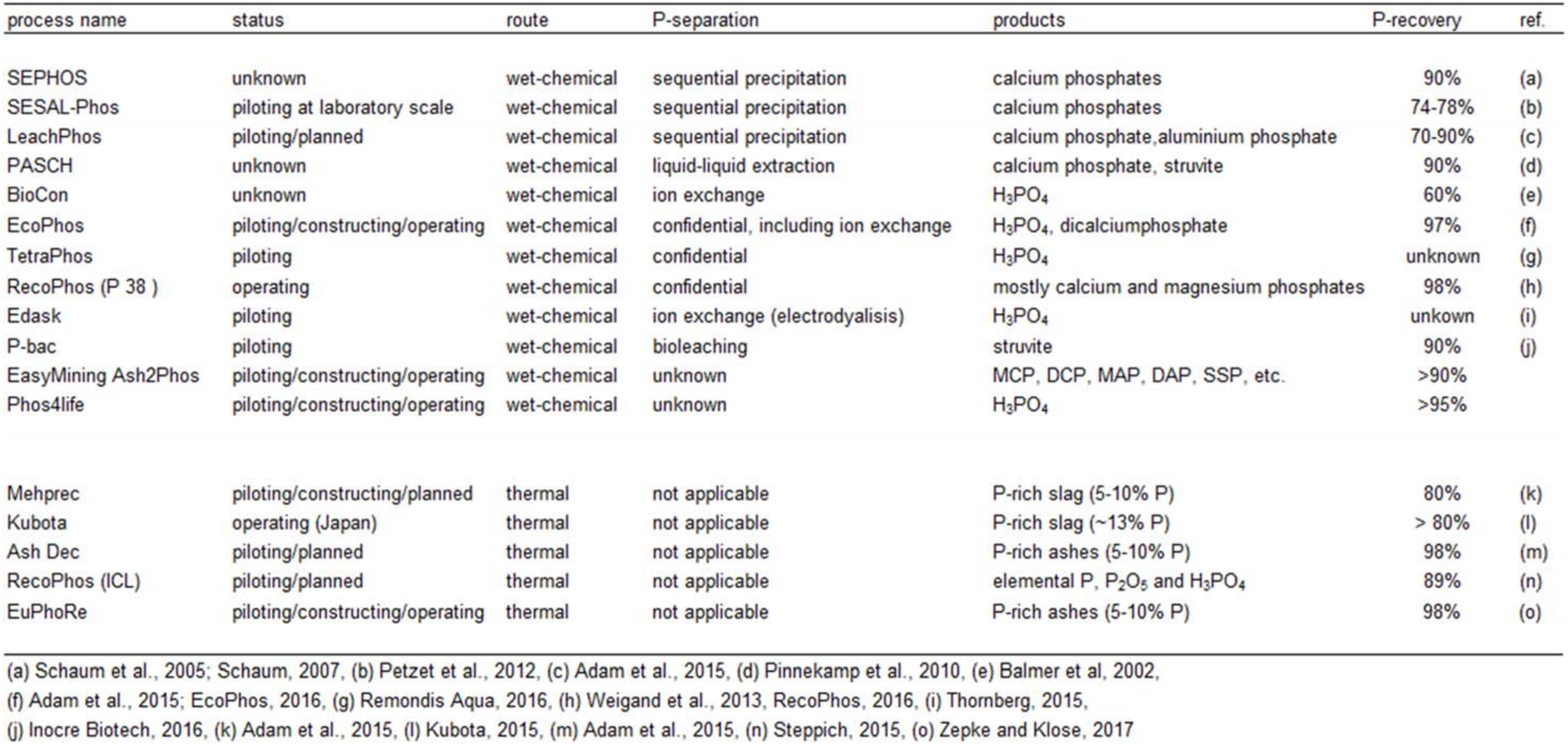
    - The melting process of the Japanese company **Kubota** has been developed for municipal solid waste, sewage sludge, landfill waste and ashes thereof. It enables the separation of P-slag through a temperature treatment of 1250~1350°C. P is immobilized in the slag (~13% P) with a recovery rate of > 80% (Kubota, 2015).

- The **ASH DEC** process (OutoTec) treats mono-incinerated sewage sludge ashes by a **sodium sulphate dosage and thermal treatment** below the melting point of sewage sludge ash in order to remove heavy metals making the product suitable for agricultural use by increasing the plant availability of P. An ASH DEC plant could stand alone and being operated as greenfield facility. For economic and ecological reasons it is planned to **combine the ASH DEC plant with mono-incineration**. The main advantage of the combination is the possibility of feeding hot ash directly from the mono-incineration plant to the ASH DEC facility, thus saving energy and equipment (Adam et al., 2015). In the first step, the ash is mixed with Na_2SO_4 in the thermally treated ash. Fresh Na_2SO_4 input could be partly replaced by recycled Na_2SO_4 from the mono-incineration. Alternatively (older process), $MgCl_2$ can be used for higher removal rates of heavy metals in the process, but this pathway results in reduced plant availability of the recovered phosphates. The dried sewage sludge (> 80 % dry matter) is charged in granules and is used as reducing agent in the ASH DEC process for the reduction of sulphate in the Na_2SO_4 and metal compounds. The thermal reaction is performed in a directly heated rotary kiln in counter flow having maximum temperatures of 900-950°C. At this temperature, metals/metalloids react with the salts, become gaseous, and evaporate. After cooling, the P-rich ashes (P content of about 5-10%) are in the form of small granules and may be finished on site or in cooperation with a customer at the site to further increase agronomic values.

- The **EuPhoRe®-Process** begins with application of additives into the dewatered or dried sewage sludge (Zepke and Klose, 2017). The following energy utilisation is characterised by volatile components degassing during a reduction period under application of medium temperature pyrolysis at 650 to 750 °C and is linked to an immediate subsequent post-combustion of the remaining fixed carbon at temperatures between 900 and 1.100 °C. During the first reductive process step, the heavy metal compounds contained in the sewage sludge are already partly transformed into the gas phase, although it is continued throughout the oxidative second process step, the carbon post-combustion. The metal compounds are being efficiency released and significantly improved through additive compounds of alkaline and/or earth alkali salts, such as $MgCl_2$. Magnesium remains into the phosphate-fertiliser and improves the plant availability. Chlorides take the reduced heavy metals into the gas stream. The generated phosphate fertiliser contains low-carbon (2 – 6% C) and the heavy metal compounds are to a large extent depleted (up to > 98%). The fertiliser contains nearly the entire phosphate load of the input material and after grinding as well as dust collection consequently allows for a direct agricultural utilisation. The energy content of dewatered sewage sludge (>25% dried matter) is sufficient for a complete thermic, self-sustaining production facility operation including the required drying process.

- The **thermo-reductive RecoPhos** is a thermo-chemical process involving the fractioned extraction of P and heavy metals from sewage sludge, meat and bone meal and sewage sludge ashes at high temperatures under reducing conditions (Steppich, 2015). Thermal process uses **electro-magnetically induced heating** of a reactor bed consisting of coke or graphite. The induction heating systems serve alternating magnetic fields with high energy density and thus provide the reaction conditions required for the molten ash to react with the C. The reductive processes taking place within the reactor are based on the Whoëler reaction at a temperature of 1200-1400°C, which is the same chemical principle as the one used in the well-established submerged arc furnace, producing high grade elemental P as vapour, which can be either condensed and harvested as P4 or subsequently oxidised to P_2O_5 or converted into H_3PO_4. The process enables the use of waste materials as heat sources, reducing agents or additives, including dried sewage sludge, foundry ash, waste salts or meat and bone meal, and low grade phosphate rock, with as advantage that in the RecoPhos process no pre-agglomeration of powder feedstock is needed and no dioxin emissions takes place. The Recophos process also claims to be able to recover P from raw materials containing significant levels of Fe from ChemP waste water treatment plants. The end product P4 can then be used for production of flame retardants or lubrication additives while H_3PO_4 can then be further used for the production of inorganic P-fertilisers and other P-containing products. The thermo-reductive RecoPhos technology has been acquired by **ICL** (Israel Chemicals Ltd) for the industrial scale production of P-fertilisers derived from secondary raw materials.

Table 27: Overview of the principles and properties of P-recovery processes for thermal oxidation materials & derivates.



process name	status	route	P-separation	products	P-recovery	ref.
SEPHOS	unknown	wet-chemical	sequential precipitation	calcium phosphates	90%	(a)
SESAL-Phos	piloting at laboratory scale	wet-chemical	sequential precipitation	calcium phosphates	74-78%	(b)
LeachPhos	piloting/planned	wet-chemical	sequential precipitation	calcium phosphate,aluminium phosphate	70-90%	(c)
PASCH	unknown	wet-chemical	liquid-liquid extraction	calcium phosphate, struvite	90%	(d)
BioCon	unknown	wet-chemical	ion exchange	H_3PO_4	60%	(e)
EcoPhos	piloting/constructing/operating	wet-chemical	confidential, including ion exchange	H_3PO_4, dicalciumphosphate	97%	(f)
TetraPhos	piloting	wet-chemical	confidential	H_3PO_4	unknown	(g)
RecoPhos (P 38)	operating	wet-chemical	confidential	mostly calcium and magnesium phosphates	98%	(h)
Edask	piloting	wet-chemical	ion exchange (electrodyalisis)	H_3PO_4	unkown	(i)
P-bac	piloting	wet-chemical	bioleaching	struvite	90%	(j)
EasyMining Ash2Phos	piloting/constructing/operating	wet-chemical	unknown	MCP, DCP, MAP, DAP, SSP, etc.	>90%	
Phos4life	piloting/constructing/operating	wet-chemical	unknown	H_3PO_4	>95%	
Mehprec	piloting/constructing/planned	thermal	not applicable	P-rich slag (5-10% P)	80%	(k)
Kubota	operating (Japan)	thermal	not applicable	P-rich slag (~13% P)	> 80%	(l)
Ash Dec	piloting/planned	thermal	not applicable	P-rich ashes (5-10% P)	98%	(m)
RecoPhos (ICL)	piloting/planned	thermal	not applicable	elemental P, P_2O_5 and H_3PO_4	89%	(n)
EuPhoRe	piloting/constructing/operating	thermal	not applicable	P-rich ashes (5-10% P)	98%	(o)

(a) Schaum et al., 2005; Schaum, 2007, (b) Petzet et al., 2012, (c) Adam et al., 2015, (d) Pinnekamp et al., 2010, (e) Balmer et al, 2002, (f) Adam et al., 2015; EcoPhos, 2016, (g) Remondis Aqua, 2016, (h) Weigand et al., 2013, RecoPhos, 2016, (i) Thornberg, 2015, (j) Inocre Biotech, 2016, (k) Adam et al., 2015, (l) Kubota, 2015, (m) Adam et al., 2015, (n) Steppich, 2015, (o) Zepke and Klose, 2017

15.2.3 Deliberateness of the nutrient recovery

Thermal oxidation processes may be performed for sanitisation and volume reduction of waste-based materials. Synergies with energy recovery are possible for organic materials. The chemical or thermal post-incineration manufacturing processes target the **specific aim of P recovery and/or to improve material properties**. Finally, P-slags from the steel industry are produced as a residue from the steelmaking industry.

15.3 Pyrolysis & gasification materials

15.3.1 Pyrolysis spectrum production techniques

Pyrolysis spectrum techniques take place in an oxygen-deficit environment or with a controlled amount of oxygen and/or steam that limit the chemical reactions that transform input materials into chars. The extent to which pyrolysis & gasification materials burn depends on the ratio between the number of moles of oxygen admitted in the reactor and the moles of oxygen required for complete combustion. The less oxygen present in the reactor, the more solid pyrolysis material is produced. There are several available thermochemical technologies that operate in an oxygen-limited environment:

- **Hydrothermal carbonisation** involves treatment in a closed system at moderate temperatures (~ 180°C - 300°C) and a pressure of approximately 10-30 bar over an aqueous solution of biomass for several hours (2h-10h), resulting in the production of char-type like materials as residues.
- **The pyrolysis** process produces three different products that depend on the technology used, namely **biochar** (solid), **syngas** (non-condensable gases), and **bio-oil** (condensable liquid residue). The thermochemical decomposition of the organic materials takes place by heating in an oxygen-deficient environment at moderate to high temperatures (~ 300°C - 700°C). Pyrolysis systems use kilns or retorts, and exclude oxygen while allowing the pyrolysis gases, or "syngas" to escape and be captured for combustion.
- **Gasification** is a process that converts organic feedstocks into carbon monoxide, hydrogen and carbon dioxide. The material is treated at high temperatures (> 700°C), with a controlled amount of oxygen and/or steam. Gasification generally produces less solid materials compared to pyrolysis, because some oxygen is intentionally introduced in the system.

For simplicity, pyrolysis & gasification materials will refer to all three techniques as the critical factor for this CMC is that the chemical reactions that transform input materials are limited by the amount of oxygen and/or steam.

There are a number of different reactor configurations that can achieve this including ablative systems, fluidised beds, stirred or moving beds and vacuum pyrolysis systems. For a more detailed description, it is referred to Venderbosch and Prins (2010).

Pyrolysis can be an **endothermic or exothermic reaction** depending on the reactor temperature and the moisture content of the input materials, becoming increasingly exothermic as the reaction temperature decreases (Mok and Antal, 1983). The exothermicity of the slow pyrolysis reaction per unit of biochar yield is reported to range from 2.0 to 3.2 kJ g^{-1} biochar (Mok and Antal, 1983; Milosavljevic et al., 1996).

15.3.2 Spectrum of pyrolysis & gasification materials

Materials produced by pyrolysis spectrum techniques largely reflect the elemental composition of the input material that was used for the process. The **organic carbon content of pyrolysed chars fluctuates between 5% and 95%** of the dry mass, dependent on the feedstock and process temperature used. Some pyrolysis & gasification materials made of plant-based materials often have a high organic C content, but low nutrient content. An important defining feature of these materials is a certain level of organic C forms, called fused aromatic ring structures that relate to many of the soil improving properties ascribed to the material. Such materials are typically defined as biochar, and have organic C contents > 50%. Pyrolysis materials derived from mineral-rich input materials (e.g. manure, animal bones) are much lower in organic C. Therefore, the European Biochar Certificate refers to pyrolysed organic matter with a C content lower than 50% as pyrogenic carbonaceous materials, instead of biochar. In the STRUBIAS framework, the name pyrolysis & gasification materials has been proposed as a common name for all material produced in an oxygen-limiting environment, although a distinction has been made between C-rich (e.g. woody biomass) and nutrient-rich pyrolysis & gasification materials.

15.3.3 Deliberateness of the nutrient recovery

Pyrolysis processes are mostly performed with the specific aim of producing a high-value product with a set of specific properties that relate to its function (e.g. soil improver, P-fertiliser). Pyrolysis can also be performed in order to enable a weight reduction of the input materials, facilitating its further handling, transport, and distribution. Also synergies with energy recovery is possible for organic materials, but the energy recovery potential in pyrolysis plants is largely dependent on the moisture content of the input materials.

Pyrolysis & gasification materials can also be produced for objectives other than nutrient recovery. Biochar application to soil is described as a **climate change mitigation strategy** (Woolf et al., 2010). The transformation of labile to recalcitrant C compounds in the biochar production process has been suggested as a means of abating climate change. Apart from its application to the soil, **biochar can also be used for other applications** (Schmidt and Wilson, 2016):

- The cascaded use of biochar in animal farming (silage agent, slurry treatment, feed additive);

- Use as a soil conditioner (carbon amendment, compost additive, plant protection);
- Use in the building sector (insulation, air decontamination, humidity regulation);
- The treatment of waste water (active carbon filter, pre-rinsing additive);
- The treatment of drinking water (micro filters);
- Other uses (exhaust filters, carbon fibers, semiconductors, etc.).

16 Chemical composition of STRUBIAS materials

16.1 Precipitated phosphate salts & derivates

16.1.1 Macroelements

Table 28: Macronutrient and organic C content precipitated phosphate salts from different input materials.

input material	dry matter content	P	N	Mg	Ca	K	organic C	Reference
	(%, dried at 105°C)			(%, dried at 40°C)				
urban wastewater -Pearl		13.4	5.7					(Kraus and Seis, 2015)
urban wastewater - Airpex		11.7	4.7					(Kraus and Seis, 2015)
urban wastewater - Stuttgart		10.1	5					(Kraus and Seis, 2015)
urban wastewater - Gifhorn								(Kraus and Seis, 2015)
urban wastewater	58.3	10.0	2.1	7.3	6.7		3.7	(STOWA, 2015)
urban wastewater	77.3	7.5	3.0	5.9	4.3		0.3	(STOWA, 2015)
urban wastewater	52.1	12.1	2.0	8.6	1.6		0.3	(STOWA, 2015)
urban wastewater	52.8	13.5	5.5	10.8	1.0		0.3	(STOWA, 2015)
urban wastewater		12.9	5.7	9.8				(Ueno and Fujii, 2001)
urban wastewater		12.4	5.1	9.1				(Münch and Barr, 2001)
urban wastewater		12.1	4.2	9.3	0.49	0.06		(Vogel et al., 2015)
urban wastewater	69.7	9.5	6.0	8.3	0.5	0.14		STRUBIAS - confidential data provider
urban wastewater	61.8	10.5	4.7	7.7	5.6	0.47		STRUBIAS - confidential data provider
urban wastewater	57.1	11.0	5.0	9.5			6.2	(ADEME - Naskeo Rittmo Timab, 2016)
urban wastewater	61.2	10.9	4.7				6.2	(ADEME - Naskeo Rittmo Timab, 2016)
manure (Stichting Mestverwerking Gelderland)		5.9	0.8	8.0	1.5	4.8	3.2	(Ehlert et al., 2016a)
manure		10.1	5.8	6.4		3.7		(Katanda et al., 2016)
urban wastewater		10.4	4.4	13.1	1.2	0.08		(Plaza et al., 2007)
dairy industry		11.3	3.3	8.64		0.73		(Uysal and Kuru, 2015)
dairy industry		12.4		4.2	17.9			(Massey et al., 2009)
potato industry	92	9.4	5.2				4.8	(Sigurnjak et al., 2016)

potato industry - Nuresys	56.1	12.8	5.1	9.7	0.25	(Vanhoof and Tirez, 2014)
food processing (vegetable oil) - Nuresys	55.9	12.6	5.2	9.7	0.1	(Vanhoof and Tirez, 2014)
urban wastewater - Nuresys	58.4	12.2	5.1	9.1	0.3	(Vanhoof and Tirez, 2014)
potato industry - Crustell	13.6	10.7	4.7	9.2	3.3	(Vanhoof and Tirez, 2014)
urban wastewater - Aquafin	56.8	11.5	5.4	9.2	0.4	(Vanhoof and Tirez, 2014)

16.1.2 Metals and metalloids

Table 29: Metal/metalloid contents (mg kg^{-1} dry weight) for precipitated P-salts (confidential information received from the STRUBIAS subgroup is not included in this Table).

input material	As	Cd	Cu	Cr	Hg	Ni	Pb	Zn	Reference
				(mg kg^{-1}, dry matter)					
urban wastewater -Pearl	3	0.1	3	3	0	3	1	15	(Kraus and Seis, 2015)
urban wastewater - Airpex	1	0.3	42	16	0	16	13	90	(Kraus and Seis, 2015)
urban wastewater - Stuttgart	2	0.4	30	4	0	5	7	47	(Kraus and Seis, 2015)
urban wastewater - Gifhorn		0.2	12	2	0	2	1	24	(Kraus and Seis, 2015)
urban wastewater	<0.05	<0.01		<dl	<dl	<dl	<dl		(Ueno and Fujii, 2001)
urban wastewater		<0.4			0		5		(Münch and Barr, 2001)
urban wastewater		<0.4	2	2	0	<0.4	0		(Antakyal et al., 2011)
urban wastewater (unwashed product)	0	0	12	6	0	6	10	42	(STOWA, 2015)
urban wastewater (unwashed product)	0	0	5	12	0	6	2	16	(STOWA, 2015)
urban wastewater (unwashed product)	0	<0.03	2	3	0	<0.6	1	12	(STOWA, 2015)
urban wastewater (unwashed product)	<0.05	<0.6	<1.1	3	<0.01	2	<0.5	2	(STOWA, 2015)
urban wastewater	<1	<0.3	48	8	0	5	11	90	STRUBIAS - confidential data provider
urban wastewater	<0.6	<0.3	30	<0.3 (VI)	<0.06	2	6	67	STRUBIAS - confidential data provider
urban wastewater		<0.5	2	9	0	1	<0.5	5	(Weidelener et al., 2005)
manure			16					81	(Liu et al., 2011)
manure (Stichting Mestverwerking Gelderland)	<2	<1	5	2	0	<2	<0.1	59	(Ehlert et al., 2016a)
urban wastewater		<0.2	7	4		11	<0.2	19	(Plaza et al., 2007)
dairy industry		<dl	<dl	<dl	<dl	<dl	<dl	<dl	(Uysal and Kuru, 2015)
urine	<11	<1.6	89	<1.6		2	<21	224	(Gell et al., 2011)
waste water	<6	<1.1	36	<1	<0.1	<0.5	<16	<15	(Gell et al., 2011)
potato industry	<6	1	42	17		26	7	336	(Abma et al., 2009)
potato industry			2					9	(Sigurnjak et al., 2016)

potato industry - Nuresys	<0.5	<0.12	1	0.5		<0.25	1.0	6	(Vanhoof and Tirez, 2014)
food processing (vegetable oil) - Nuresys	1.4	<0.12	0	0.7		11.0	0.9	5	Vanhoof and Tirez, 2014
urban wastewater - Nuresys	<1.25	<0.31	11	1.3		1.2	3.9	22	(Vanhoof and Tirez, 2014)
potato industry - Crustell	0.6	0.9	34	5.9		7.3	2.6	179	(Vanhoof and Tirez, 2014)
urban wastewater - Aquafin	<0.5	<0.12	3	1.7		1.3	3.9	28	(Vanhoof and Tirez, 2014)

<dl: below detection limit

16.2 Thermal oxidation materials & derivates

16.2.1 Macroelements

Table 30: Macroelements (%, dry basis) for different types of thermal oxidation materials & derivates.

	n	Si (%)	Ca (%)	K (%)	P (%)	Al (%)	Mg (%)	Fe (%)	S (%)	Na (%)	Ti (%)	reference
plant biomass												
wood and woody biomass												
Wood ash	1		19.4	3.1	0.2		1.5		0.0			STRUBIAS contribution - confidential data provider
Wood ash	1		24.7	4.8	0.3		2.0		0.1			STRUBIAS contribution - confidential data provider
Wood ash	1		22.8	2.7	0.1		1.3		0.2			STRUBIAS contribution - confidential data provider
Wood ash	1		8.7	2.1	0.1		0.7		0.0			STRUBIAS contribution - confidential data provider
Wood ash	1		8.3	1.7	0.1		0.7		0.0			STRUBIAS contribution - confidential data provider
Wood ash	1		13.1	1.4	0.1		0.8		0.0			STRUBIAS contribution - confidential data provider
Wood ash	1		5.8	1.2	0.1		0.5		0.0			STRUBIAS contribution - confidential data provider
Wood ash	1		9.3	2.0	0.1		0.7		0.1			STRUBIAS contribution - confidential data provider
Wood ash	1		18.5	3.4	0.2		1.4		0.1			STRUBIAS contribution - confidential data provider
Wood ash	1		21.3	2.1	0.1		1.6		0.0			STRUBIAS contribution - confidential data provider
Wood ash	23		18.1	6.1	1.9							STRUBIAS contribution - DK
Alder-fir sawdust #	1	17.5	18.8	5.1	0.9	6.5	2.4	5.7	0.3	1.3	0.6	(Miles et al., 1996)
Balsam bark #	1	12.2	32.5	8.9	2.1	1.0	1.4	1.9	1.1	2.0	0.1	(Bryers, 1996)
Beech bark #	1	5.8	48.4	2.2	1.0	0.1	6.9	0.8	0.3	0.7	0.1	(Bryers, 1996)
Birch bark #	2	2.0	49.0	7.5	1.8	0.3	3.6	1.6	1.1	1.4	0.1	(Bryers, 1996)
Christmas trees #	1	18.6	6.9	6.7	1.1	8.0	1.6	6.7	4.7	0.4	0.2	(Miles et al., 1996)
Elm bark #	1	2.1	59.3	4.5	0.7	0.1	1.5	0.3	0.4	0.6	0.1	(Bryers, 1996)
Eucalyptus bark #	1	4.7	41.0	7.7	1.0	1.6	6.5	0.8	1.4	1.4	0.1	(Theis et al., 2006)
Fir mill residue #	2	9.0	10.7	7.4	1.6	2.7	3.5	5.8	1.5	22.1	0.2	(Bryers, 1996; Thy et al., 2006)
Forest residue #	3	9.6	33.8	8.5	2.2	1.6	4.3	1.0	1.2	1.2	0.2	(Miles et al., 1995; Miles et al., 1996; Zevenhoven-Onderwater et al., 2000)
Hemlock bark #	1	5.2	42.3	4.2	1.0	1.2	8.7	1.0	0.8	0.9	0.1	(Bryers, 1996)
Land clearing wood #	2	30.7	4.1	1.8	0.3	7.9	1.1	3.7	0.1	2.0	0.3	(Miles et al., 1995)
Maple bark #	1	4.2	47.8	5.8	0.3	2.1	4.0	1.0	0.8	1.3	0.1	(Bryers, 1996)

	n	Si (%)	Ca (%)	K (%)	P (%)	Al (%)	Mg (%)	Fe (%)	S (%)	Na (%)	Ti (%)	reference
Oak sawdust #	2	14.0	11.0	26.6	0.8	2.3	3.6	2.9	1.5	1.5	0.2	(Miles et al., 1995)
Oak wood #	1	22.8	12.4	7.9	0.8	5.0	0.7	5.9	1.0	0.4	0.1	(Misra et al., 1993; Demirbas, 2004)
Olive wood #	2	4.8	29.4	20.9	4.7	1.1	1.8	0.6	1.1	2.7	0.1	(Vamvuka and Zografos, 2004)
Pine bark #	1	4.3	40.3	6.5	2.2	3.8	3.7	2.0	1.1	1.5	0.1	(Misra et al., 1993; Bryers, 1996)
Pine chips #	1	31.8	5.6	3.7	0.7	3.7	1.5	3.8	0.5	0.9	0.3	(Masia et al., 2007)
Pine pruning #	2	3.6	31.3	18.5	2.5	1.5	6.8	0.9	1.7	0.3	0.1	(Lapuerta et al., 2008)
Pine sawdust #	3	4.5	34.7	11.9	2.7	1.2	8.3	1.5	0.9	0.3	0.1	(Etiegni and Campbell, 1991)
Poplar #	1	1.8	40.7	15.5	0.4	0.4	7.9	0.8	1.5	0.2	0.2	(Misra et al., 1993; Miles et al., 1995)
Poplar bark #	2	0.9	54.9	7.4	1.1	0.3	1.4	0.5	0.3	3.6	0.1	(Bryers, 1996)
Sawdust #	3	12.2	31.3	9.0	1.0	2.4	3.2	1.3	0.8	1.8	0.2	(Tillman, 2000; Wigley et al., 2007)
Spruce bark #	1	2.9	51.4	6.0	1.2	0.4	3.0	1.3	0.8	1.5	0.1	(Bryers, 1996; Demirbas, 2005)
Spruce wood #	1	23.0	12.2	8.0	0.8	5.0	0.7	5.8	1.0	0.4	0.1	(Demirbas, 2005)
Tamarack bark #	11	3.6	38.0	4.7	2.2	4.7	5.4	2.7	1.1	2.5	0.1	(Bryers, 1996)
Willow #	1	2.8	32.7	19.4	5.7	1.0	2.4	0.5	1.2	1.2	0.0	(Miles et al., 1995; Zevenhoven-Onderwater et al., 2000)
Wood	1	10.8	26.5	9.6	1.3	3.0	4.4	2.3	2.0	1.9	0.7	(Wei et al., 2005)
Wood residue	2	24.8	8.3	4.0	0.6	6.7	1.8	4.4	0.8	3.3	0.3	(Miles et al., 1995)
Wood fly ash	1		24.7	5.0	1.3	1.0	2.4	1.6		0.4		(ECN, 2017); biodat_sample_#326
Wood ash	1	27.0	14.2	3.5	0.7	1.6	1.5	1.1		0.4		(ECN, 2017); biodat_sample_#327
Wood bottom ash	1		34.9	8.6	2.2		2.7			0.7		(ECN, 2017); biodat_sample_#328
Wood ash	1		29.0	10.7	2.0		2.5			0.9		(ECN, 2017); biodat_sample_#329
Wood fly ash	1		25.4	8.9	1.9		2.1			0.8		(ECN, 2017); biodat_sample_#330
Wood ash	1	12.0	21.0	7.4	2.4	1.2	2.8	2.0		0.9	0.1	(ECN, 2017); biodat_sample_#331
Wood fly ash	1		16.0	1.1	0.6	1.1	2.1	1.2		0.3		(ECN, 2017); biodat_sample_#332
Wood fly ash	1	19.0	14.0	3.5	0.8	4.0	1.7	2.7		1.5		(ECN, 2017); biodat_sample_#333
Wood fly ash	1	19.0	14.0	4.0	1.0	4.1	1.7	2.5		1.5	0.2	(ECN, 2017); biodat_sample_#334
Wood fly ash	1	3.6	22.0	9.0	1.4	0.9	3.7	1.4		1.0	0.1	(ECN, 2017); biodat_sample_#335
Wood bottom ash	1	5.8	5.7	1.8	0.3	1.0	0.9	0.7		0.4	0.1	(ECN, 2017); biodat_sample_#336
Wood fly ash	1	16.5	18.9	3.8	1.1	4.3	2.2	3.6	1.5	1.2	0.5	(ECN, 2017); biodat_sample_#352
Wood fly ash	1	19.8	14.9	4.5	0.9	5.0	1.5	2.9	1.1	1.5	0.4	(ECN, 2017); biodat_sample_#353
Wood fly ash	1	14.2	19.1	3.6	1.1	4.5	1.6	4.9	1.9	1.3	0.2	(ECN, 2017); biodat_sample_#354
Wood fly ash	1	15.7	17.2	3.0	0.9	4.6	1.4	4.1	1.9	1.0	0.3	(ECN, 2017); biodat_sample_#355
Wood fly ash	1	17.9	16.9	2.9	0.6	4.7	1.6	3.4	2.0	1.2	0.4	(ECN, 2017); biodat_sample_#356
Wood fly ash	1	23.4	12.3	3.8	0.8	4.6	1.5	3.8	0.8	1.1	0.5	(ECN, 2017); biodat_sample_#357
Wood fly ash	1	20.6	13.2	3.4	0.6	5.3	1.6	4.2	1.3	1.2	0.6	(ECN, 2017); biodat_sample_#358
Wood fly ash	1	21.3	12.8	3.7	0.6	5.2	1.5	3.8	1.1	1.2	0.5	(ECN, 2017); biodat_sample_#359
Wood fly ash	1	16.5	17.9	3.9	1.0	4.3	2.1	3.7	1.5	1.2	0.8	(ECN, 2017); biodat_sample_#360

	n	Si (%)	Ca (%)	K (%)	P (%)	Al (%)	Mg (%)	Fe (%)	S (%)	Na (%)	Ti (%)	reference
Wood fly ash	1	19.2	16.7	3.8	0.9	4.5	1.9	3.3	1.2	1.3	0.6	(ECN, 2017); biodat_sample_#361
Wood ash	1		21.0	3.8	1.8	1.0	2.9	0.5	0.4	0.3		(ECN, 2017); biodat_sample_#362
Wood fly ash	1	17.1	18.4	3.7	1.1	4.4	2.1	3.4	1.3	1.2	0.5	(ECN, 2017); biodat_sample_#363
Wood fly ash	1	19.9	15.8	4.0	1.0	4.6	1.9	3.3	1.0	1.3	0.4	(ECN, 2017); biodat_sample_#364
Wood fly ash	1	21.6	13.9	4.3	0.8	4.9	1.8	3.1	1.0	1.3	0.4	(ECN, 2017); biodat_sample_#365
Wood fly ash	1	20.3	14.9	4.2	1.0	4.9	1.9	3.4	1.5	1.3	0.4	(ECN, 2017); biodat_sample_#366
Wood fly ash	1	17.1	18.7	3.5	1.0	4.6	2.0	3.4	2.2	1.1	0.4	(ECN, 2017); biodat_sample_#367
Wood fly ash	1	17.1	19.4	3.5	0.9	4.5	1.9	3.4	2.0	1.0	0.4	(ECN, 2017); biodat_sample_#368
Wood ash	1		8.8	5.4	1.3	0.6	1.2	0.4	1.2	0.5		(ECN, 2017); biodat_sample_#369
Wood ash	1		8.1	2.0	0.5	1.7	1.0	2.2	0.2	0.7		(ECN, 2017); biodat_sample_#370
Wood ash	1		9.9	3.6	1.1	0.7	1.2	0.5	0.4	0.4		(ECN, 2017); biodat_sample_#371
Wood ash	1		16.0	7.6	2.3	0.9	2.9	0.6	0.4	0.6		(ECN, 2017); biodat_sample_#372
Wood ash	1		12.0	5.4	1.5	1.0	1.7	0.3	0.7	0.5		(ECN, 2017); biodat_sample_#373
Wood ash	1		13.0	6.2	1.1	2.1	1.6	1.5	1.1	1.0		(ECN, 2017); biodat_sample_#374
Wood ash	1		16.0	5.0	2.0	0.8	2.7	0.5	0.5	0.3		(ECN, 2017); biodat_sample_#375
Wood ash	1		11.0	7.8	1.7	0.8	1.6	0.6	1.6	0.7		(ECN, 2017); biodat_sample_#376
Wood fly ash	1	9.4	22.0	6.1	1.8	0.6	0.9	1.6		0.4		(ECN, 2017); biodat_sample_#377
Wood fly ash	1	11.5	20.3	7.8	2.1	1.8	2.6	0.9	2.7	1.1	0.1	(ECN, 2017); biodat_sample_#378
Wood bottom ash	1	36.1	3.4	2.5	0.3	2.2	0.5	0.5	0.0	0.7	0.0	(ECN, 2017); biodat_sample_#379
Wood fly ash	1	13.0	19.9	6.6	1.9	2.1	2.5	1.1	2.4	1.1	0.3	(ECN, 2017); biodat_sample_#380
Wood bottom ash	1	36.7	3.4	2.3	0.3	2.1	0.5	0.5	0.0	0.6	0.1	(ECN, 2017); biodat_sample_#381
Wood fly ash	1	13.0	18.9	6.6	1.6	2.6	2.3	1.5	2.4	1.2	0.5	(ECN, 2017); biodat_sample_#382
Wood bottom ash	1	36.1	3.3	2.4	0.3	2.3	0.5	0.5	0.0	0.7	0.1	(ECN, 2017); biodat_sample_#383
Wood fly ash	1	14.4	17.8	6.0	1.4	3.0	2.2	1.7	2.1	1.4	0.5	(ECN, 2017); biodat_sample_#384
Wood bottom ash	1	36.5	3.0	2.5	0.2	2.6	0.5	0.6	0.0	0.9	0.1	(ECN, 2017); biodat_sample_#385
Wood fly ash	1	14.4	18.0	5.7	1.3	3.0	2.1	1.8	1.8	1.3	0.6	(ECN, 2017); biodat_sample_#386
Wood bottom ash	1	36.9	3.3	2.6	0.3	2.6	0.5	0.6	0.0	0.9	0.1	(ECN, 2017); biodat_sample_#387
Wood fly ash	1	13.4	17.1	6.8	1.4	2.9	2.3	1.6	2.2	1.5	1.0	(ECN, 2017); biodat_sample_#388
Wood bottom ash	1	35.6	3.6	2.8	0.3	2.8	0.5	0.7	0.0	1.0	0.1	(ECN, 2017); biodat_sample_#389
Wood fly ash	1	15.0	16.4	6.5	1.4	3.1	2.1	1.5	2.1	1.4	0.8	(ECN, 2017); biodat_sample_#390
Wood bottom ash	1	36.3	3.6	3.0	0.3	2.8	0.5	0.6	0.0	0.9	0.1	(ECN, 2017); biodat_sample_#391
Wood fly ash	1	6.7	26.4	7.8	1.9	1.4	2.4	0.8	2.2	0.5	0.1	(ECN, 2017); biodat_sample_#392
Wood fly ash	1	3.7	27.0	8.9	1.9	0.5	1.9	0.3		0.3		(ECN, 2017); biodat_sample_#393
Wood fly ash	1	14.0	15.0	5.0	1.2	1.5	1.3	0.5	1.9	0.6	0.1	(ECN, 2017); biodat_sample_#394
Wood fly ash	1	0.9	25.3	7.6	1.4	1.2	2.2	0.5		1.3	0.0	(ECN, 2017); biodat_sample_#395
Wood fly ash	1	1.3	20.7	8.4	1.2	1.3	1.9	0.4		1.6	0.0	(ECN, 2017); biodat_sample_#396

	n	Si (%)	Ca (%)	K (%)	P (%)	Al (%)	Mg (%)	Fe (%)	S (%)	Na (%)	Ti (%)	reference
Wood bottom ash	1	30.0	7.4	7.3	0.3	4.6	1.1	1.4		1.4	0.1	(ECN, 2017); biodat_sample_#397
Wood fly ash	1	21.7	12.4	4.1	0.8	2.1	1.2	0.6	1.6	0.7	0.1	(ECN, 2017); biodat_sample_#398
Wood fly ash	1	8.8	14.0	6.3	1.6	1.4	1.7	0.6	3.4	0.6	0.0	(ECN, 2017); biodat_sample_#399
Wood fly ash	1	14.0	22.0	5.4	1.5	0.7	1.7	0.5	2.7	0.6	0.0	(ECN, 2017); biodat_sample_#400
Wood fly ash	1	11.7	15.9	1.1		7.2	1.3	0.2	2.4	0.3		(ECN, 2017); biodat_sample_#401
Wood fly ash	1	3.8	27.5	5.0	1.3	0.9	1.9	0.6	1.4	0.4		(ECN, 2017); biodat_sample_#402
Wood fly ash	1	17.9	14.6	3.8	0.8	6.1	2.0	4.6		1.4	0.3	(ECN, 2017); biodat_sample_#403
Wood bottom ash	1	30.1	4.6	4.9	0.1	6.6	0.7	2.6		2.1	0.1	(ECN, 2017); biodat_sample_#404
Wood fly ash	1	20.0	8.5	4.1	0.4	6.6	1.2	4.1		1.4	0.3	(ECN, 2017); biodat_sample_#405
Wood bottom ash	1	27.0	7.6	6.0	0.3	6.0	1.0	2.6		1.6	0.2	(ECN, 2017); biodat_sample_#406
Wood fly ash	1	9.1	20.0	4.2	1.2	3.2	2.1	3.9		0.9	0.2	(ECN, 2017); biodat_sample_#407
Wood bottom ash	1	26.0	7.2	5.7	0.3	5.8	0.9	1.7		1.7	0.1	(ECN, 2017); biodat_sample_#408
Wood fly ash	1	13.0	20.0	4.1	1.1	4.0	1.8	5.2		1.2	0.2	(ECN, 2017); biodat_sample_#409
Wood bottom ash	1	30.0	3.3	3.3	0.1	6.7	1.5	3.1		2.2	0.3	(ECN, 2017); biodat_sample_#410
Wood fly ash	1	12.6	20.2	3.8	1.0	4.2	1.9	3.6	2.6	1.2	0.6	(ECN, 2017); biodat_sample_#411
Wood bottom ash	1	31.0	6.3	6.3	0.4	4.0	0.8	1.2		1.4	0.1	(ECN, 2017); biodat_sample_#412
Wood bottom ash	1	34.0	3.5	4.3	0.2	4.5	0.5	1.1		1.6	0.1	(ECN, 2017); biodat_sample_#413
Wood bottom ash	1	32.0	5.5	5.2	0.4	3.9	0.8	1.2		1.2	0.1	(ECN, 2017); biodat_sample_#414
Wood bottom ash	1	31.0	7.4	5.7	0.4	3.7	1.6	1.4		1.1	0.1	(ECN, 2017); biodat_sample_#415
Wood bottom ash	1	29.0	7.1	6.7	0.4	4.8	0.9	1.6		1.5	0.1	(ECN, 2017); biodat_sample_#416
Wood bottom ash	1	28.0	8.5	7.6	0.4	4.3	1.0	1.4		1.5	0.1	(ECN, 2017); biodat_sample_#417
Wood bottom ash	1	30.0	6.9	7.2	0.3	4.4	0.8	1.2		1.4	0.1	(ECN, 2017); biodat_sample_#418
Wood bottom ash	1		8.1	4.5	1.3	0.6	1.0	0.5	0.9	0.6		(ECN, 2017); biodat_sample_#419
Wood bottom ash	1	31.0	6.2	4.9	0.3	4.6	0.8	1.2		1.7	0.1	(ECN, 2017); biodat_sample_#420
Wood fly ash	1	1.0	15.0	24.0	0.9	0.3	2.1	0.5		1.3	0.0	(ECN, 2017); biodat_sample_#421
Wood fly ash	1	0.9	15.0	24.0	0.9	0.3	2.2	0.8		1.2	0.0	(ECN, 2017); biodat_sample_#422
Wood bottom ash	1	8.3	9.0	1.9	0.5	1.0	1.5	1.4		0.3	0.1	(ECN, 2017); biodat_sample_#423
Wood fly ash	1	26.0	7.2	3.0	0.4	4.4	1.0	1.6		1.5	0.4	(ECN, 2017); biodat_sample_#424
Wood fly ash	1	29.0	5.1	7.1	0.3	6.1	0.7	1.4		2.1	0.2	(ECN, 2017); biodat_sample_#425
Wood ash	1	6.2	31.0	4.5	2.5		2.3	0.6		0.3	0.1	(ECN, 2017); biodat_sample_#426
Wood bottom ash	1	25.0	10.0	3.5	0.4	4.5	1.4	1.7		1.3	0.2	(ECN, 2017); biodat_sample_#427

	n	Si (%)	Ca (%)	K (%)	P (%)	Al (%)	Mg (%)	Fe (%)	S (%)	Na (%)	Ti (%)	reference
Wood bottom ash	1	27.0	6.7	3.2	0.3	2.2	1.3	1.5		1.2	0.1	(ECN, 2017); biodat_sample_#428
Wood bottom ash	1	31.0	6.4	3.2	0.4	3.5	1.4	1.5		1.1	0.1	(ECN, 2017); biodat_sample_#429
Wood fly ash	1	9.0	24.0	10.0	1.4	2.3	2.6	1.8		0.6	0.1	(ECN, 2017); biodat_sample_#444
Wood fly ash	1	11.0	27.0	5.8	1.0	2.3	2.0	1.4		0.5	0.1	(ECN, 2017); biodat_sample_#445
Wood fly ash	1	12.0	21.0	5.6	1.1	0.9	2.4	1.6		0.6	0.1	(ECN, 2017); biodat_sample_#446
Wood fly ash	1	14.1	18.1	5.9	1.1	2.0	2.7	1.7		0.6	0.1	(ECN, 2017); biodat_sample_#447
Wood fly ash	1		22.1	6.9	1.3		2.4					(ECN, 2017); biodat_sample_#449
Wood fly ash	1		30.2	1.9	1.7		2.6					(ECN, 2017); biodat_sample_#450
Wood fly ash	1		27.1	9.4	2.1		3.4					(ECN, 2017); biodat_sample_#451
Wood fly ash	1		31.2	6.5	1.3		2.1					(ECN, 2017); biodat_sample_#452
Wood fly ash	1		24.1	6.8	1.6		2.7					(ECN, 2017); biodat_sample_#453
Wood fly ash	1	9.3	17.5	4.7	1.0	4.5	1.8	1.2		0.7	0.4	(ECN, 2017); biodat_sample_#454
Wood bottom ash	1	23.0	15.0	4.2	0.7	3.9	1.9	1.8		1.3	0.2	(ECN, 2017); biodat_sample_#455
Wood fly ash	1	5.6	18.0	15.0	1.0	1.7	1.9	1.0		1.6	0.1	(ECN, 2017); biodat_sample_#456
Wood fly ash	1	12.0	18.0	5.0	1.0	2.7	1.8	1.1	1.3	0.7	0.2	(ECN, 2017); biodat_sample_#457
Wood bottom ash	1	25.0	13.0	4.0	0.6	4.6	1.5	1.8	0.1	1.2	0.2	(ECN, 2017); biodat_sample_#458
Wood bottom ash	1	12.0	13.0	3.8	0.7	5.7	1.4	1.3		0.7	0.7	(ECN, 2017); biodat_sample_#459
Wood bottom ash	1	21.5	11.4	3.9	0.6	5.8	1.5	2.1		1.4	0.3	(ECN, 2017); biodat_sample_#460
Wood fly ash	1	21.0	14.0	3.3	0.7	8.4	1.6	2.5		1.1	1.0	(ECN, 2017); biodat_sample_#461
Wood ash	1		24.0	5.9	1.4	2.1		4.3	0.6	0.8		(ECN, 2017); biodat_sample_#462
Wood ash	1		26.1	4.0	2.2	1.1		1.8	0.4	0.5		(ECN, 2017); biodat_sample_#463
Wood ash	1		9.5	3.0	0.7	1.5		2.0	0.3	0.5		(ECN, 2017); biodat_sample_#464
Wood ash	1		8.9	2.4	0.4	1.6		2.1	0.3	0.7		(ECN, 2017); biodat_sample_#465
Wood ash	1		13.0	6.6	1.4	1.6	1.9	1.1	0.5	0.8		(ECN, 2017); biodat_sample_#466
Wood ash	1		5.3	5.4	0.4	1.2	0.8	1.2	1.6	0.8		(ECN, 2017); biodat_sample_#467
Wood bottom ash	1		29.0	4.6	2.7	1.0	2.9	0.6	0.2	0.2		(ECN, 2017); biodat_sample_#468
Wood bottom ash	1		13.0	5.1	1.2	1.3	1.6	1.0	0.4	0.6		(ECN, 2017); biodat_sample_#469
Hemp ash	1	16.0	15.0	6.9	2.2	2.4	1.2	4.5		0.9	0.2	(ECN, 2017); biodat_sample_#470
Hemp ash	1	12.0	20.0	8.9	3.0	1.3	1.2	1.6		0.6	0.1	(ECN, 2017); biodat_sample_#471
Hemp ash	1	12.0	17.0	11.0	2.5	1.2	1.2	1.3		0.6	0.1	(ECN, 2017); biodat_sample_#472
Hemp ash	1	11.0	21.0	10.0	2.4	1.0	1.5	1.2		0.8	0.1	(ECN, 2017); biodat_sample_#473

	n	Si (%)	Ca (%)	K (%)	P (%)	Al (%)	Mg (%)	Fe (%)	S (%)	Na (%)	Ti (%)	reference
Hemp ash	1	11.0	22.0	8.4	2.0	1.1	1.3	1.3		0.6	0.1	(ECN, 2017); biodat_sample_#474
mean		*16.6*	*18.2*	*6.2*	*1.2*	*2.9*	*2.1*	*1.9*	*1.2*	*1.2*	*0.2*	*n =204*
median		*14.4*	*16.0*	*5.1*	*1.0*	*2.4*	*1.8*	*1.5*	*1.1*	*1.1*	*0.1*	
minimum		*0.9*	*3.0*	*1.1*	*0.1*	*0.1*	*0.5*	*0.2*	*0.0*	*0.2*	*0.0*	
10th percentile		*3.6*	*5.7*	*2.9*	*0.3*	*0.7*	*0.8*	*0.5*	*0.2*	*0.4*	*0.1*	
90th percentile		*31.0*	*32.6*	*9.1*	*2.2*	*5.7*	*3.5*	*3.9*	*2.2*	*1.6*	*0.5*	
maximum		*36.9*	*59.3*	*26.6*	*5.7*	*8.4*	*8.7*	*6.7*	*4.7*	*22.1*	*1.0*	
coefficient of variation		*0.6*	*0.6*	*0.7*	*0.7*	*0.7*	*0.7*	*0.8*	*0.7*	*1.5*	*1.0*	
grass												
Arundo grass #	1	22	2	27	3	0	2	1	2	0	0	(Miles et al., 1995)
Bamboo whole #	1	5	3	44	9	0	4	0	1	0	0	(Scurlock et al., 2000)
Bana grass #	1	18	3	41	1	0	1	1	0	0	0	(Miles et al., 1995)
Buffalo gourd grass #	1	4	10	34	5	1	3	1	4	5	0	(Miles et al., 1995)
Kenaf grass #	1	4	31	16	2	1	5	1	3	1	0	(Miles et al., 1995; Moilanen, 2006)
Miscanthus grass #	4	26	8	16	2	0	2	1	1	0	0	(Miles et al., 1995; Wigley et al., 2007)
Reed canary grass #	1	40	2	2	2	1	1	1	0	0	0	(Moilanen, 2006)
Sorghastrum grass #	1	34	5	7	2	1	1	1	0	0	0	(Miles et al., 1995)
Sweet sorghum grass #	1	31	7	8	2	0	2	0	1	1	0	(Moilanen, 2006)
Switchgrass #	3	31	7	8	2	1	3	1	0	0	0	(Miles et al., 1995)
mean		*21.5*	*8.0*	*20.4*	*2.9*	*0.7*	*2.4*	*0.7*	*1.5*	*0.9*	*0.0*	*n = 15*
median		*24.2*	*6.1*	*16.1*	*1.8*	*0.6*	*1.9*	*0.7*	*1.2*	*0.4*	*0.0*	
minimum		*4.1*	*2.1*	*2.4*	*1.4*	*0.4*	*0.9*	*0.4*	*0.3*	*0.1*	*0.0*	
10th precentile		*4.4*	*2.3*	*6.9*	*1.5*	*0.4*	*1.1*	*0.5*	*0.4*	*0.2*	*0.0*	
90th percentile		*34.7*	*12.6*	*41.1*	*5.2*	*1.2*	*4.1*	*1.0*	*3.3*	*1.7*	*0.1*	
maximum		*39.6*	*31.5*	*44.3*	*8.9*	*1.4*	*5.2*	*1.2*	*4.0*	*4.6*	*0.2*	
coefficient of variation		*0.6*	*1.1*	*0.7*	*0.8*	*0.5*	*0.6*	*0.3*	*0.9*	*1.5*	*1.0*	
straw												
Wheat	1	32.1	5	30.3	2.8	3.6	2.2	0.4	1.4	18.2		(Demirbas, 2004)
Alfalfa #	1	3.7	17.7	31.7	4.6	0.1	8.5	0.3	1.0	1.1	0.0	(Miles et al., 1996)

	n	Si (%)	Ca (%)	K (%)	P (%)	Al (%)	Mg (%)	Fe (%)	S (%)	Na (%)	Ti (%)	reference
Barley #	2	23.7	7.0	23.4	1.3	0.4	1.7	0.7	0.9	1.0	0.0	(Risnes et al., 2003)
Corn #	1	23.3	10.5	15.4	1.1	2.7	2.7	1.8	0.7	0.1	0.2	(Masia et al., 2007)
Mint #	1	11.0	12.5	26.6	2.5	2.9	4.1	2.0	1.4	1.5	0.2	(Miles et al., 1996)
Oat #	1	17.6	8.5	22.3	2.7	2.5	2.7	1.5	2.0	0.5	0.1	(Theis et al., 2006)
Rape #	3	19.0	21.8	11.2	1.0	2.9	1.2	1.4	1.1	0.3	0.2	(Masia et al., 2007)
Rice #	2	36.0	1.7	10.4	0.4	0.3	1.6	0.3	0.5	1.3	0.0	(Miles et al., 1996; Thy et al., 2000; Thy et al., 2006)
Unknown	14	26.7	4.8	21.4	1.2	0.4	1.0	0.4	1.6	0.5	0.0	(Wieck-Hansen et al., 2000; Wei et al., 2005)
Wheat	9	23.5	5.8	20.7	1.6	0.8	1.6	0.6	1.7	2.6	0.1	(Miles et al., 1995; Bryers, 1996; Miles et al., 1996; Risnes et al., 2003; Demirbas, 2004; Thy et al., 2006; Nutalapati et al., 2007)
mean		*21.7*	*9.5*	*21.3*	*1.9*	*1.7*	*2.7*	*0.9*	*1.2*	*2.7*	*0.1*	*n = 35*
median		*23.4*	*7.8*	*21.9*	*1.4*	*1.6*	*2.0*	*0.6*	*1.2*	*1.1*	*0.1*	
minimum		*3.7*	*1.7*	*10.4*	*0.4*	*0.1*	*1.0*	*0.3*	*0.5*	*0.1*	*0.0*	
10th precentile		*10.2*	*4.5*	*11.1*	*0.9*	*0.3*	*1.2*	*0.3*	*0.7*	*0.3*	*0.0*	
90th percentile		*32.5*	*18.1*	*30.4*	*3.0*	*3.0*	*4.6*	*1.8*	*1.7*	*4.2*	*0.2*	
maximum		*36.0*	*21.8*	*31.7*	*4.6*	*3.6*	*8.5*	*2.0*	*2.0*	*18.2*	*0.2*	
coefficient of variation		*0.4*	*0.7*	*0.3*	*0.6*	*0.8*	*0.8*	*0.7*	*0.4*	*2.0*	*0.8*	
other residues												
Almond hulls #	1	5.2	6.9	53.0	2.7	1.3	2.4	0.6	0.2	0.8	0.0	(Miles et al., 1996)
Almond shells #	1	7.9	8.2	44.4	2.2	1.6	2.7	1.9	0.4	1.3	0.1	(Miles et al., 1996; Demirbas, 2004)
Coconut shells #	1	31.2	1.7	7.0	0.7	4.5	0.9	4.3	0.0	3.4	0.0	(Miles et al., 1996)
Coffee husks #	1	6.8	9.3	43.5	2.2	3.7	2.6	1.4	0.2	0.5	0.2	(Miles et al., 1996)
Cotton husks #	1	5.1	14.9	41.7	1.8	0.7	4.6	1.3	0.7	1.0	0.0	(Miles et al., 1996)
Grape #	1	4.4	20.2	30.6	3.9	1.4	2.9	1.2	2.5	0.5	0.1	(Lapuerta et al., 2008)
Groundnut shells #	1	12.9	17.6	7.1	1.6	4.4	3.2	7.2	4.2	0.6	0.1	(Miles et al., 1996)
Hazelnut shells #	1	15.7	10.9	25.2	1.4	1.6	4.7	2.7	0.4	1.0	0.1	(Demirbas, 2004)
Mustard husks #	1	8.1	31.3	6.3	0.9	0.8	5.7	0.6	5.9	1.5	0.1	(Werther et al., 2000)
Olive husks #	1	15.3	10.3	3.6	1.1	4.4	2.5	4.4	0.2	19.4	0.2	(Demirbas, 2004)
Olive pits #	2	10.0	14.2	13.6	4.3	3.2	2.3	3.0	0.9	11.7	0.2	(Miles et al., 1996) (Demirbas, 2004)
Olive residue #	1	10.4	9.2	35.5	2.7	2.2	3.5	1.4	1.5	0.1	0.1	(Masia et al., 2007)
Palm fibres-husks #	1	29.5	6.4	7.5	1.2	2.4	2.3	2.7	1.1	0.6	0.1	(Werther et al., 2000)
Palm kernels #	1	8.5	6.6	13.7	13.7	3.3	4.0	6.5	1.0	0.1	0.1	(Masia et al., 2007)

	n	Si (%)	Ca (%)	K (%)	P (%)	Al (%)	Mg (%)	Fe (%)	S (%)	Na (%)	Ti (%)	reference
Pepper plant #	1	5.9	22.9	20.4	2.3	2.6	4.4	1.4	3.9	0.7	0.3	(Masia et al., 2007)
Pepper residue #	1	7.2	7.1	29.3	4.9	4.4	2.7	2.4	4.2	0.8	0.1	(Werther et al., 2000)
Pistachio shells #	1	3.9	7.3	15.5	5.3	1.2	2.0	25.4	1.6	3.4	0.1	(Miles et al., 1996)
Plum pits #	5	1.7	10.6	37.8	9.0	0.1	7.1	0.5	1.0	0.3	0.0	(Miles et al., 1996)
Rice husks #	1	44.1	0.7	1.9	0.2	0.1	0.1	0.2	0.4	0.1	0.0	(Bryers, 1996; Miles et al., 1996; Vassilev et al., 2000; Feng et al., 2004; Umantaheswaran and Batra, 2008)
Soya husks #	2	0.9	17.9	29.9	2.5	4.6	5.0	2.1	1.7	4.6	0.1	(Werther et al., 2000)
Sugar cane #	2	21.8	3.5	5.8	1.7	7.7	2.7	7.8	1.4	1.2	1.2	(Miles et al., 1996)
Sunflower husks #	1	11.0	10.9	23.7	3.1	4.6	4.4	3.0	1.6	0.6	0.1	(Werther et al., 2000; Demirbas, 2004)
Walnut blows #	1	3.0	19.6	28.8	4.5	1.2	8.6	0.7	0.9	0.7	0.1	(Miles et al., 1996)
Walnut hulls #	1	3.9	14.2	32.9	3.3	1.5	9.7	1.0	1.1	0.9	0.1	(Miles et al., 1996)
Walnut shells #	1	10.9	11.9	27.4	2.7	1.3	8.1	1.0	0.9	0.7	0.1	(Demirbas, 2004)
mean		*11.4*	*11.8*	*23.4*	*3.2*	*2.6*	*4.0*	*3.4*	*1.5*	*2.3*	*0.1*	*n = 31*
median		*8.1*	*10.6*	*25.2*	*2.5*	*2.2*	*3.2*	*1.9*	*1.0*	*0.8*	*0.1*	
Minimum		*0.9*	*0.7*	*1.9*	*0.2*	*0.1*	*0.1*	*0.2*	*0.0*	*0.1*	*0.0*	
10th percentile		*3.3*	*4.6*	*6.0*	*1.0*	*0.7*	*2.1*	*0.6*	*0.2*	*0.2*	*0.0*	
90th percentile		*26.4*	*20.0*	*42.8*	*5.2*	*4.6*	*7.7*	*6.9*	*4.0*	*4.1*	*0.2*	
Maximum		*44.1*	*31.3*	*53.0*	*13.7*	*7.7*	*9.7*	*25.4*	*5.9*	*19.4*	*1.2*	
coefficient of variation		*0.9*	*0.6*	*0.6*	*0.9*	*0.7*	*0.6*	*1.5*	*1.0*	*1.9*	*1.7*	
unknown plant origin												
unknown plant origin	24			37.8	3.4		4.5		1.8	0.3		STRUBIAS - ECOFI contribution
animal biomass												
poultry manure												
poultry manure	1			21.3	8.2							STTRUBIAS contribution - IE
poultry manure	1	0.9	12.7	5.6	10.4	0.9	7.9	1.5	2.4	1.2	0.0	STRUBIAS contribution - confidential data provider
poultry manure	1		32.2	4.6	4.4			4.4	2.7	1.0		STRUBIAS contribution - confidential data provider
poultry manure	1	2.7	26.5	5.7	7.2	0.5	1.9	0.2	1.7	0.3	0.0	(Masia et al., 2007)
poultry manure	1	1.5	17.4	8.0	10.6	0.4	3.1	0.5	3.1	1.7		STRUBIAS contribution - confidential data provider
poultry manure	?		1.9	10.3			1.0		2.1	5.5		STRUBIAS contribution - ESSP (Kalfos)
poultry manure	415		21.0	14.0	6.7					2.2		STRUBIAS contribution - ESSP (BMC Moerdijk, Billen et al.)
pig manure												

	n	Si (%)	Ca (%)	K (%)	P (%)	Al (%)	Mg (%)	Fe (%)	S (%)	Na (%)	Ti (%)	reference
pig manure	1	5.1	11.9	6.4	9.7	0.5	6.6	8.3	4.7	1.3	0.0	STRUBIAS contribution - confidential data provider
pig manure	1	0.9	20.5	23.6	3.6	0.1	1.8	0.9	2.3	1.9	0.0	STRUBIAS contribution - confidential data provider
slaughterhouse waste												
meat and bone meal	1	0.0	29.3	2.6	18.0	1.3	0.8	0.2	1.7	4.7	0.0	(Masia et al., 2007)
meat and bone meal	1	0.0	30.7	2.5	18.4	0.2	0.8	0.5	1.6	2.7	0.0	(Deydier et al., 2005b)
meat and bone meal	1	0.2	30.0	1.0	15.0	0.5	0.8		0.3	2.5		STRUBIAS contribution - ESPP
mean		*1.4*	*21.3*	*8.8*	*10.2*	*0.5*	*2.7*	*2.0*	*2.3*	*2.3*	*0.0*	*n = 425*
median		*0.9*	*21.0*	*6.0*	*9.7*	*0.5*	*1.8*	*0.7*	*2.2*	*1.9*	*0.0*	
Minimum		*0.9*	*12.7*	*4.6*	*4.4*	*0.4*	*1.9*	*0.2*	*1.7*	*0.3*	*0.0*	
10th percentile		*0.0*	*11.9*	*2.5*	*4.4*	*0.1*	*0.8*	*0.2*	*1.4*	*1.0*	*0.0*	
90th percentile		*3.4*	*30.7*	*20.6*	*18.0*	*1.0*	*6.9*	*5.6*	*3.3*	*4.7*	*0.0*	
Maximum		*5.1*	*32.2*	*23.6*	*18.4*	*1.3*	*7.9*	*8.3*	*4.7*	*5.5*	*0.0*	
coefficient of variation		*1.2*	*0.5*	*0.8*	*0.5*	*0.7*	*1.0*	*1.4*	*0.5*	*0.7*	*1.3*	
Contaminated biomass												
Sewage sludge (raw)	1	15.5	9.3	1.3	7.0	6.8	1.5	11.0	0.8	1.7	0.5	(Werther et al., 2000)
Sewage sludge (raw)	1	17.9	6.5	1.8	6.8	7.8	1.7	8.7	0.4	1.6	0.5	(Wei et al., 2005)
Sewage sludge (raw)	1	13.2	12.1	0.8	7.2	5.8	1.3	13.2	1.2	1.7	0.5	STRUBIAS contribution - confidential data provider
Sewage sludge (raw)	1	11.3	0.9	0.2	10.6	0.2	0.2	10.7	0.4	0.0		STRUBIAS contribution - confidential data provider
Sewage sludge (raw - DE)	252	12.1	13.8	0.9	7.3	5.2	1.4	9.9	1.5	0.7	0.4	(Krüger and Adam, 2015)
Sewage sludge (post-processed)	1		9.3	1.1	7.7		1.3		2.5			STRUBIAS contribution - ESPP (AshDec process)
slaughterhouse waste and sewage sludge mix	1		28.3	0.3	2.6							STRUBIAS contribution - FEhS
Currency shredded #	1	1.6	10.0	1.8	0.4	7.2	0.9	15.5	4.2	3.0	16.5	(Miles et al., 1995)
Demolition wood #	3	16.9	15.2	5.8	2.2	5.1	2.9	5.1	1.6	2.1	1.0	(Miles et al., 1995; Thy et al., 2000; Masia et al., 2007)
Furniture waste #	1	26.7	9.8	3.1	0.2	6.4	2.0	3.9	0.4	1.7	0.3	(Miles et al., 1995)
Mixed waste paper #	1	13.4	5.4	0.1	0.1	28.3	1.4	0.6	0.7	0.4	2.6	(Miles et al., 1995)
Greenhouse-plastic waste #	1	13.3	18.3	8.1	1.7	2.1	3.4	12.9	1.1	0.6	0.5	(Masia et al., 2007)
Refuse-derived fuel #	1	18.0	19.0	0.2	0.3	7.7	3.9	4.4	1.2	1.0	1.1	(Miles et al., 1995)
Wood yard waste #	1	28.0	17.0	2.5	0.9	1.6	1.3	1.4	1.0	0.7	0.2	(Miles et al., 1995)

	n	Si (%)	Ca (%)	K (%)	P (%)	Al (%)	Mg (%)	Fe (%)	S (%)	Na (%)	Ti (%)	reference
municipal solid waste	1	14.2	13.6	5.0	0.1	6.9	1.9	1.7	0.9	5.2		(Demirbas, 2004)
municipal solid waste (UK)	8	0.3	26.0	2.8	0.5	1.5	0.6	0.9	0.5	2.2		(Bogush et al., 2015)
paper industry waste	?		14.8	2.7	0.7							STRUBIAS contribution - CEPI
paper industry waste	391		12.4	2.5	0.2							STRUBIAS contribution - CEPI
mean		*14.5*	*13.4*	*2.3*	*3.1*	*6.6*	*1.7*	*7.1*	*1.2*	*1.6*	*2.2*	
median		*13.8*	*13.0*	*1.8*	*1.3*	*6.1*	*1.4*	*6.9*	*1.0*	*1.6*	*0.5*	*n = 667*
Minimum		*0.3*	*0.9*	*0.1*	*0.1*	*0.2*	*0.2*	*0.6*	*0.4*	*0.0*	*0.2*	
10th percentile		*4.5*	*6.1*	*0.2*	*0.2*	*1.5*	*0.7*	*1.0*	*0.4*	*0.5*	*0.3*	
90th percentile		*24.1*	*21.1*	*5.2*	*7.4*	*7.8*	*3.2*	*13.1*	*2.2*	*2.8*	*2.6*	
Maximum		*28.0*	*28.3*	*8.1*	*10.6*	*28.3*	*3.9*	*15.5*	*4.2*	*5.2*	*16.5*	
coefficient of variation		*0.5*	*0.5*	*0.9*	*1.1*	*1.0*	*0.6*	*0.7*	*0.8*	*0.8*	*2.2*	
Solid fossil fuels												
Peat #	1	17.5	7.1	0.9	1.2	10.7	1.3	9.7	4.8	0.1	0.2	(Theis et al., 2006)
Coal #	37	25.2	4.7	1.3	0.2	12.3	1.1	4.8	1.4	0.6	0.6	(Vassilev and Vassileva, 2007; Vassilev and Vassileva, 2009)
Lignite #	5	20.9	9.3	1.2	0.1	9.1	1.5	7.6	3.5	0.4	0.5	(Vassilev and Vassileva, 2007; Vassilev and Vassileva, 2009)
Sub-bituminous #	10	25.5	5.0	1.4	0.0	12.1	1.3	3.7	1.6	0.8	0.6	(Vassilev and Vassileva, 2007; Vassilev and Vassileva, 2009)
Bituminous #	22	26.2	3.5	1.3	0.1	13.1	0.9	4.7	0.9	0.6	0.7	(Vassilev and Vassileva, 2007; Vassilev and Vassileva, 2009)

values of the oxides are Si, Ca, K, P, Al, Mg, Fe, S, Na and Ti normalised to 100%

Proposal for the Revised Fertiliser Regulation

straight inorganic macronutrient fertiliser: the CE product shall contain one of the nutrients in the minimum quantity stated			**8.5**	**5.0**	**5.3**		**3.0**		**4.0**	**0.7**	
compound inorganic macronutrient fertiliser: the CE product shall contain more than one of the nutrients in the minimum quantity stated			**1.1**	**2.5**	**1.3**		**0.9**		**0.6**	**0.7**	

16.2.2 Metals and metalloids

Table 31: Metal and metalloid contents (mg kg^{-1}, dry basis) for different types of thermal oxidation materials & derivates.

Input material	n	Cd	Cr (total)	Hg	Ni	Pb	As	B	Ba	Be	Co	Cu	Mn	Mo	Sb	Se	Sn	Sr	V	Zn	refe
plant biomass											(mg kg^{-1})										
wood and woody biomass																					
Wood bottom ash																					reference
Wood bottom ash	1	2.3	40	0.01	12	16		99	1410		5.1	136	5900						14	446	(ECN, 2017) biodat_sample_#336
Wood bottom ash	1	0.73	102	0.82	59	116	26				14	116	5700						43	430	(ECN, 2017) biodat_sample_#337
Wood bottom ash	1	0.142	32.7		4	8.1			757		2.72	33.9	3990					229	8.66	697	(ECN, 2017) biodat_sample_#379
Wood bottom ash	1		40.1		4.57	9.07	6.13		781		3.43	48.2	4360					228	11.6	799	(ECN, 2017) biodat_sample_#381
Wood bottom ash	1		64.2		3.71	10.6	18.3		833		3.63	63.2	4040					227	8.77	985	(ECN, 2017) biodat_sample_#383
Wood bottom ash	1		47.4		3.2	12.7	14.7		832	0.542	3.11	50.7	3690					222	10.4	971	(ECN, 2017) biodat_sample_#385
Wood bottom ash	1		61.8		4.95	13.5	20.8		881		3.74	77.2	3920					235	9.28	1160	(ECN, 2017) biodat_sample_#387
Wood bottom ash	1		62		5.22	19.5	17.7		971		4.62	58	3940					255	10.4	1210	(ECN, 2017) biodat_sample_#389
Wood bottom ash	1	0.143	71.9		6.89	17	72.1		929	0.608	5.14	96.6	3530					254	10.2	1340	(ECN, 2017) biodat_sample_#391
Wood bottom ash	1		59		11	72	3.4	7.2	1510		7.8	56	3600						20	1240	(ECN, 2017) biodat_sample_#397
Wood bottom ash	1		50		19	14		26	1130		7	47	1900						33	856	(ECN, 2017) biodat_sample_#404
Wood bottom ash	1		54	0.01	28	13	7.5	76	1990		9.4	77	3000						40	1630	(ECN, 2017) biodat_sample_#406
Wood bottom ash	1		42		18	12		74	1770		6.5	62	3500						27	1330	(ECN, 2017) biodat_sample_#408
Wood bottom ash	1		118		65	19	11	13	763		14	28	900						74	606	(ECN, 2017) biodat_sample_#410
Wood bottom ash	1	0.15	30		11	12		72	1490		5.1	51	3600						15	1850	(ECN, 2017) biodat_sample_#412
Wood bottom ash	1	0.12	15		12	15		34	1040		4.2	28	1700						12	427	(ECN, 2017) biodat_sample_#413
Wood bottom ash	1		40		17	8.7		72	1270		26	47	2600						20	893	(ECN, 2017) biodat_sample_#414
Wood bottom ash	1	0.29	91		62	11	62	81	1420		20	45	2900						18	1070	(ECN, 2017) biodat_sample_#415
Wood bottom ash	1	0.14	22		9.6	9.8	5	55	1290		6.4	71	2800						20	1300	(ECN, 2017) biodat_sample_#416
Wood bottom ash	1	0.13	37					64	1600		13	64	3200						22	1430	(ECN, 2017) biodat_sample_#417
Wood bottom ash	1	0.56	16		10	13		58	1300		5.8	67	3100						18	1180	(ECN, 2017) biodat_sample_#418
Wood bottom ash	1	15	50	0.35	12	97	7.1	200			5.1	110	9000						6.9	2900	(ECN, 2017) biodat_sample_#419
Wood bottom ash	1	0.46	43		9.2	8.5		87	1470		5.3	36	3800						20	1170	(ECN, 2017) biodat_sample_#420
Wood bottom ash	1	0.48	71		17	8.2		98	1150		6.8	62	7100						18	106	(ECN, 2017) biodat_sample_#423
Wood bottom ash	1	0.52	37		15	20	3	51	1200		7.6	39	4100						30	190	(ECN, 2017) biodat_sample_#427
Wood bottom ash	1		120		31	21		40	1200		6.3	36	4400	4.6					17	240	(ECN, 2017) biodat_sample_#428
Wood bottom ash	1		360		53	12		43	1100		6	36	4000	7					19	270	(ECN, 2017) biodat_sample_#429
Wood bottom ash	1	0.83	130		34	19	8.8				7.9	230	5400						26	380	(ECN, 2017) biodat_sample_#430

Input material		n	Cd	Cr (total)	Hg	Ni	Pb	As	B	Ba	Be	Co	Cu	Mn	Mo	Sb	Se	Sn	Sr	V	Zn	
	Wood bottom ash	1		62		22	27	4				9.1	60	7600						18	400	(ECN, 2017) biodat_sample_#431
	Wood bottom ash	1	1.1	78	0.046	32	50					7.3	530	4100						20	370	(ECN, 2017) biodat_sample_#432
	Wood bottom ash	1	1	53	0.045	30	14					11	70							18	380	(ECN, 2017) biodat_sample_#433
	Wood bottom ash	1	4.9	49		18	45					10	59	7300						28	730	(ECN, 2017) biodat_sample_#434
	Wood bottom ash	1		91		33	7.6	12				7.3	38	2800						52	110	(ECN, 2017) biodat_sample_#435
	Wood bottom ash	1		63		19	16	3.9				8.7	36	5700						34	140	(ECN, 2017) biodat_sample_#436
	Wood bottom ash	1	0.76	78		39	19					8	62	4900						23	240	(ECN, 2017) biodat_sample_#437
	Wood bottom ash	1		49		20	10					5.1	52	6800						15	210	(ECN, 2017) biodat_sample_#438
	Wood bottom ash	1	0.56	39		22	10					5.8	40	4600						20	170	(ECN, 2017) biodat_sample_#439
	Wood bottom ash	1	0.67	30		34	6.2					7.6	73	3900						9.3	230	(ECN, 2017) biodat_sample_#440
	Wood bottom ash	1	1	36		21	9.1	4.1				6.9	48	5200						16	190	(ECN, 2017) biodat_sample_#441
	Wood bottom ash	1	0.92	39		20	10					7.1	50	6300						20	210	(ECN, 2017) biodat_sample_#442
	Wood bottom ash	1	0.88	30		18	7.1					6.4	42	5100						14	140	(ECN, 2017) biodat_sample_#443
	Wood bottom ash	1	0.73	80	0.02	39	37		135	1670		9.3	62	5000						26	719	(ECN, 2017) biodat_sample_#455
	Wood bottom ash	1	0.46	72		39	60	3.4	107			13	50	5000						29	257	(ECN, 2017) biodat_sample_#458
	Wood bottom ash	1	9.9	109	0.04	55	43		192	1060		7.5	76	4900						33	2230	(ECN, 2017) biodat_sample_#459
	Wood bottom ash	1	0.14	67		35	56		91	1360		9.1	62	5300						33	155	(ECN, 2017) biodat_sample_#460
	Wood bottom ash	1	1.4	24	0.02	25	19		330			4.9	150	7600						10	840	(ECN, 2017) biodat_sample_#468
	Wood bottom ash	1	6.5	47	0.69	13	70	6	160			7.2	110	13000						15	1900	(ECN, 2017) biodat_sample_#469
	mean		***1.8***	***64***	***0.2***	***23***	***24***	***15***	***91***	***1221***	***0.6***	***7.8***	***75***	***4669***	***5.8***				***236***	***21.6***	***781***	*n =62*
	median		***0.7***	***50***	***0.0***	***19***	***14***	***8***	***74***	***1200***	***0.6***	***7.0***	***59***	***4100***	***5.8***				***229***	***19.0***	***697***	
	minimum		***0.1***	***15***	***0.0***	***3***	***6***	***3***	***7***	***757***	***0.5***	***2.7***	***28***	***900***	***4.6***				***222***	***6.9***	***106***	
	10th precentile		***0.1***	***30***	***0.0***	***5***	***8***	***3***	***29***	***817***	***0.5***	***4.0***	***36***	***2800***	***4.8***				***225***	***9.7***	***164***	
	90th percentile		***5.1***	***105***	***0.7***	***46***	***58***	***26***	***179***	***1621***	***0.6***	***13.0***	***112***	***7200***	***6.8***				***254***	***33.4***	***1510***	
	maximum		***15.0***	***360***	***0.8***	***65***	***116***	***72***	***330***	***1990***	***0.6***	***26.0***	***530***	***13000***	***7.0***				***255***	***74.0***	***2900***	
	coefficient of variation		***1.9***	***0.8***	***1.5***	***0.7***	***1.0***	***1.2***	***0.8***	***0.3***	***0.1***	***0.5***	***1.0***	***0.4***	***0.3***				***0.1***	***0.6***	***0.8***	
	Wood fly ash																					
	Wood fly ash	1	34	35	0.26	30	34	11	404	109		16	133	15200	4.6					17	3660	(ECN, 2017) biodat_sample_#326
	Wood fly ash	1	6.7	13	0.33	13	42	4.7	151			6.3	48	8400	1.1					35	530	(ECN, 2017) biodat_sample_#332
	Wood fly ash	1	8.3	49	0.37	23	91	29	171	2060		11	76	7700						45	1120	(ECN, 2017) biodat_sample_#333
	Wood fly ash	1	10	56	0.36	31	91	17	222	2090		13	81	7800						43	1370	(ECN, 2017) biodat_sample_#334
	Wood fly ash	1	26	73	0.22	35	209	22	480	3970		13	226	23000						16	3420	(ECN, 2017) biodat_sample_#335
	Wood fly ash	1	5.95	136	0.815	58.7	171	34.6	141	1900	1.18	19.9	137	10000	7.51				613	76.3	1900	(ECN, 2017) biodat_sample_#352
	Wood fly ash	1	4.6	104	0.309	33.2	108	53.2	140	1620	1.81	13	121	7100	14.2				589	48.1	1330	(ECN, 2017) biodat_sample_#353
	Wood fly ash	1	5.93	56	0.742	52	124	45.6	191	1860	1.79	17.3	82.8	8300	12.6				555	38.3	1600	(ECN, 2017) biodat_sample_#354

Input material		n	Cd	Cr (total)	Hg	Ni	Pb	As	B	Ba	Be	Co	Cu	Mn	Mo	Sb	Se	Sn	Sr	V	Zn	
	Wood fly ash	1	4.75	78.1	0.57	51.2	225	44.3	144	1430	2.17	13.5	106	4700	11.5				538	55.5	1670	(ECN, 2017) biodat_sample_#355
	Wood fly ash	1	4.97	119	0.47	44.9	221	215	151	1460	1.51	10.1	135	4500	6.01				471	61	1570	(ECN, 2017) biodat_sample_#356
	Wood fly ash	1	4.43	178	0.563	47.6	218	83.2	153	1910	1.3	15.2	193	5900	7.62				483	71.6	2530	(ECN, 2017) biodat_sample_#357
	Wood fly ash	1	5.93	232	0.512	68.9	355	90.4		1620	1.59	26.7	205	4900	11				451	75.7	2590	(ECN, 2017) biodat_sample_#358
	Wood fly ash	1	5.13	172	0.453	63.2	257	60.7	154	1500	1.6	22.8	160	5100	8.17				449	66.6	1940	(ECN, 2017) biodat_sample_#359
	Wood fly ash	1	7.5	149	0.818	56.6	245	48.5	144	2060	1.16	19.8	168	9700	8.95				615	70.5	2400	(ECN, 2017) biodat_sample_#360
	Wood fly ash	1	5.76	129	0.821	53.3	184	33.6	97.2	1720	1.23	18.9	134	8200	6.28				549	71.1	1790	(ECN, 2017) biodat_sample_#361
	Wood fly ash	1	5.82	112	0.737	59.1	170	32.2	135	1830	1.11	21.2	122	10100	6.48				598	72.8	1710	(ECN, 2017) biodat_sample_#363
	Wood fly ash	1	5.46	99.2	0.441	50.1	131	20.3	96.3	1700	1.22	17.9	98.7	9300	6				542	70.9	1500	(ECN, 2017) biodat_sample_#364
	Wood fly ash	1	4.33	88.3	0.484	48	101	17.2	70.6	1540	1.38	17	81.4	8200	6				498	72.3	1300	(ECN, 2017) biodat_sample_#365
	Wood fly ash	1	4.73	99.7	1.38	60.7	124	19.4	117	1630	1.36	19.3	125	9900	6.96				520	72.8	1590	(ECN, 2017) biodat_sample_#366
	Wood fly ash	1	6.17	106	0.822	62.7	191	27.9	183	1690	1.34	20.5	155	10100	8.18				588	77.5	1960	(ECN, 2017) biodat_sample_#367
	Wood fly ash	1	5.32	95.4	0.8	61.6	143	27.2	161	1720	1.29	38.1	125	9000	7.52				583	74.8	1320	(ECN, 2017) biodat_sample_#368
	Wood fly ash	1	13	28	0.95	63	71	4.4	800	2300		7.8	360	24400						20	4300	(ECN, 2017) biodat_sample_#377
	Wood fly ash	1	17.9	57.4	1.07	45.2	258	12		2600		13.1	137	29000					974	39.2	3100	(ECN, 2017) biodat_sample_#378
	Wood fly ash	1	22.1	144	1.21	60.3	386	88.2		2680		16.4	207	29400	6.41				930	36.6	4120	(ECN, 2017) biodat_sample_#380
	Wood fly ash	1	22	291	0.769	55.6	573	176		3490		19.1	315	26600	9.69				895	37.8	5810	(ECN, 2017) biodat_sample_#382
	Wood fly ash	1	21	225	0.906	60.5	510	115		3110		18.8	264	26700	6.69				866	40.4	4410	(ECN, 2017) biodat_sample_#384
	Wood fly ash	1	17.1	266	0.939	54.3	616	160		3160		19.3	301	23400	7.19				795	39.3	5870	(ECN, 2017) biodat_sample_#386
	Wood fly ash	1	19.3	252	0.856	58.6	709	139		3960		24.8	315	20100	8.8				845	39.2	5900	(ECN, 2017) biodat_sample_#388
	Wood fly ash	1	16.4	272	0.992	68.2	688	156		3360		21.7	302	17500	7.64				809	35.2	5120	(ECN, 2017) biodat_sample_#390
	Wood fly ash	1	18	45.8	0.848	33.5	105	39.2		3700		10.4	111	28500					956	27.2	4720	(ECN, 2017) biodat_sample_#392
	Wood fly ash	1	14	12	0.63	25	88	5.2		2800		10	130	17000			2.1			15	2900	(ECN, 2017) biodat_sample_#393
	Wood fly ash	1	10	28	0.35	11	58	8.4	190	1700		5	56	9300	2.9					12	2000	(ECN, 2017) biodat_sample_#394
	Wood fly ash	1	7.53	45.5	0.602	75.1	32.9	8.21		2460		8.09	93.1	13000					836	31.2	3330	(ECN, 2017) biodat_sample_#395
	Wood fly ash	1	6.61	47.4	0.663	122	33	6		2360		10.6	78	11800					757	72.7	2960	(ECN, 2017) biodat_sample_#396
	Wood fly ash	1	6.8	37	0.34	13	52	4.2	130	1400		4.1	52	7100	3					11	1100	(ECN, 2017) biodat_sample_#398
	Wood fly ash	1	12	56	0.21	24	74	5.1	200	1900		8.6	120	13000	4.9					14	2700	(ECN, 2017) biodat_sample_#399
	Wood fly ash	1	9.4	23	0.46	22	82		650	2100		5.3	76	15000	3					11	2400	(ECN, 2017) biodat_sample_#400
	Wood fly ash	1	11	22	0.45	24	110	4.3	480	1700		6.4	91	9900	4.7					14	1900	(ECN, 2017) biodat_sample_#401
	Wood fly ash	1	9.5	32	0.28	25	64		260	2400		6.3	96	17100						12	2800	(ECN, 2017) biodat_sample_#402
	Wood fly ash	1	6	113	0.62	97	61	21	174	1560		21	130	6700	12.5					90	1060	(ECN, 2017) biodat_sample_#403
	Wood fly ash	1	1.6	88	0.2	74	38	15	72	1240		14	150	2700	7.3					89	1050	(ECN, 2017) biodat_sample_#405
	Wood fly ash	1	9.2	109	1.1	77	105	29	313	2320		17	180	8700	11					50	2290	(ECN, 2017) biodat_sample_#407
	Wood fly ash	1	7.9	71	1.1	84	85	48	205	1930		19	90	7500	11					48	1610	(ECN, 2017) biodat_sample_#409
	Wood fly ash	1	10	464	2.51	69.8	514	148	249	2000	0.883	20.5	352	10500	13.1				636	74.3	3280	(ECN, 2017) biodat_sample_#411

Input material	n	Cd	Cr (total)	Hg	Ni	Pb	As	B	Ba	Be	Co	Cu	Mn	Mo	Sb	Se	Sn	Sr	V	Zn	
Wood fly ash	1	69	251	0.42	23	234	9.4	428	1830		8.2	894	14000						5.8	13300	(ECN, 2017) biodat_sample_#421
Wood fly ash	1	75	144	0.24	27	226	23	372	1930		10	794	15000	11					8.5	13800	(ECN, 2017) biodat_sample_#422
Wood fly ash	1	6.9	147	0.31	33	462	86	115	1750		9.2	375	3200						28	2540	(ECN, 2017) biodat_sample_#424
Wood fly ash	1		155	0.01	15	76	54	45	1730		5.2	367	2200						21	3310	(ECN, 2017) biodat_sample_#425
Wood fly ash	1	24	61	0.61	41	130	14	370	2100		14	140	9700	8.1	2.5				37	5900	(ECN, 2017) biodat_sample_#444
Wood fly ash	1	16	54	0.55	27	94	9.4	320	1700		11	95	7900	4.5					28	2800	(ECN, 2017) biodat_sample_#445
Wood fly ash	1	19	140	0.27	48	82	13	260	2100		12	110	9100	17					29	2700	(ECN, 2017) biodat_sample_#446
Wood fly ash	1	24	160	0.24	67	91	10	310	2100		11	120	9600	27					27	3400	(ECN, 2017) biodat_sample_#447
Wood fly ash	1	6.8	57		36	33	8.7				8.3	72	6500						25	980	(ECN, 2017) biodat_sample_#448
Wood fly ash	1	27	150	0.39	49	90	17	330				120							32	4100	(ECN, 2017) biodat_sample_#449
Wood fly ash	1	29	44	0.21	20	75	7.5	370				130							13	5300	(ECN, 2017) biodat_sample_#450
Wood fly ash	1	32	69	0.18	31	69	5.9	400				120							23	3000	(ECN, 2017) biodat_sample_#451
Wood fly ash	1	32	26	0.13	31	68		450				150							15	7000	(ECN, 2017) biodat_sample_#452
Wood fly ash	1	26	44	0.28	23	37	13	440				140							16	7800	(ECN, 2017) biodat_sample_#453
Wood fly ash	1	14	109	0.17	78	67		264	1500		10	109	7700						36	3360	(ECN, 2017) biodat_sample_#454
Wood fly ash	1	24	96	0.06	94	152	7.8	439	1820		10	158	6600	17					39	16500	(ECN, 2017) biodat_sample_#456
Wood fly ash	1	12	92		67	52	3.9	320			10	84	7100						25	3310	(ECN, 2017) biodat_sample_#457
Wood fly ash	1		155		60	10		127	1320		10	67	5200	8.3					49	388	(ECN, 2017) biodat_sample_#461
mean		***14.9***	***110.7***	***0.6***	***49***	***174***	***42***	***257***	***2065***	***1.4***	***14***	***174***	***11681***	***8.6***	***2.5***	***2.1***		***664***	***41***	***3436***	*n =104*
median		***10.0***	***95.7***	***0.5***	***50***	***105***	***21***	***200***	***1900***	***1.3***	***13***	***130***	***9300***	***7.6***	***2.5***	***2.1***		***598***	***37***	***2700***	
minimum		***1.6***	***12.0***	***0.0***	***11***	***10***	***4***	***45***	***109***	***0.9***	***4***	***48***	***2200***	***1.1***	***2.5***	***2.1***		***449***	***6***	***388***	
10th precentile		***4.9***	***28.4***	***0.2***	***23***	***38***	***5***	***111***	***1472***	***1.1***	***6***	***76***	***5020***	***4.5***	***2.5***	***2.1***		***478***	***13***	***1138***	
90th percentile		***27.2***	***231.3***	***1.0***	***75***	***454***	***125***	***442***	***3145***	***1.8***	***21***	***315***	***23800***	***13.1***	***2.5***	***2.1***		***909***	***74***	***5897***	
maximum		***75.0***	***464.0***	***2.5***	***122***	***709***	***215***	***800***	***3970***	***2.2***	***38***	***894***	***29400***	***27.0***	***2.5***	***2.1***		***974***	***90***	***16500***	
coefficient of variation		***0.9***	***0.8***	***0.7***	***0.5***	***1.0***	***1.2***	***0.6***	***0.3***	***0.2***	***0.5***	***0.9***	***0.6***	***0.5***				***0.3***	***0.6***	***0.9***	
Wood (unknown)																					
Wood ash	1	3.5	91	0.006	39	6.5	3.2	102	779		9.7	58	9050	5.7					16	497	(ECN, 2017) biodat_sample_#327
Wood ash	1	14	160	0.29	79	120	4.7	360	2300		11	170	18000	9.6					30	2100	(ECN, 2017) biodat_sample_#331
Wood ash	1	3.3	39	0.012	17	26	1.8	220			7.1	100	9400						14	870	(ECN, 2017) biodat_sample_#362
Wood ash	1	12	29	1.1	14	80	8	220			4.1	100	11000						12	2300	(ECN, 2017) biodat_sample_#369
Wood ash	1	11	26	0.47	17	36		110			11	59	5000						34	1500	(ECN, 2017) biodat_sample_#370
Wood ash	1	2.2	31	0.08	13	17	3.2	200			5.1	68	7800						13	770	(ECN, 2017) biodat_sample_#371
Wood ash	1	5.8	41	0.36	25	59	2.5	340			8.2	160	22000						20	1900	(ECN, 2017) biodat_sample_#372
Wood ash	1	5.3	24	0.37	15	49	2.8	240			5.9	110	13000						11	1300	(ECN, 2017) biodat_sample_#373
Wood ash	1	15	49	0.43	21	120	4.2	210			13	120	9800						28	3400	(ECN, 2017) biodat_sample_#374

Input material	n	Cd	Cr (total)	Hg	Ni	Pb	As	B	Ba	Be	Co	Cu	Mn	Mo	Sb	Se	Sn	Sr	V	Zn	
Wood ash	1	4.3	41	0.27	16	44		200			5.1	180	17000						13	1500	(ECN, 2017) biodat_sample_#375
Wood ash	1	21	44	0.97	20	200	5.8	190			6.4	230	15000						13	5600	(ECN, 2017) biodat_sample_#376
Wood ash	1	2.3	10		20	17	6	260	2300		7.2	110	8100	3.2					7.9	210	(ECN, 2017) biodat_sample_#426
Wood ash	1	11	39	0.08	47	47					18	140	21000								(ECN, 2017) biodat_sample_#462
Wood ash	1	12	29	0.163	28	45					11	134	14200								(ECN, 2017) biodat_sample_#463
Wood ash	1	4.9	21	0.25	12	39					6.9	58	7400								(ECN, 2017) biodat_sample_#464
Wood ash	1	6.5	23	0.28	13	60					7.5	72	5600								(ECN, 2017) biodat_sample_#465
Wood ash	1	6.5	40	0.43	14	59					5.9	130	11000								(ECN, 2017) biodat_sample_#466
Wood ash	1	11	22	0.38	12	60					6.8	53	4140								(ECN, 2017) biodat_sample_#467
Wood (unknown)	1	6.2			100.6	5.7	18.3		2324	1.1	13	193	23562		0.78	0.9	1.6	2178	37.1	1086	(Thy et al., 2008)
Wood (unknown)	1	4.5			99	2.9	10.7		2476	1.2	14	215	26451		0.79	0.4	1.6	2692	36.1	1112	(Thy et al., 2008)
Wood (unknown)	1	2.1			103.5	0.3	17.8		2604	1.1	14	201	26039		0.82	0.9	1.6	2450	40.9	1172	(Thy et al., 2008)
Wood (unknown)	1	7.1			85.1	1.8	17.5		2390	1.0	10	183	22215		0.92	1	1.4	2263	35.2	997	(Thy et al., 2008)
Wood (unknown)	1	7.5			71.2	4.9	13.6		1811	1.0	9	166	19745		0.78	1.2	1.2	2116	30.2	870	(Thy et al., 2008)
Wood (unknown)	1	7.5			92	9.5	16.5		2201	1.0	9	162	17799		0.66	1.1	1.1	2128	30.5	806	(Thy et al., 2008)
Wood (unknown)	1	8.9			72.8	12.2	19.4		2380	1.1	10	197	22063		0.71	1.3	1.3	2382	35.8	992	(Thy et al., 2008)
Wood (unknown)	1	8.4			61.5	10.3	17.8		2098	1.1	9	176	19581		0.67	1	1.1	2110	32.3	968	(Thy et al., 2008)
Wood (unknown)	1	7.8			64.3	11.3	18.6		2219	1.0	9	171	17937		0.6	1.2	1.2	2086	32.5	875	(Thy et al., 2008)
Wood (unknown)	1	7.5			59.6	10.7	17.6		2210	1.0	8	160	17731		0.57	0.7	1.1	2095	30.8	812	(Thy et al., 2008)
Wood (unknown)	1	5.9			43.1	8.6	9.6		1731	1.0	6	117	13298		1.27		0.9	1844	21.1	603	(Thy et al., 2008)
Balsam bark	1												20160								(Bryers, 1996)
beech wood	1	16.6	76.2	1.1	34.6	325	16.9				9.6	358	90.5	15.7	63.5	85.8	78.9		23.6		(Demirbas, 2005)
Beech bark	1												3100								(Bryers, 1996)
Birch bark	2												22870								(Bryers, 1996)
Elm bark	1												775								(Bryers, 1996)
Eucalyptus bark	1												10850								(Theis et al., 2006)
Fir mill residue	2												13640								(Miles et al., 1995) (Thy et al., 2008)
Forest residue	3												13180								(Miles et al., 1995; Miles et al., 1996; Zevenhoven-Or
Hemlock bark	1												9300								(Bryers, 1996)
Maple bark	2												5430								(Bryers, 1996)
Oak wood	2												14900								(Misra et al., 1993; Demirbas, 2004)
Pine bark	2												12400								(Bryers, 1996; Moilanen, 2006)
Pine chips	1								558.0				2090							1495.0	(Masia et al., 2007)
Pine sawdust	2												10550								(Etiegni and Campbell, 1991; Moilanen, 2006)
Poplar	3												4500								(Misra et al., 1993; Miles et al., 1995)
Poplar bark	1												2330								(Bryers, 1996)

Input material		n	Cd	Cr (total)	Hg	Ni	Pb	As	B	Ba	Be	Co	Cu	Mn	Mo	Sb	Se	Sn	Sr	V	Zn	
	Sawdust	2												27910								(Tillman, 2000; Wigley et al., 2007)
	Spruce bark	1												13950								(Demirbas, 2005)
	Tamarack bark	1												26360								(Bryers, 1996)
	Wood (unknown)	1												35740								(Wei et al., 2005)
	Wood (unknown)	1																				(Zevenhoven et al., 2012)
	Wood (unknown)	1												98								(Zevenhoven et al., 2012)
	Wood (unknown)	1												100								(Zevenhoven et al., 2012)
	Wood (unknown)	1												0								(Zevenhoven et al., 2012)
	Wood (unknown)	1												87								(Zevenhoven et al., 2012)
	Wood (unknown)	1												167								(Zevenhoven et al., 2012)
	Wood (unknown)	1												92								(Zevenhoven et al., 2012)
	Wood (unknown)	1												166								(Zevenhoven et al., 2012)
	Wood (unknown)	1												112								(Zevenhoven et al., 2012)
	Wood (unknown)	1												26								(Zevenhoven et al., 2012)
	Forest residue (unknown)	1												270								(Zevenhoven et al., 2012)
	Forest residue (unknown)	1												164								(Zevenhoven et al., 2012)
	Forest residue (unknown)	1												408								(Zevenhoven et al., 2012)
	Forest residue (unknown)	1												545								(Zevenhoven et al., 2012)
	Forest residue (unknown)	1												440								(Zevenhoven et al., 2012)
	Forest residue (unknown)	1												440								(Zevenhoven et al., 2012)
	Forest residue (unknown)	1												370								(Zevenhoven et al., 2012)
	Forest residue (unknown)	1												290								(Zevenhoven et al., 2012)
	Forest residue (unknown)	1												300								(Zevenhoven et al., 2012)
	Forest residue (unknown)	1												330								(Zevenhoven et al., 2012)
	Forest residue (fly ash)	1	25.0	290.0	1.7	47.0	76.0	4.0		4260.0		13.0	200	20000						39.0	3630.0	(Pöykiö et al., 2014)
	Forest residue (bootom ash)	1	5.7	318.0		36.0	29.0	14.0		2210.0		11.0	196.0	15600						41.0	950.0	(Pöykiö et al., 2009)
	Forest residue fly-ash (sawdust and bark)	1	19	92	0.8	31.0	352	19	263	484		39	11	8400	24.0	10.0			284	36		(Lanzerstorfer, 2015)
	Forest residue fly-ash (chips; 80% softwood)	1	105	76	0.1	23.0	602	36	221	91		39	140	4100	43.0	12.0			242	38		(Lanzerstorfer, 2015)
	Forest residue fly-ash (chips; 80% softwood)	1	41	87	1.7	27.0	250	27	671	191		25	145	14600	5.0	10.0			641	10		(Lanzerstorfer, 2015)
	Forest residue fly ash (chips; 90% softwood)	1	77	70	4.2	4.0	892	62	292	136		25	156	6800	5.0	10.0			283	10		(Lanzerstorfer, 2015)
	Forest residue fly-ash (rubber tree; 95% chips, 5% bark)	1	9	15	0.1	31.0	53	15	227	357		34	5	2900	27.0	10.0			461	30		(Lanzerstorfer, 2015)
	Forest residue fly-ash (90% chips, 10% horse dung)	1	32	41	0.1	6.0	228	19	114	144		34	5	1000	32.0	11.0			141	10		(Lanzerstorfer, 2015)
	Wood bark (unknown)	1												274								(Zevenhoven et al., 2012)
	Wood bark (unknown)	1												300								(Zevenhoven et al., 2012)
	Wood bark (unknown)	1												0								(Zevenhoven et al., 2012)
	Wood bark (unknown)	1												4210								(Zevenhoven et al., 2012)

Input material	n	Cd	Cr (total)	Hg	Ni	Pb	As	B	Ba	Be	Co	Cu	Mn	Mo	Sb	Se	Sn	Sr	V	Zn	
Wood bark (unknown)	1												1630								(Zevenhoven et al., 2012)
Wood bark (unknown)	1												99								(Zevenhoven et al., 2012)
Wood bark (unknown)	1												500								(Zevenhoven et al., 2012)
Wood bark (unknown)	1												550								(Zevenhoven et al., 2012)
Wood bark (unknown)	1												270								(Zevenhoven et al., 2012)
Wood bark (unknown)	1												530								(Zevenhoven et al., 2012)
Wood bark (unknown)	1												470								(Zevenhoven et al., 2012)
Wood bark (unknown)	1												290								(Zevenhoven et al., 2012)
Wood bark (unknown)	1												202								(Zevenhoven et al., 2012)
Wood bark (unknown)	1												428								(Zevenhoven et al., 2012)
Wood bark (unknown)	1												532								(Zevenhoven et al., 2012)
Wood bark (unknown)	1												465								(Zevenhoven et al., 2012)
bark, sawdust and offcuts	1		38.8	0.5	74.0	160	7.4	438.0	2600		17	144	11040		4			741	24	9100	(Kröppl et al., 2011)
bark, trimmings and offcuts, left over wood, sawdust	1		33.2	2.9	81.5	213	32.8	194.0	1400		9	100	10900		1			516	28	4100	(Kröppl et al., 2011)
raw wood	1		5.0	0.9	15.8	13.7	5.0	20.0	275.0		5.0	5.0	2000.0		1.0			105.0	2.0	330.0	(Kröppl et al., 2011)
mean		***14.6***	***96.9***	***1.3***	***34***	***261***	***22***	***271***	***1059***		***23***	***101***	***5174***	***22.7***	***7.6***			***379***	***24***	***3268***	*n = 107*
median		***7.7***	***70.0***	***0.9***	***31***	***213***	***19***	***227***	***421***		***25***	***140***	***532***	***25.5***	***10.0***			***284***	***28***	***2563***	
minimum		***2.1***	***5.0***	***0.1***	***4***	***14***	***4***	***20***	***91***		***5***	***5***	***0***	***5.0***	***1.0***			***105***	***2***	***330***	
10th precentile		***3.4***	***15.0***	***0.1***	***6***	***29***	***5***	***95***	***137***		***9***	***5***	***99***	***5.0***	***1.0***			***134***	***10***	***640***	
90th percentile		***27.1***	***290.0***	***3.0***	***74***	***602***	***36***	***485***	***2561***		***39***	***196***	***14600***	***37.5***	***11.2***			***661***	***39***	***6600***	
maximum		***105.0***	***318.0***	***4.2***	***82***	***892***	***62***	***671***	***4260***		***39***	***200***	***35740***	***43.0***	***12.0***			***741***	***41***	***9100***	
coefficient of variation		***1.4***	***1.1***	***1.1***	***0.7***	***1.0***	***0.8***	***0.7***	***1.2***		***0.5***	***0.8***	***1.5***	***0.7***				***0.6***	***0.6***	***1.0***	
grass																					
Miscanthus	4												3100								(Miles et al., 1995; Moilanen, 2006; Wigley et al., 200
straw																					
wheat	1	3	22	0.3	5	20	7	17	271		25	5	700	5	10			55	10	238	(Lanzerstorfer, 2015)
wheat	1	8	7	0.4	5	20	10	103	33		33	5	40	46	10			25	13	325	(Lanzerstorfer, 2015)
Corn	1								338				620							374	(Masia et al., 2007)
Oat	1												775								(Theis et al., 2006)
Rape	1								338				310							249	(Masia et al., 2007)
Rice	3												2790								(Miles et al., 1995; Thy et al., 2000; Thy et al., 2006)
Straw	2												155								(Wieck-Hansen et al., 2000; Wei et al., 2005)
straw ash	1	0.1	4.7	0.0	3.7	1.0														81.0	(Schiemenz et al., 2011)
Wheat	14												540								(Risnes et al., 2003; Moilanen, 2006; Thy et al., 2006

Input material	n	Cd	Cr (total)	Hg	Ni	Pb	As	B	Ba	Be	Co	Cu	Mn	Mo	Sb	Se	Sn	Sr	V	Zn	
																					(Miles et al., 1995; Bryers, 1996; Miles et al., 1996; D
Wheat (unknown)	1	1.4			14.3		3.8		684	0.9	2	38	752			3	0.4		12.8	65	(Thy et al., 2008)
Wheat (unknown)	1	1.9			31.5		4.4		786	0.9	6	68	896			0.8	2		16.8	94	(Thy et al., 2008)
Wheat (unknown)	1	1.8			23.5		4.6		826	1.0	7	56	915			0.9	0.9		18.6	98	(Thy et al., 2008)
Wheat (unknown)	1	1.6			22.3		4.5		741	0.9	5	50	842			1.1	1.2		15.3	80	(Thy et al., 2008)
Wheat (unknown)	1	1.7			20.5		4.5		800	0.9	9	48	851			1.5	0.9		16.6	77	(Thy et al., 2008)
Wheat (unknown)	1	2			19.1		5		783	0.9	6	52	892			1.7	20.6		17	85	(Thy et al., 2008)
Wheat (unknown)	1	1.4			19.6		4.3		750	0.9	3	52	822			2.7	20.7		16.5	77	(Thy et al., 2008)
Wheat (unknown)	1	1.8			17.1		5.6		711	0.9	17	76	811			2.7	0.6		14.6	94	(Thy et al., 2008)
Wheat (unknown)	1	1.4			17.6		4.1		699	0.9	2	42	779			2.7	0.7		15.4	70	(Thy et al., 2008)
Wheat (unknown)	1	1.2			19		11.8		647	0.9	2	32	732			2.5	0.4		17.7	61	(Thy et al., 2008)
Straw (unknown)	1												200								(Zevenhoven et al., 2012)
Straw (unknown)	1												0								(Zevenhoven et al., 2012)
Straw (unknown)	1												55								(Zevenhoven et al., 2012)
Straw (unknown)	1												66								(Zevenhoven et al., 2012)
Straw (unknown)	1												11								(Zevenhoven et al., 2012)
Straw (unknown)	1												9								(Zevenhoven et al., 2012)
Rice (unknown)	1	0.2			9.9		3.1		131	1.1	7	20	4348				0.4		4	162	(Thy et al., 2008)
Rice (unknown)	1	0.1			10.4		2.8		118	1.1	7	29	3899			0.1	0.8		3.3	184	(Thy et al., 2008)
Rice (unknown)	1	0.3			11.8		4		129	1.1	7	26	4127			0.1	0.5		4.1	192	(Thy et al., 2008)
Rice (unknown)	1	0.3			16.7		6		118	1.1	6	24	3726				0.4		3.6	185	(Thy et al., 2008)
Rice (unknown)	1	0.3			13		6.6		113	0.8	7	41	4054			0.1	0.4		3.4	201	(Thy et al., 2008)
Rice (unknown)	1	0.3			7		6.9		118	0.9	6	55	4201				0.4		3.4	218	(Thy et al., 2008)
Rice (unknown)	1	0.5			7.9		5.6		98	0.8	5	37	3559				0.3		2.8	175	(Thy et al., 2008)
Rice (unknown)	1	1			7		6.7		114	0.9	6	32	4175				0.5		3.4	201	(Thy et al., 2008)
Rice (unknown)	1	0.3			12.6		6		107	0.8	4	32	3798				0.3		3.1	187	(Thy et al., 2008)
Rice (unknown)	1	0.4			8.6		6.5		109	0.8	2	60	3921				0.4		3.3	213	(Thy et al., 2008)
Rice (unknown)	1	0.3			8.6		6.4		114	0.9	1	32	4001				0.3		3.2	197	(Thy et al., 2008)
Rice (unknown)	1	0.2			11.1		6.2		104	0.8	1	54	3879				0.4		3.2	211	(Thy et al., 2008)
Rice (unknown)	1	0.3			7.1		5.8		102	0.9	1	33	3557				0.3		2.8	172	(Thy et al., 2008)
Rice (unknown)	1	0.2			5.5		6.1		96	0.9	1	15	3675				0.3		3.3	161	(Thy et al., 2008)
mean		***1.2***	***11.2***	***0.2***	***13.2***	***14***	***5.7***	***60***	***356***	***0.9***	***6.8***	***39.0***	***1829***	***25.5***	***10.0***	***1.5***	***2.3***	***40***	***8.9***	***163***	*n =104*
median		***0.5***	***7.0***	***0.3***	***11.8***	***20***	***5.7***	***60***	***130***	***0.9***	***6.0***	***37.5***	***847***	***25.5***	***10.0***	***1.5***	***0.4***	***40***	***4.1***	***175***	
minimum		***0.1***	***4.7***	***0.0***	***3.7***	***1***	***2.8***	***17***	***33***	***0.8***	***1.0***	***5.0***	***0***	***5.0***	***10.0***	***0.1***	***0.3***	***25***	***2.8***	***61***	
10th precentile		***0.2***	***5.2***	***0.1***	***5.3***	***5***	***3.9***	***26***	***101***	***0.8***	***1.0***	***17.5***	***51***	***9.1***	***10.0***	***0.1***	***0.3***	***28***	***3.2***	***76***	
90th percentile		***1.9***	***19.0***	***0.4***	***21.2***	***20***	***7.0***	***94***	***784***	***1.1***	***13.0***	***58.0***	***4076***	***41.9***	***10.0***	***2.7***	***1.8***	***52***	***16.9***	***240***	

Input material	n	Cd	Cr (total)	Hg	Ni	Pb	As	B	Ba	Be	Co	Cu	Mn	Mo	Sb	Se	Sn	Sr	V	Zn	
maximum		***8.0***	***22.0***	***0.4***	***31.5***	***20***	***11.8***	***103***	***826***	***1.1***	***33.0***	***76.0***	***4348***	***46.0***	***10.0***	***3.0***	***20.7***	***55***	***18.6***	***374***	
coefficient of variation		***1.3***	***0.8***	***0.8***	***0.5***	***0.8***	***0.3***	***1.0***	***0.8***	***0.1***	***1.1***	***0.5***	***0.9***	***1.1***	***0.0***	***0.7***	***2.5***	***0.5***	***0.7***	***0.5***	
other residues																					
hazelnut shell	1	11	58	0.5	32.6	181	6.9				7.5	416	86.3	11.4	51.5	42	62		14.6	1180	(Demirbas, 2005)
wheat straw	1	9.2	62.5	0.4	30.4	184	7.2				4.2	284	64.1	15.5	56.3	58.5	82		11.2	2680	(Demirbas, 2005)
olive husk	1	12.8	35.2	0.3	26.8	226	8.7				2.8	197	73.5	8.1	50.2	76.6	44.6		16.4	3870	(Demirbas, 2005)
waslnut shell	1	8.4	70.4	0.1	32.7	214	6.4				5	326	48.5	13.7	45.1	67.1	79		12.8	1850	(Demirbas, 2005)
almond shell	1	6.5	18.6	0.2	21.5	132	7.5				3.6	174	29.7	6.8	65.4	49.9	53.6		9.4	1250	(Demirbas, 2005)
sunflower shell	1	7.1	47.3	0.4	29.5	168	5.1				7.1	261	34.6	10.9	37.2	56.4	64.5		15	668	(Demirbas, 2005)
mustard stalks	1	7	61	0.1	35	26	0.1		376			113	77	10		0.1		885	59	161	(Singh et al., 2011b)
Olive residue	1								112.0				310.0							249.0	(Masia et al., 2007)
Palm kernels	1								<112				4570.0							748.0	(Masia et al., 2007)
Pepper plant	1												1320.0								(Masia et al., 2007)
Rice husks	5												155.0								(Umantaheswaran and Batra, 2008; Madhiyanon et a
																					(Bryers, 1996; Feng et al., 2004)
Rice waste	1												660.0								(Zevenhoven et al., 2012)
Rice waste	1												176.0								(Zevenhoven et al., 2012)
Rice waste	1												182.0								(Zevenhoven et al., 2012)
oil residue	1												12.0								(Zevenhoven et al., 2012)
oil residue	1												25.0								(Zevenhoven et al., 2012)
oil residue	1												60.0								(Zevenhoven et al., 2012)
shell seeds and hulls	1												0.0								(Zevenhoven et al., 2012)
shell seeds and hulls	1												12.0								(Zevenhoven et al., 2012)
shell seeds and hulls	1												4.0								(Zevenhoven et al., 2012)
bagasse	1												48.0								(Zevenhoven et al., 2012)
bagasse	1												43.0								(Zevenhoven et al., 2012)
rape meal ash	1	0.5	228.0	0.0	274.0	11.9														249.0	(Schiemenz et al., 2011)
cereal ash	1	1.3	13.7	0.0	13.1	2.6														750.0	(Schiemenz et al., 2011)
unknown plant origin																					
unknown plant origin	24																				STRUBIAS - ECOFI contribution
animal biomass																					
poultry manure																					
poultry manure	1	1.8	25.3	0.0	21.6	11.9	3.3					585.0								2379.0	STRUBIAS contribution - IE

Input material		n	Cd	Cr (total)	Hg	Ni	Pb	As	B	Ba	Be	Co	Cu	Mn	Mo	Sb	Se	Sn	Sr	V	Zn	
	poultry manure	1								<112											998.0	(Masia et al., 2007; Tortosa Masiá et al., 2007)
	poultry manure	1	0.2	4.5	0.0	5.0	0.9	0.3					69.7	4395.0						2.6	389.7	STRUBIAS contribution - confidential data provider
	poultry manure	1	1.9	23.0	0.0	31.2	11.3	2.7					526.0	24.0						23.9	2494.0	STRUBIAS contribution - confidential data provider
	poultry manure	1							110.0			5.6	333.0	1950.0	12.1						1621.0	STRUBIAS contribution - confidential data provider
	poultry manure	1	3.4	16.2	0.0	18.4	43.6	4.5					216.0							9.2	1652.0	STRUBIAS contribution - confidential data provider
	poultry manure	?							130.0			2.0	300.0	1200.0	5.0		2.0					STRUBIAS contribution - ESSP (Fibrphos)
	poultty manure	415																				STRUBIAS contribution - ESSP (BMC Moerdijk, Biller
	poultry manure	1		24.0						72.0			176.0	1196							991.0	(Staroń et al., 2016)
	poultry manure	1											165.0	209			2.4				136.0	(Blake and Hess, 2014)
	poultry manure	1	2.0	14.3		42.0	3.7	1.0				7.0	553.0	4143	96.0					12.0	3795.0	(Lynch et al., 2014)
	poultry manure	1		112.0		10.0							71.0	596							209.0	(Abelha et al., 2003)
	poultry manure	1	0.4			14.8	6.0	15.0					43.1	1600							600.0	(Codling et al., 2002)
	poultry manure	1							249.0				1222.0	3120							2670.0	(Reiter and Middleton, 2016)
	poultry manure	1							139.0				1089.0	2160							1510.0	(Reiter and Middleton, 2016)
	poultry manure	1							221.0				3429.0	4600							2888.0	(Reiter and Middleton, 2016)
	poultry manure	1							234.0				1861.0	2940							2515.0	(Reiter and Middleton, 2016)
	poultry manure	1							98.0				809.0	860							2879.0	(Reiter and Middleton, 2016)
	poultry manure	1							242.0				3252.0	2450							1793.0	(Reiter and Middleton, 2016)
mean			***1.6***	***31***	***0.0***	***20***	***13***	***4.5***	***178***	***72***		***4.9***	***865***	***2096***	***38***		***2.2***			***12***	***1736***	*n =440*
median			***1.9***	***23***	***0.0***	***18***	***9***	***3.0***	***180***	***72***		***5.6***	***526***	***1950***	***12***		***2.2***			***11***	***1652***	
minimum			***0.2***	***4***	***0.0***	***5***	***1***	***0.3***	***98***	***72***		***2.0***	***43***	***24***	***5***		***2.0***			***3***	***136***	
10th precentile			***0.3***	***10***	***0.0***	***8***	***2***	***0.6***	***106***	***72***		***2.7***	***70***	***364***	***6***		***2.0***			***5***	***317***	
90th percentile			***2.7***	***60***	***0.0***	***36***	***28***	***9.7***	***244***	***72***		***6.7***	***2417***	***4294***	***79***		***2.4***			***20***	***2883***	
maximum			***3.4***	***112***	***0.0***	***42***	***44***	***15.0***	***249***	***72***		***7.0***	***3429***	***4600***	***96***		***2.4***			***24***	***3795***	
coefficient of variation			***0.7***	***1.2***	***0.4***	***0.6***	***1.2***	***1.2***	***0.4***			***0.5***	***1.2***	***1***	***1***		***0.1***			***0.7***	***0.6***	
slaughterhouse waste																						
	meat and bone meal	1								<112											623.0	(Masia et al., 2007)
	meat and bone meal	1																			0.0	(Deydier et al., 2005b)
	meat and bone meal (bottom ash)	1	0.4	50.0		25.0	2.0		5.0				50.0							2.0	100.0	STRUBIAS contribution - ESPP
	slaughterhouse waste (bottom ash)	1	0.3	136.2		93.0	15			208.4		9.5	189.5			8.7		10	273.8	206.1	262.0	(Coutand et al., 2008)
	slaughterhouse waste (fly ash)	1	1.7	115.3		97.0	15			158.0		73.5	133.2			12.1		23	104.1	177.6	1349.0	(Coutand et al., 2008)
	slaughterhouse waste (fly ash washed)	1	0.4	155.3		119.9	15			184.6		24.3	213.7			36.9		48	237.0	197.3	3372.0	(Coutand et al., 2008)
	chicken feathers	1		51.0						112.0			582.0	1770.0							8444.0	(Staroń et al., 2016)
	meat and bone meal	1		36.0						154.0			42.5	76.0							521.0	(Staroń et al., 2016)
	meat and bone meal	1	1.0	13.0	5.0	9.0	15	5.0					270.0								940.0	(Skodras et al., 2006)

Input material	n	Cd	Cr (total)	Hg	Ni	Pb	As	B	Ba	Be	Co	Cu	Mn	Mo	Sb	Se	Sn	Sr	V	Zn	
meat and bone meal	1		5.0	0.0	5.0	5	25.0					5.0	5.0							87.1	(Gulyurtlu et al., 2007)
meat and bone meal (bottom ash)	1	0.3	32.4		17.7	1.3	0.4		43.0	0.9	1.2	70.0		2.7	4.1		1	145.0	3.1	39.2	(Cyr and Ludmann, 2006)
mean		***1.1***	***60***	***0.9***	***41***	***13***	***8.1***	***121***	***120***		***15.7***	***575***	***1587***	***37***		***1.7***			***71***	***1516***	*n =15*
median		***0.6***	***50***	***0.0***	***25***	***15***	***5.0***	***106***	***112***		***6.9***	***133***	***364***	***6***		***2.2***			***20***	***521***	
minimum		***0.2***	***1***	***0.0***	***1***	***1***	***0.3***	***0***	***43***		***0.5***	***1***	***1***	***1***		***0.1***			***1***	***0***	
10th precentile		***0.3***	***6***	***0.0***	***5***	***1***	***0.5***	***2***	***66***		***1.0***	***13***	***3***	***2***		***0.7***			***2***	***16***	
90th percentile		***2.7***	***130***	***2.2***	***97***	***26***	***18.0***	***247***	***187***		***34.1***	***1867***	***4386***	***88***		***2.4***			***198***	***3584***	
maximum		***3.4***	***155***	***5.0***	***120***	***44***	***25.0***	***249***	***208***		***73.5***	***3429***	***4600***	***96***		***2.4***			***206***	***8444***	
coefficient of variation		***1.0***	***0.9***	***2.1***	***1.0***	***1.0***	***1.1***	***0.9***			***1.5***	***1.8***	***1***	***1***		***0.5***			***1.3***	***1.5***	
Contaminated biomass																					
Waste water treatment sludge (raw)																					
waste water treatment sludge (raw)	1																			0.5	(Werther et al., 2000)
waste water treatment sludge (raw)	1																			0.5	(Wei et al., 2005)
waste water treatment sludge (raw)	1																			0.5	STRUBIAS contribution - confidential data provider
waste water treatment sludge (raw)	1	4.1	142.0	0.1	92.0	440.0				0.5	26.0	1300.0	1900.0	36.0	14.0	1.0			50.0	3600.0	STRUBIAS contribution - confidential data provider
wastewater treatment sludge (raw - DE)	252	3.3	267.0	0.8	105.8	151.0	17.5		2173.0		28.1	916.0	1914.0	25.3	23.0	2.5	194.0	578.0	136.0	2535.0	(Krüger and Adam, 2015)
waste water treatment sludge (raw, mono-inc)	191	1.8	66.6	0.1	37.9	50.4	11.1					703.0								1650.0	STRUBIAS contribution - confidential data provider
waste water treatment sludge (raw, mono-inc)	196	1.7	89.0	0.5	742.0	77.0	9.3					48.0								2160.0	STRUBIAS contribution - confidential data provider
waste water treatment fly ash	1	7.0	1047.0	0.0	119.5	138.0	16.2				27.2	665.5		25.5	6.9		364.0			4472.0	(Kasina et al., 2016)
wastewater treatment sludge (raw - FI)	1																				(Zevenhoven et al., 2012)
wastewater treatment sludge (raw - FI)	1												171.0								(Zevenhoven et al., 2012)
wastewater treatment sludge (raw - FI)	1												98.0								(Zevenhoven et al., 2012)
wastewater treatment sludge (raw - FI)	1												50.0								(Zevenhoven et al., 2012)
wastewater treatment sludge (raw - FI)	1												15.0								(Zevenhoven et al., 2012)
wastewater treatment sludge (raw - FI)	1												20.0								(Zevenhoven et al., 2012)
wastewater treatment sludge (raw - FI)	1												20.0								(Zevenhoven et al., 2012)
wastewater treatment sludge (raw - FI)	1												160.0								(Zevenhoven et al., 2012)
wastewater treatment sludge (raw - FI)	1												94.0								(Zevenhoven et al., 2012)
wastewater treatment sludge (raw - FI)	1												57.0								(Zevenhoven et al., 2012)
wastewater treatment sludge (raw - FI)	1												70.0								(Zevenhoven et al., 2012)
wastewater treatment sludge (raw - FI)	1												211.0								(Zevenhoven et al., 2012)
wastewater treatment sludge (raw - FI)	1												60.0								(Zevenhoven et al., 2012)
wastewater treatment sludge (raw - FI)	1												58.0								(Zevenhoven et al., 2012)
wastewater treatment sludge (raw - UK)	1	8.7	621.5	1.4	212.8	575.0	17.9					563.0		44.8	31.2	1.3				2179.0	(Donatello et al., 2010)
wastewater treatment sludge (raw - UK)	1	3.0	169.0	0.1	70.0	361.5	18.8					305.0		16.7	36.1	3.2				1355.0	(Donatello et al., 2010)

Input material	n	Cd	Cr (total)	Hg	Ni	Pb	As	B	Ba	Be	Co	Cu	Mn	Mo	Sb	Se	Sn	Sr	V	Zn	
wastewater treatment sludge (raw - UK)	1	2.3	248.5	0.1	70.9	285.0	50.7					317.0		15.7	52.1	2.1				1410.5	(Donatello et al., 2010)
wastewater treatment sludge (raw - UK)	1	1.6	264.5	0.1	81.6	229.5	9.6					354.5		12.6	20.0	2.4				1136.0	(Donatello et al., 2010)
wastewater treatment sludge (raw - UK)	1	17.4	383.5	0.5	143.0	554.5	161.0					556.0		31.4	160.0	2.5				2337.5	(Donatello et al., 2010)
wastewater treatment sludge (raw - UK)	1	1.8	59.6	2.9	61.8	238.5	10.1					398.0		17.4	14.1	5.3				1105.0	(Donatello et al., 2010)
waste water treatment sludge (raw)	1	270.0	490.0	0.5	100.0	4600.0	460.0	300.0	400.0	2.0	12.0	2000.0	810.0	36.0	####	5.0	1200.0		49.0	37.0	(Kalmykova and Karlfeldt Fedje, 2013)
mean		***26.9***	***320.7***	***0.6***	***153***	***642***	***71***					***642***		***25***		***3.1***				***1366***	*n =665*
median		***3.2***	***256.5***	***0.3***	***96***	***262***	***18***					***398***		***17***		***2.5***				***1355***	
minimum		***1.6***	***59.6***	***0.0***	***38***	***50***	***9***					***305***		***13***		***1.3***				***37***	
10th precentile		***1.7***	***68.8***	***0.1***	***63***	***83***	***10***					***312***		***14***		***1.7***				***678***	
90th percentile		***16.5***	***608.4***	***1.3***	***206***	***573***	***161***					***1138***		***40***		***5.1***				***2242***	
maximum		***270.0***	***1047.0***	***2.9***	***742***	***4600***	***460***					***2000***		***45***		***5.3***				***2338***	
coefficient of variation		***2.9***	***0.9***	***1.4***	***1.2***	***2.0***	***1.9***					***0.9***		***0***		***0.5***				***0.6***	
Waste water treatment sludge (post-processed)																					
wastewater treatment sludge (post-processed)	1	0.3	0.1 (IV)	0.3	56.0	60	3.6					601.0								1710.0	STRUBIAS contribution - ESPP (AshDec process)
wastewater treatment sludge (post-processed)	1	0.7	<1 (IV)	0.1	29.0	14	9.9	74.0			4.0	330.0	2400.0	10.0					32.0	290.0	STRUBIAS contribution - confidential data provider
wastewater treatment sludge (post-processed)	1	0.0	<1 (IV)	0.0	<15	<20	0.6					74.0								85.0	STRUBIAS contribution - confidential data provider
wastewater treatment sludge (post-processed)	4	0.3	109.5	0.7	17.0	4.2	4.7					115.0								85.0	P-REX (Mephrec process)
wastewater treatment sludge (post-processed)	4	3.9	34.2	0.2	13.9	25.3	10.0					853.0								1394.0	P-REX (LeachPhos process)
wastewater treatment sludge (post-processed)	1	0.4	1.4	0.4	0.4	0.43						1.0								1.0	P-Rex (Ecophos process)
mean		***0.9***	***48.4***	***0.3***	***23.3***	***20.8***	***5.8***					***329.0***								***594.2***	*n =12*
median		***0.4***	***34.2***	***0.3***	***17.0***	***14.0***	***4.7***					***222.5***								***187.5***	
minimum		***0.0***	***0.9***	***0.0***	***0.4***	***0.4***	***0.6***					***0.9***								***0.6***	
10th precentile		***0.2***	***8.0***	***0.0***	***5.8***	***1.9***	***1.8***					***37.5***								***43.0***	
90th percentile		***2.3***	***94.4***	***0.6***	***45.2***	***46.1***	***10.0***					***727.0***								***1552.0***	
maximum		***3.9***	***109.5***	***0.7***	***56.0***	***60.0***	***10.0***					***853.0***								***1710.0***	
coefficient of variation		***1.6***	***1.1***	***0.9***	***0.9***	***1.2***	***0.7***					***1.0***								***1.3***	
mix of wood, treated wood and sewage sludge	1		41.5	0.5	66.5	110	19.2	20.0	780		6	82	2700		1			415	34	11400	(Kröppl et al., 2011)
treated and untreated wood	1		108.0	2.5	65.0	1500	32.9	91.5	3550		14	291	6050		24			410	30	6700	(Kröppl et al., 2011)
treated wood, saw mills, swarf, trimmings	1		215.0	0.5	92.1	3030	59.5	288.0	6000		17	1100	2040		146			360	69	10600	(Kröppl et al., 2011)
slaughterhouse waste and sewage sludge mix	1														1.0		2				STRUBIAS contribution - FEhS
Currency shredded	1																			16.5	
Demolition wood	3																			1.0	
Waste wood	1												77								(Zevenhoven et al., 2012)

Input material	n	Cd	Cr (total)	Hg	Ni	Pb	As	B	Ba	Be	Co	Cu	Mn	Mo	Sb	Se	Sn	Sr	V	Zn	
Waste wood	1												83								(Zevenhoven et al., 2012)
Waste wood	1												84								(Zevenhoven et al., 2012)
Waste wood	1												76								(Zevenhoven et al., 2012)
Waste wood	1												54								(Zevenhoven et al., 2012)
Waste wood	1												107								(Zevenhoven et al., 2012)
Waste wood	1												77								(Zevenhoven et al., 2012)
Waste wood	1												355								(Zevenhoven et al., 2012)
Waste wood	1												545								(Zevenhoven et al., 2012)
Waste wood	1												159								(Zevenhoven et al., 2012)
Furniture waste	1																			0.3	
Mixed waste paper	1																			2.6	
Greenhouse-plastic waste	1																			0.5	
Refuse-derived fuel	1																			1.1	
Wood yard waste	1																			0.2	
municipal solid waste	1	160	204	8.9	42	1530	18.8				14.2	680	420	16.2	90.2	40.8	130		23.4	3840	(Demirbas, 2005)
municipal solid waste (UK)	8																				(Bogush et al., 2015)
paper industry waste	?	1.7		0.3	32.1	35.9						38.2								110.0	STRUBIAS contribution - CEPI
paper industry waste	391	5.7		0.3	40.0	72.8	17.6					200.9								1545.0	STRUBIAS contribution - CEPI

16.3 Pyrolysis & gasification materials

16.3.1 Macroelements

Table 32: Macroelements (%, dry matter) for C-rich and nutrient-rich pyrolysis materials (references: Kleber et al. (2015), Someus (2015), Zwetsloot et al. (2016))

	C	N	P	K	S	Ca	Mg	Fe
	(%, dry matter)							
C-rich pyrolysis materials								
Corn	58.8	1.06	0.2	1.9	0.0	0.9	0.7	0.7
Wheat/barley	60.8	1.41		0.1		1.3	1.0	0.2
Rice straw/husk	43.6	1.4	0.1	0.1	0.4			
Sorghum	56.4	0.74	0.2	0.4				
Soybean stover	75.4	1.59			0.0			
Peanut shell	75.3	1.83	0.2	1.1	0.1	0.3	0.1	
Pecan shell	75.9	0.26		11.6	0.0	0.6	0.1	0.0
Hazelnut shell	77.5	0.52	0.0	0.5		0.3	0.1	
Switchgrass	73.9	0.98	0.2	0.8		0.3		0.0
Bagasse	78.6	0.87	0.1	0.2		0.7	0.2	0.0
Coconut coir	73.8	0.88						
Other	64.9	1.16	0.2	1.4	0.1	0.6	0.3	0.1
Hardwoods	74.4	0.72	0.1	0.9	1.6	1.0	1.0	0.2
Softwoods	74.6	0.79	0.1	1.7	0.0	2.1	1.8	1.0
Food waste	44.4	3.28	0.7	0.9		5.2	0.5	
nutrient-rich pyrolysis materials								
Papermill waste	19.9	0.09	0.1	0.3		28.1	0.3	
Poultry manure/litter	35.3	2.15	3.3	6.0	0.9	10.3	1.2	0.3
Turkey manure/litter	31.8	2.02	3.1	4.8	0.5	4.8	1.0	0.3

Swine manure	44.9	2.79	6.1	2.3	0.8	4.8	2.9	0.6
Dairy manure	58.1	2.37	0.9	1.7	0.3	2.7	1.2	0.6
Cattle manure	48.5	1.9	0.9	4.1	0.4	2.9	1.0	0.3
Animal bone	8.0		12.4	2.0		24.3	5.7	
Animal bone	8.2	1.5	15.3	0.3	0.1	33.7	0.6	0.0
Sewage sludge	23.8	1.12	4.2					

16.3.2 Metals and metalloids

Table 33: Metal and metalloid contents (mg kg^{-1}, dry basis) and persistent organic pollutants for different types of pyrolysis & gasification materials

		Cd	Cr (total)	Hg	Ni	Pb	As	B	Ba	Co	Cu	Mn	Mo	Sb	Se	V	Zn	PAH	PCB¥	reference
	n	(mg kg^{-1} dry matter)																		
C-rich pyrolysis materials																				
soybean stover	1										34									(Ippolito et al., 2015)
pecan shell	1										8.28									(Ippolito et al., 2015)
coconut coir	1										66.2									(Ippolito et al., 2015)
rice (300°C)	1																	2		(Freddo et al., 2012)
rice (600°C)	1																	1		(Freddo et al., 2012)
bamboo (300°C)	1	0.03	4.3		1.4	1.9	0.3				10						124	2		(Freddo et al., 2012)
bamboo (600°C)	1	0.03	3.4		1.2	3.9	0.3				6.3						207	1		(Freddo et al., 2012)
redwood (300°C)	1	0.94	4.5		0.4	0.6	0.1				2						38	4		(Freddo et al., 2012)
redwood (600°C)	1	0.02	3.4		0.6	0.9	0.2				2						38	0		(Freddo et al., 2012)
maize (300°C)	1	0.03	5.1		0.4	0.1	0.2				10.6						92	4		(Freddo et al., 2012)
maize (600°C)	1	0.03	6.5		0.6	1.1	0.2				13.2						54	5		(Freddo et al., 2012)
softwood (500°C)	1	0.02	0.1		0.1	0.1	0				0.04						0.9	9		(Freddo et al., 2012)
pine	1	0.1	2.8			1					14						16			(Knowles et al., 2011)
sawdust	1				7						48	185					31			(Mankasingh et al., 2011)
palm leaves	1				7						87	193					46			(Mankasingh et al., 2011)
rice paddy husk	1				10						27	704					77			(Mankasingh et al., 2011)
rice paddy husk	1				2						8	321					36			(Mankasingh et al., 2011)
Prosopis	1				26						20	940					48			(Mankasingh et al., 2011)
cassia stems	1				12						29	191					46			(Mankasingh et al., 2011)
citrous wood	1							60			39	145					505			(Graber et al., 2010b)
peanut hulls (400°C)	1	1	4		2			32			16	116	5				35			(Gaskin et al., 2008)

	n	Cd	Cr (total)	Hg	Ni	Pb	As	B	Ba	Co	Cu	Mn	Mo	Sb	Se	V	Zn	PAH	PCB¥	reference
peanut hulls (500°C)	1		4		2			34			19	131					37			(Gaskin et al., 2008)
pine chips (400°C)	1				2			6			25	274					15			(Gaskin et al., 2008)
pine chips (500°C)	1		3		3			4			9	258					18			(Gaskin et al., 2008)
wood	5	1	10	0.1	11	9	3	19	20	3	12	215	3.5	5		3	102	10.5		(ECN, 2017)
herbaceous plants	9	0.1	12.1	0	17	45	4	55	97	6	19	380	2.3	6	2	8	48	15.5		(ECN, 2017)
plant (unknown)					3	1	<1			<1	3				<0.3		19	1	-	(Someus, 2015)
plant (unknown)	1	<0.3	9	0.04	13	8	<1			1	9				<0.3		150	5	-	(Someus, 2015)
plant (unknown)	1	0.4	15	<1	14	14	1			3	49				0.5		294	0	-	(Someus, 2015)
other	1										4.76									(Ippolito et al., 2015)
shrub cutting (untreated)	1	0.56	81.5	0.01	75.4	23	1.9													STRUBIAS - EUROFEMA
slug pellet (98% wheat flour, 2% ferric phosphate)	1	0.49	69.5	0.01	60.4	19	1.6													STRUBIAS - EUROFEMA
tobacco flour	1	0.42	46.1	0.01	31.1	4	0.55													STRUBIAS - EUROFEMA
carbonaceous product	1	0	4	0	5	16	3				25						45			STRUBIAS - EUROFEMA
mineral-rich pyrolysis materials																				
fermentation residues	1	0.58	181	0.01	237	29	2.33													
poultry manure/litter	1										472								0.4	(Ippolito et al., 2015)
poultry litter (400°C)	1	3	28		14			91			805	596	17				628			(Gaskin et al., 2008)
poultry litter (500°C)	1		59		20			100			1034	725	14				752			(Gaskin et al., 2008)
poultry litter (350°C)	1	0.25			8	1					213									(Uchimiya et al., 2012)
poultry litter (700°C)	1	0.11			11	1					310									(Uchimiya et al., 2012)
turkey manure/litter	1										107								0.4	(Ippolito et al., 2015)
turkey litter (350°C)	1	0.7			29	2					535									(Uchimiya et al., 2012)
turkey litter (700°C)	1	0.7			40	-					762									(Uchimiya et al., 2012)

	n	Cd	Cr (total)	Hg	Ni	Pb	As	B	Ba	Co	Cu	Mn	Mo	Sb	Se	V	Zn	PAH	PCB¥	reference
swine manure	1	<1	11	<0.01	18	<10	1.2				377						1098	<0.1		STRUBIAS - confidential
swine manure	1										114								0.4	(Ippolito et al., 2015)
swine solids (350°C)	1	0.57			16	3					1538									(Uchimiya et al., 2012)
swine solids (700°C)	1	0.23			26	-					2446									(Uchimiya et al., 2012)
dairy manure	1										222								0.4	(Ippolito et al., 2015)
dairy manure (350°C)	1	0.2			16	1					99									(Uchimiya et al., 2012)
dairy manure (700°C)	1	-			25	0					163									(Uchimiya et al., 2012)
paved feedlock manure (350°C)	1	0.2			4	1					92									(Uchimiya et al., 2012)
paved feedlock manure (700°C)	1	0.02			7	0					136									(Uchimiya et al., 2012)
animal bone material	1	<0.3	<1	<0.03	<1	<1	<1			<1	8				<0.3		203			(Someus, 2015)
papermill waste	1										513									(Ippolito et al., 2015)
undetermined	1	<0.5	11	<0.05	7.9	<5	<4	98.2		5.52	158	1070	10.9		<2		1500			STRUBIAS - confidential

¥: WHO eq, ng kg-1 dry matter

17 Methods

17.1 Soil screening values and acceptable soil screening limit concentrations

Soil Screening Values are concentration limits (mg kg^{-1} soil dry weight) of contaminants in the soil above which certain actions are recommended, enforced or adopted in many Member States in Europe in order to protect the environment and human health (Carlon, 2007). Hence, **this assessment cannot be interpreted as a risk assessment as the relationship between the soil screening limit values and the actual risk for human health or the environment has been established by the individual Member States.** Rather, the calculations should aid Member States and stakeholders to make a valid approximation of the limit concentrations for fertilising materials in order to ensure that the long-term use of such materials does not lead to exceeding the soil screening values that have been established at Member State level based on scientific and other concerns.

Clear **advantages** of the use of soil screening values are the speed and ease of the assessment, and the comparability, transparency and the straightforward understanding by a wide variety of non-specialist stakeholders (Carlon, 2007). One of the major **limitations** is that crucial site-specific considerations cannot be included for which reason rather conservative limit values are typically set.

The **type of soil screening values** can be related to **different levels of risk**, e.g. negligible risk or potentially unacceptable risk levels (Carlon, 2007). On the one hand, the derivation of *negligible risk* levels aims at excluding any type of adverse effect on even the most sensitive land. It is characterized by a very high conservatism, the comprehensive protection of the natural environment and the definition of long term sustainability objectives. On the other hand, the derivation of *potentially unacceptable risk levels* aims at preventing significant adverse effects. It is characterised by a low conservatism and a functional perspective of soil protection directed to the support of human living and main ecological functions. In some cases, the need for further investigations is related to some intermediate risk levels. A useful intermediate risk is then associated with a scenario based on generic (protective) assumptions, the validity of which could be checked in a site-specific risk assessment. Therefore, in some cases three sets of soil screening values can be derived on the basis of negligible, *intermediate (warning)* and potentially unacceptable risk levels, and these soil screening values may be applied as long term quality objectives, trigger values and cut-off (remediation needed) values, respectively, as it is exemplified in Figure 33.

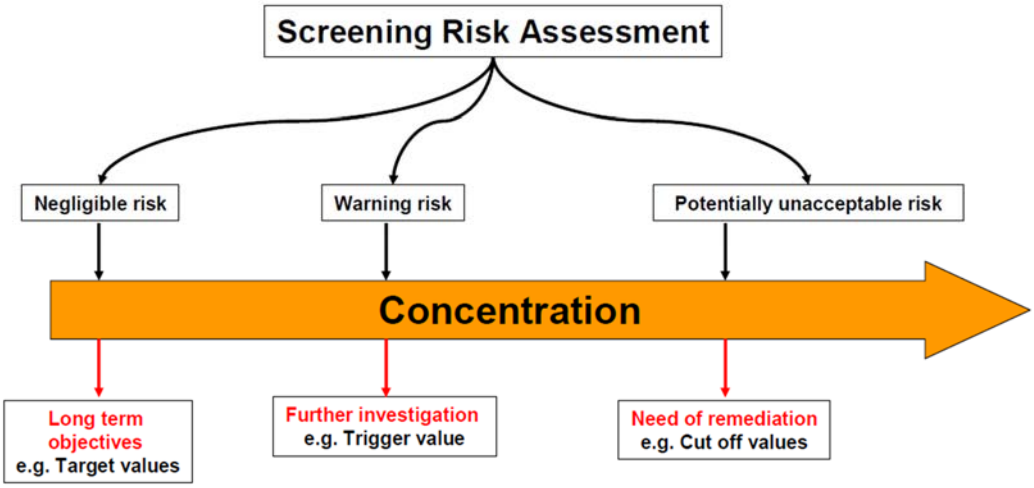


Figure 33: Derivation of soil screening values based on various risk levels and application of the different screening values (adopted from Carlon, 2007)

In summary, screening values can be classified into different risk categories, broadly termed negligible risk, *warning risk* and potentially unacceptable risk. The appreciation triggered by each risk category as well as the exact interpretation thereof, however, depends on the national regulation. Moreover, they can be distinguished into screening risk and site-specific risk concentration values (Carlon, 2007).

The applied **category of risk** in the derivation of a specific screening value is usually related to the intended application within the legal framework. In this regard, there are no fixed rules, but common practices. Long term objectives for soil quality, for example, are usually based on the negligible risk level; in this case soil screening values might relate to multifunctional uses of the site or could be a representation of sustainable soil quality. By definition and for practical reasons, natural average background values are often regarded to be associated to negligible risk level (soil quality objectives lower than the average background level would not be feasible). On the other hand, the possible need for actions is often related to levels indicating a potential unacceptable risk. In an extended definition, actions can include remediation, restrictions in land use, urgency for remediation, further investigations and/or the application of site-specific risk assessment.

Soil screening values adopted in **European countries vary widely** in multiple aspects (Carlon, 2007). The use of soil screening values varies from setting long term quality objectives, via triggering further investigations, to enforcing remedial actions. Derivation methods of soil screening values have scientific and political bases. In relation to the common market and common environmental policies in Europe, this variability has raised concern among both regulators and risk assessors (Carlon, 2007).

The ***predicted accumulation*** estimates how much of a trace metal accumulates in soil following annual applications (over years of farming) and takes into account an estimated loss of trace metals in soil from transport of the trace metal into surrounding media – a mass-balance approach (The Weinberg Group Inc., 2000). The accumulation and behaviour of trace metals in soil from agricultural application depends essentially on (1) **farming duration (years), (2) the application rate of the fertilising products, (3) the concentration of the trace metal in the fertiliser and (4) the fate and transport of the trace metal in soil**.

Because soil accumulation depends on so many different factors, which all vary given any situation, not all situations can be represented when deriving the predicted accumulation**.** The soil accumulation calculation is based on the most important parameters and loss pathways, and is estimated based on representative high-end (general, not site-specific) assumptions resulting in **more protective limit metal and metalloid concentrations**.

Therefore, the assessment deployed in this work is based on following principles and assumptions:

- The *warning risk* will be considered as the level of risk for the derivation of the soil screening values. Some Member States (e.g. Italy, Lithuania, the Netherlands, Poland, UK) apply only two levels of risk (negligible risk and potentially unacceptable risk), in which case their screening value for potentially unacceptable risk will be considered. Soil screening values differ largely between Member States across Europe (Table 34). **The value of the *25th percentile of the distribution of soil screening values across EU Member States* has been selected as the maximum accumulation** (i.e. 75% of the soil screening values across EU Member States are higher than the selected value). For Ba, however, the 25th percentile values is close to its average background concentration in European soils for which reason the 50th percentile value was selected as predicted no-effect concentration (Table 34).

Table 34: Soil screening values for different EU Member States (regular format: warning risk; italic format: potentially unacceptable risk for metals and metalloids (mg kg^{-1} dry weight) (Source: Carlon, 2007). The values given in bold indicate the predicted no-effect concentrations (PNEC) applied for the assessment of this study based on the interpolated 25th percentile of the distribution of soil screening values across EU Member States, with the exception of Ba where the median value (50th percentile) was used.

	AT	BE -Fl	BE - Wa	CZ	FI	DE	SK	DK	SE	*IT*	*LT*	*NL*	*PL*	*UK*	median value	25th percentile
As	20	58	40	65	5	50	30	10	15	*20*	*10*	*55*	*22.5*	*20*	21.25	**16**
Ba				1000							*600*	*625*	*285*		**612.5**	521
Be				15			20			*2*	*10*	*30*			15	**10**
Co				180	20		50		200	*20*	*30*	*240*	*45*		47.5	**28**
Cr	50		125	450	100	400	250	50	250	*150*	*100*	*380*	*170*	*130*	150	**100**
Mo				50			40	2			*5*	*200*	*25*		32.5	**10**
Sb	2			25	2					*3*	*5*	*100*		*35*	5	**3**
Se							5			*3*	*5*	*100*		*35*	5	**5**
V				340	100		200		200	*90*	*150*	*250*			200	**125**

- With the exception of Tl, the screening of the list of metals and metalloids revealed that the list of elements is complete and encompasses all different potentially toxic metals/metalloids that can be found in thermal oxidation materials & derivates and pyrolysis & gasification materials derived from the eligible input materials. **Thallium is considered as toxic for human and animal organisms, microorganisms and plants (Nriagu, 1998; Peter and Viraraghavan, 2005).** The toxicity of this element is higher compared to Hg, Cd and Pb (Repetto et al., 1998; Peter and Viraraghavan, 2005). **Major sources, which could lead to increased concentrations of Tl in the environment, include materials derived from fossil fuels and mineral ores, both present on the eligible input material list for this CMC (Antonia López Antón et al., 2013; Karbowska, 2016).** Environmentally safe limits for Tl in soils vary from 1 to 2 mg kg^{-1} (Canadian Council of Ministers of the Environment (CCME), 2003; Xiao et al., 2004; van Vlaardingen et al., 2005). In this assessment, **the upper value of 2 mg kg^{-1} will be used for deriving soil screening acceptable limit concentrations of selected metals and metalloids.**

- The **application scenario** of the STRUBIAS materials is a challenging aspect to consider given their wide-ranging nutrient concentration. Compared to traditional fertilisers, much bigger single doses of wood ash should be applied to get plant growth responses or liming effects from ash additions (up to 3 tonnes of wood ash per hectare per year (Haglund and Expertsgroup, 2008)). The application rates of 3 tonnes ha^{-1} yr^{-1} is retained in this assessment. On croplands, the fertilising products are often ploughed into the soil, but this is not the case for applications at forests and permanent grasslands. A farming duration of *100 years* and a soil-fertiliser mixing layer of 20 cm is assumed for this evaluation. The assumed application rates vary as a function of STRUBIAS material, with assumed rates of *3, 5 and 20 tonnes ha^{-1} yr^{-1}* for thermal oxidation materials & derivates, nutrient-rich pyrolysis & gasification materials and carbon-rich pyrolysis & gasification materials, respectively. This high-end scenario enables to consider more readily available, average values for background trace metal concentrations in soils, atmospheric trace metal deposition and solid-liquid partition coefficients. Moreover, the soil bulk density is assumed to be 1.4 g cm^{-3} (Table 35).

Table 35: Assumptions made for the application scenario for STRUBIAS materials

Parameter	Description	value applied	unit
AR	application rate	3/5/20	tonne ha^{-1} yr^{-1}
T	deposition period	100	yr
Z	soil mixing depth	20	cm
BD	soil bulk density	1.4	g cm^{-3}

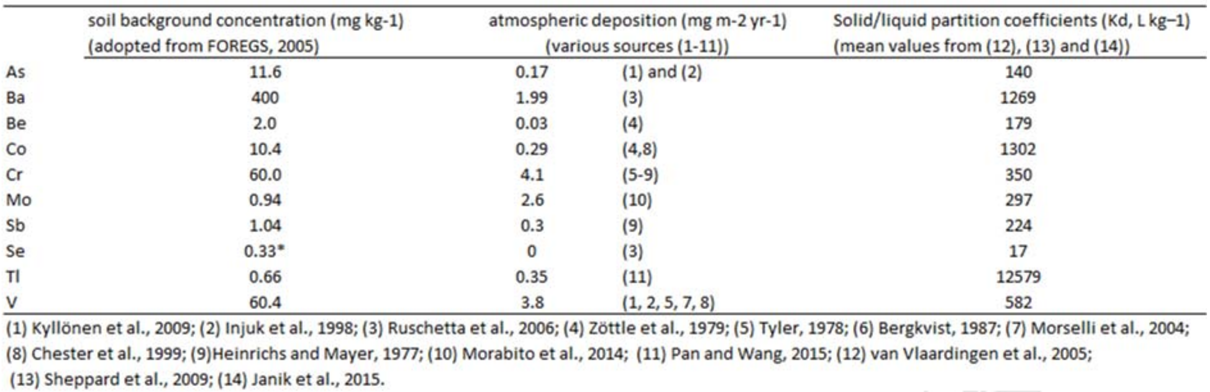
- European *averages* of metals and metalloids for soil background concentrations are used as these are well-documented for most metals and metalloids (FOREGS, 2005) (Table 36). As no value was available for Se in the FOREGS database, the values documented by De Temmerman et al. (2014) were used (Table 36).

- *Average* data on atmospheric deposition at agricultural and forested European ecosystems is preferentially used (Heinrichs and Mayer, 1977; Tyler, 1978; Zöttle et al., 1979; Bergkvist, 1987; Injuk et al., 1998; Chester et al., 1999; Ruschetta et al., 2003; Morselli et al., 2004; Kyllonen et al., 2009; Morabito et al., 2014; Pan and Wang, 2015) (Table 36). Data available for the different metals and metalloids is, however, rather limited. Moreover, the data do not show good geographic coverage for Europe. If no values for particular elements are available for terrestrial ecosystems, best estimates from atmospheric deposition at sea or other geographic regions are used. A sensitivity analysis was performed to assess the importance of any variations on atmospheric deposition rates of metals and metalloids, but indicated that the outcomes are only insignificantly affected by variations in atmospheric deposition.

- Leaching and plant uptake are considered as outputs of metals and metalloids from the soil. Fate and transport of trace metals depends on the soil condition, climatic conditions and trace metal behaviour and adsorption kinetics. The deployed approach is based on the assumption of *elemental solid-liquid partitioning* for the elements, and it is considered that any metals available in the liquid fraction are removed from the soil through leaching and plant uptake. Liquid-solid partition coefficients applied in this assessment are average values as collected from different studies and soil types (van Vlaardingen et al., 2005; Sheppard et al., 2009; Janik et al., 2015b). Based on the results from the GEMAS project (Janik et al., 2015b), it is recognised that the use of Kd coefficients to model sorption of metals and metalloids is associated to a high degree of uncertainty since Kd values are extremely variable as a result of the impact of, for instance, pH, organic matter and to some extent clay and oxides on the retention of metals in soils. Therefore, sensitivity analyses with different Kd values were executed to evaluate the robustness of the analysis performed.

- A default percolation (precipitation minus evapotranspiration) estimate of 200 mm year^{-1} and a gravimetric soil water content of 0.3 (v/v) are assumed (conservative estimates from an EU perspective). Sensitivity analyses indicated that the liquid-solid partition coefficient was the most important parameter determining trace metal losses, and that the outcome was relatively insensitive to variations in precipitation and soil moisture content.

Table 36: Applied values of soil background concentration, solid/liquid partition coefficients and atmospheric depositions as applied for deriving soil screening acceptable limit concentrations of selected metals and metalloids in CE fertilising products derived from STRUBIAS materials.



	soil background concentration (mg kg-1) (adopted from FOREGS, 2005)	atmospheric deposition (mg m-2 yr-1) (various sources (1-11))		Solid/liquid partition coefficients (Kd, L kg–1) (mean values from (12), (13) and (14))
As	11.6	0.17	(1) and (2)	140
Ba	400	1.99	(3)	1269
Be	2.0	0.03	(4)	179
Co	10.4	0.29	(4,8)	1302
Cr	60.0	4.1	(5-9)	350
Mo	0.94	2.6	(10)	297
Sb	1.04	0.3	(9)	224
Se	0.33*	0	(3)	17
Tl	0.66	0.35	(11)	12579
V	60.4	3.8	(1, 2, 5, 7, 8)	582

(1) Kyllönen et al., 2009; (2) Injuk et al., 1998; (3) Ruschetta et al., 2006; (4) Zöttle et al., 1979; (5) Tyler, 1978; (6) Bergkvist, 1987; (7) Morselli et al., 2004; (8) Chester et al., 1999; (9)Heinrichs and Mayer, 1977; (10) Morabito et al., 2014; (11) Pan and Wang, 2015; (12) van Vlaardingen et al., 2005; (13) Sheppard et al., 2009; (14) Janik et al., 2015.

- Trace metals are added to soil over years of farming. Because of losses from the root zone, the rate of accumulation of the trace metals in the soil will slow down over the years. Following application year after year, on the same soil, the concentrations of the trace metals are expected to reach a steady state. The rate at which a metal/metalloid is lost from the soil through leaching and plant uptake is defined as the soil loss constant. Following equation is then used to calculate the soil loss constant (The Weinberg Group Inc., 2000):

$$Ks = \frac{P}{\theta * Z * (1 + BD * \frac{Kd}{\theta})} \qquad \text{(Equation 1)}$$

where:
Ks = soil loss constant (yr^{-1})
P = average annual precipitation ($cm\ yr^{-1}$)
Z = soil mixing depth (cm)
BD = soil bulk density ($g\ cm^{-3}$)
K_d = soil-water partitioning coefficient ($mL\ g^{-1}$)
Θ = soil volumetric water content ($mL\ cm^{-3}$)

The predicted accumulation is then modelled using following equation (The Weinberg Group Inc., 2000):

$$PA = \frac{(AR + AD) * [1 - \exp(-Ks * T)] * 1e4}{Z * BD * Ks} \qquad \text{(Equation 2)}$$

where:
PA: predicted accumulation ($mg\ kg^{-1}$)

AR: application rate (tonne ha^{-1} yr^{-1})
AD: atmospheric deposition (tonne ha^{-1} yr^{-1})

In a final step, the metal/metalloid concentration in the STRUBIAS material is then optimised so that the predicted accumulation is lower than the soil screening acceptable limit concentration.

17.2 Meta-analyses

17.2.1 Data sources

Data sources that provided pertinent evidence from trustworthy sources in a manner that is comprehensive, scientifically robust, objective and transparent were collected. In order to safeguard transparency, confidential and non-publically available works were not considered in this meta-analysis.

Studies that quantitatively reported DMY and/or PUE for recovered P and mineral P-fertiliser treatments with a minimum of three experimental replicates were selected. Only assessments that were performed on soils and plant species from boreal, temperate and Mediterranean climate regions - within or outside Europe - were retained in order to provide an assessment that is relevant for the EU-27 (i.e. geographic coordinated > 35°N/S). F_{prim} treatments included different P fertilising substances, such as triple superphosphate, monoammonium phosphate, diammonium phosphate, calcium super phosphate, single superphosphate, and potassium phosphate. Dry matter yield and plant P uptake was mostly measured for aboveground plant biomass yield, but some studies assessed whole plant biomass or specific plant organs. If not directly reported, PUE was derived from the DMY and plant P concentration, and concomitant standard deviations were calculated assuming error propagation rules for normal distributions. When data were only provided in graphical format, the corresponding authors of the studies were contacted to obtain the raw numerical data. If not successful, relevant data points were extracted graphically from available figures. When studies did not report measures of variance, the corresponding author was contacted with a request to provide the raw data for the calculation of the standard deviation. For studies in which it was not possible to acquire measures of variance, the uncertainty of the missing effect sizes was drawn from a multiple imputation algorithm based on the assumption of a common underlying variance, after which Rubin's rules were applied to get the point estimates and standard errors of the meta-analysis results (Schwarzer et al., 2015).

Following studies were included in the assessment:
Precipitated phosphate salts & derivates: (Johnston and Richards, 2003; Hammond and White, 2005; Gonzalez Ponce and Garcia Lopez De Sa, 2007; Plaza et al., 2007; Massey et al., 2009; Weinfurtner et al., 2009; Ruiz Diaz et al., 2010; Cabeza et al., 2011; Gell et al., 2011; Liu et al., 2011; Antonini et al., 2012; Ackerman et al., 2013; Thompson, 2013; Achat et al., 2014b; Uysal et al., 2014; Bonvin et al., 2015; Cerrillo et al., 2015; Vogel et al., 2015;

Wragge, 2015; Hilt et al., 2016; Katanda et al., 2016; Liu et al., 2016; Sigurnjak et al., 2016; STOWA, 2016b; Talboys et al., 2016; Vaneeckhaute et al., 2016; Degryse et al., 2017).

Thermal oxidation materials & derivates: (Codling et al., 2002; Franz, 2008; Bird and Drizo, 2009; Kuligowski et al., 2010; Schiemenz and Eichler-Löbermann, 2010; Cabeza et al., 2011; Schiemenz et al., 2011; Komiyama et al., 2013; Rex et al., 2013; Weigand et al., 2013; Wells, 2013; Nanzer et al., 2014; Severin et al., 2014; Vogel et al., 2015; Wragge, 2015; Brod et al., 2016; Delin, 2016; Reiter and Middleton, 2016).

Pyrolysis & gasification materials: (Codling et al., 2002; Kuligowski et al., 2010; Müller-Stöver et al., 2012; Alotaibi et al., 2013; Collins et al., 2013; Ma and Matsunaka, 2013; Siebers et al., 2014; Reiter and Middleton, 2016).

17.2.2 Effect size

Standardisation of the raw results was undertaken through calculation of the effect size. This allows quantitative statistical information to be pooled from, and robust statistical comparisons to be made between effects from a range of studies that reported results based on different experimental variables. The effect size was calculated as the natural logarithm of the response ratio R by using the following equation (Borenstein et al., 2009):

$$\ln R = \ln RAE$$

The response ratio was then calculated for a number of "cases" where all grouping variables (such as soil and crop used, crop harvest time, P application rate, etc.; see below) are identical for both fertiliser treatments. The log response ratio and its variance was used in the analysis to yield summary effects and confidence limits in log units during the different meta-analysis steps. Each of these values was then converted back to response ratios to report the final results (Borenstein et al., 2009). All analysis were performed in the R software environment.

17.2.3 Grouping variables

Data were grouped prior to meta-analysis to enable a broad ranging assessment of fertilising effectiveness of F_{sec} as a function of grouping variables that relate to soil type, plant group and management option. For all selected studies, quantitative information on following grouping variables were recorded: soil pH, soil texture, feedstock, sowed plant species, application form, harvest time after fertiliser application, soil P fertility, and experimental design. When specific parameters were not documented in the publication, the corresponding author was requested to provide the information; in case quantitative data was not available an expert opinion on parameter categorisation into groups was requested from the lead author.

Soil pH was classified as acidic for soils with a pH value less or equal than 6.0, and as neutral/basic for soils of pH greater than 6.0. *Soil texture* was classified as coarse (sand, loamy sand and sandy loam), medium (loam, silt loam, and silt) or fine (sandy clay, sandy clay loam, clay loam, silty sandy clay loam, silty clay and clay). *Feedstock* indicated the input materials from which the STRUBIAS material was derived (e.g. sewage sludge, manure). For thermal oxidation materials & derivates, *post-processing* refers to the completing of a wet-digestion or thermal post-processing step to improve the plant P-availability of specific feedstocks (e.g. sewage sludge). *Plant groups* involved grasses (both annual and perennial species), oilseeds, cereals, legumes and others (leaf vegetable, cormous flowering plants, fruit vegetable, and pulse crops). *Application form* distinguished fertilisers that were applied as a powder or as granules. *Assessment time* was categorised as short and long for studies that harvested plants within and posterior to a period of 65 days of fertiliser application. In case of assessments on grasses, only the cumulative biomass and P uptake at the end of the experiment was considered. *Soil P status* was categorised as P-poor and P-rich, with a cut-off value of extractable Olsen-P content of 12.4 mg P kg^{-1}. The cut-off value was based on the average limit value for the "very low" P fertility category for a single soil within a number of European countries (Jordan-Meille et al., 2012). When other extractable P methods were applied, conversion methods and comparative relationships as given in Jordan-Meille et al. (2012), Neyroud and Lischer (2003) and McLaughlin (2002) were applied. When no extractable P data values were reported, expert opinions were requested from the corresponding authors. The approach applied based on a single cut-off value to discern soil P fertility for all soil-plant combinations is a simplification of a complex scientific matter (Jordan-Meille et al., 2012), but we are confident that it meets the objective of generally discerning settings in this meta-analysis study. *Experimental setting* separated pot from field studies. *Experimental design* assessed if the experimental study design involved the addition of plant nutrients, other than P, present in F_{sec} were also added in F_{prim}; "Fully balanced" corresponds to cases where all nutrients present in F_{sec} were also added in the F_{prim} treatments. "Deficient" refers to design where primary and secondary macronutrient present in F_{sec} were not added in F_{prim} (e.g. struvite as F_{sec}, but no addition of Mg in F_{prim}; poultry litter pyrolysis & gasification materials as F_{sec}, but no addition of N or K in P_{prim}).

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