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Technical proposals for selected new fertilising materials under the Fertilising Products Regulation (Regulation (EU) 2019/1009)

Process and quality criteria, and assessment of environmental and market impacts for precipitated phosphate salts & derivates, thermal oxidation materials & derivates and pyrolysis & gasification materials

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Abstract

The European Commission has recently revised the EU legislation on fertilisers, expanding its scope to secondary-raw-material-based fertilising products, and resulting in the publication of the new EU Fertilising Products Regulation ((EU) 2019/1009). This report explores a possible legal framework for the manufacturing and placing on the market of specific safe and effective fertilising products derived from biogenic wastes and other secondary raw materials. Specifically, three categories of fertilising materials have been evaluated:

- precipitated phosphate salts & derivates;
- thermal oxidation materials & derivates;
- pyrolysis & gasification materials.

The report contains technical proposals on eligible input materials, process conditions, quality requirements as well as quality management system requirements. The proposals might form the basis for the legal requirements that those candidate materials shall comply with if they become regulated under the new legislative framework. Additionally, the report assesses the possible impacts in order to shed a light on the added value that these fertilising materials could provide for food security, food safety, environmental protection, and the European fertilising and agricultural sector.

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SYNOPSIS

1 Executive summary

The European Commission has revised the **EU Fertiliser Regulation** ((EC) 2003/2003), expanding its scope to secondary-raw-material-based fertilising products, and resulting in the publication of the new EU Fertilising Products Regulation ((EU) 2019/1009)[1]. This report explores the technical and market conditions for a possible legal framework for the manufacturing and placing on the market of specific safe and effective fertilising products derived from biogenic wastes and other secondary raw materials. The acronym **STRUBIAS** (STRUvite, BIochar, or incineration AShes) was initially chosen as the working title for this project to reflect the targeted fertilising materials. Throughout the course of the STRUBIAS project, the scope of these materials has however been further refined and significantly expanded. The work delivered within this project should contribute to making the recovery of nutrients, organic matter and liming materials from secondary raw materials a more attractive practice across Europe. ‘Closing the loop – An EU action plan for the **circular economy**’, as adopted by the European Commission[2], identified the Fertilisers Regulation revision as a key legislative proposal to boost the market for secondary raw materials, and the revised Waste Framework Directive[3] establishes ambitious targets for recycling. Several STRUBIAS materials show a substantial potential to provide safe sources of **phosphorus** (P) that can constitute an alternative for the primary raw material phosphate rock. Both phosphorus and phosphate rock have been identified by the European Commission as critical raw materials, based on its supply risk and the economic importance for EU operators in particular.

The newly adopted **EU Fertilising Products Regulation ((EU) 2019/1009), which formed the basis for the present study,** includes provisions for EU fertilising products that contain requirements at two levels in accordance with their *intended function* (‘Product Function Category’, PFC), and for the *component materials* contained in the EU fertilising product (‘Component Material Categories’, CMC). Specific requirements for each of the CMCs apply because different component materials warrant different process requirements and control mechanisms adapted to their different potential hazardousness and variability, in turn dependent on the quality of the input materials applied, production process conditions, etc. Component materials for EU fertilising products should, therefore, be divided into different categories. Several STRUBIAS materials have the potential to become **component materials** in the EU Fertilising Products Regulation ((EU) 2019/1009). STRUBIAS materials might be used to produce fertilising materials with different intended functions, including inorganic and organic fertilisers, liming materials, growing media, soil improvers, plant biostimulants, and fertilising product blends.

The **objectives** of this report are twofold. Firstly, it collects and assesses information to make **technical proposals** on eligible input materials and process conditions for STRUBIAS production pathways, quality requirements for STRUBIAS materials, and quality

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[1] https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELLAR:e351eb07-9713-11e9-9369-01aa75ed71a1

[2] More information on: http://ec.europa.eu/environment/circular-economy/index_en.htm

[3] Directive (EU) 2018/851 of the European Parliament and of the Council of 30 May 2018 amending Directive 2008/98/EC on waste.

management systems, which might form the basis for the legal requirements that those candidate materials shall comply with if they become regulated under the EU Fertilising Products Regulation ((EU) 2019/1009). Secondly, the report studies the possible impacts in order to shed a light on the **possible added value** that the STRUBIAS materials could provide for food security, food safety, environmental protection, and the European fertilising and agricultural sector.

The information laid down in this document has been collated and assessed by the **European Commission's Joint Research Centre** in **Seville** (JRC - Seville) who led the work on the STRUBIAS project, guided by the principles of technical expertise, transparency and neutrality. The JRC has been supported in the process by a **technical working group** ('STRUBIAS subgroup') that constitutes a subgroup of the Commission expert group on Fertilisers, which includes representatives from EU Member States, EU trade/business/professional associations, as well as from other institutions such as think tanks, research and academic institutions. The role of the subgroup is to participate in the process of sharing knowledge and providing non-binding expert advice to the European Commission on manufacturing and use aspects for STRUBIAS materials. The STRUBIAS subgroup has been requested to provide techno-scientific data that contributed to the information collected in this report, and has been consulted through meetings and written consultation rounds on previous versions of this report. The work of the STRUBIAS subgroup and all other contributors is gratefully acknowledged.

The **technical proposals** for the STRUBIAS materials ('STRUBIAS recovery rules') are based on the **principles** that - to the best possible extent - they: (i) shall provide an avenue for the agricultural exploitation of recovered materials in high-quality fertilising products that do not pose overall adverse environmental impacts or human health risks, (ii) create a level playing field for EU fertilising products derived from primary and secondary raw materials that offers simplicity and clarity to producers and consumers that are active in the European fertilising market, (iii) shall be set in a sufficiently flexible manner to encourage industry to undertake nutrient recycling actions, (iv) apply a neutral stance towards all existing and future technological systems operating in the market and input materials available, and (v) shall be clear, concise and enforceable to support a stable legal framework.

Throughout the project, the **candidate material scope and names** have been refined. The scope of the STRUBIAS material originally referred to as 'struvite' has been widened to include the broad spectrum of phosphate salts that could be produced through a precipitation process, and can be used according to their intended function as P fertilisers. The scope of 'ash-based materials' and 'biochar' has been refined to cover the entire spectrum of thermochemical material transformation processes, and separates materials based on their degree of pollutant destruction. The oxygen supply during the thermochemical process distinguishes 'thermal oxidation materials' from those that have undergone 'gasification' (partial oxidation) and 'pyrolysis' (thermal degradation of organic material in the absence of oxygen). Precipitated phosphate salts and thermal oxidation materials can also be used as precursors or intermediates to produce fertilising materials of a different chemical composition ('derivates'; e.g. superphosphates, di-ammonium phosphates, potassium chlorides). Accordingly, the names of the STRUBIAS materials have been changed to 'precipitated phosphate salts & derivates', 'thermal oxidation materials & derivates', and 'pyrolysis & gasification materials'. The STRUBIAS acronym has, however, been maintained for simplicity and recognisability.

The work of developing the technical proposals for STRUBIAS materials that could be used as components in CE fertilising products has been guided by the following **fundamental criteria**:

I. *The use of the STRUBIAS materials will not lead to overall adverse environmental or human health impacts.*

II. *The STRUBIAS material shall provide plants/mushrooms*[4] *with nutrients or improve their nutrition efficiency, either on its own or mixed with another material.*

III. *Trade on the internal market can be expected for STRUBIAS materials, based on the current market and the future market and trade forecasts.*

Regarding the first criterion, the assessment of the available evidence shows that the potential negative health impacts associated with the manufacturing and use of STRUBIAS materials identified relate to risks of both **recycling contaminants** contained in the eligible STRUBIAS input materials and the ***de novo* production of specific contaminants**. The technical proposals pay special attention to these risks and the approach followed is to propose a set of different types of requirements to be included in the recovery rules. The eligible input materials have been delimited for all three STRUBIAS material groups, excluding for instance mixed municipal waste as feedstock for STRUBIAS production processes. Moreover, specific requirements are proposed on, for instance, conditions for thermal oxidation to effectively destroy organic contaminants. In some cases, the demonstrated relationship between specific contaminants of concern with easily measurable product quality properties is used to indirectly limit contaminants (e.g. volatile organic compounds for pyrolysis and gasification materials that are closely related to the H:C ratio of the materials). These requirements impose effective control mechanisms for the wide-ranging and heterogeneous group of organic contaminants. Many of these compounds are identified as 'contaminants of emerging concern' because the risk they pose to human health and the environment is not yet fully understood (e.g. pharmaceutical compounds and personal care products, antibiotic resistance genes, specific biocides). For those emerging contaminants, often no 'safe' environmental limit concentrations can be established, and their monitoring would imply extensive material testing and high compliance costs for STRUBIAS manufacturers. Still, particular contaminants of concern were identified that could be present in the STRUBIAS materials. This refers specifically to biological pathogens, specific persistent organic pollutants (e.g. polycyclic aromatic hydrocarbons, dioxins and dioxin-like

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[4] For reasons of simplicity, where reference is made to plants in the remainder of this document, it shall be understood that this relates both to plants and mushrooms.

compounds), macroscopic impurities, macroelements (e.g. chlorine) and metals. Therefore, direct product quality requirements are proposed to restrict those compounds in the STRUBIAS materials as part of cost-effective compliance testing schemes. Implementing and applying a **quality management system** based on the quality of the design and production phase is required to ensure full compliance with the complex and extensive legislative requirements that apply to the handling, transport, and operating conditions of the STRUBIAS production processes. A conformity assessment procedure has been proposed that requests manufacturers of STRUBIAS materials to operate an approved quality management system - assessed by a notified body - in order to ensure conformity with legislative requirements, including the inspection of the STRUBIAS material quality (‘module D1’). Altogether, the technical requirements for the different Component Material Categories should ensure that the use of the STRUBIAS materials does not lead to overall adverse environmental or human health impacts, thus safeguarding food and animal safety.

Regarding the second criterion, the availability of nutrients contained in STRUBIAS materials and effects on plant nutrition efficiency were assessed by carrying out and interpreting meta-analyses results based on studies relevant for European settings. Meta-analysis techniques enable it to be established whether the findings on **agronomic efficiency** are consistent and generalisable across European settings and facilitate an understanding of the reasons why some studies differ in their results. For these reasons, a meta-analysis of similar, well-conducted, randomised, controlled trials has been considered one of the highest levels of evidence. The JRC assessment results indicated that many STRUBIAS materials that meet the proposed STRUBIAS recovery rules provide plants with nutrients, especially P, with a similar agronomic efficiency to mined and synthetic P-fertilisers. These results are especially consistent for precipitated phosphate salts & derivates in different soil and plant types, and are thus not restricted to specific agricultural settings within an EU context. Similar results are found for thermal oxidation materials & derivates, although the feedstock applied and the length of the growing season impact upon the aggregated results for specific thermal oxidation materials. The results confirmed that post-combustion manufacturing steps applied on incineration ashes improve agronomic efficiencies to values similar to those of mined phosphate rock and processed P-fertilisers. For P-rich pyrolysis & gasification materials, it was indicated that plant responses can vary widely depending on the feedstock and production conditions. Some pyrolysis & gasification materials show similar efficiencies to mined phosphate rock and processed P-fertilisers, whereas others are not as effective relative to the latter in every agronomic environment. Nonetheless, responsible use practices for such fertilisers can also contribute to reducing the reliance on primary raw materials under specific situations and agronomic settings, as part of targeted agricultural practices to increase nutrient use efficiency in the EU. STRUBIAS materials with a low nutrient content may serve other fertilising functions (e.g. soil improver, liming material, growing media, plant biostimulant), thus contributing to improving plant nutrient uptake efficiency in the short or long-term under specific situations. For carbon-rich pyrolysis & gasification materials derived from vegetable residues, the added value for the European agricultural sector varies widely across settings as a function of soil type, climate and targeted application (e.g. growing media versus open field spreading). It is concluded that all three STRUBIAS material groups provide plants with nutrients or improve their nutrition efficiency, although the latter may only hold true under specific European settings for some pyrolysis & gasification materials. STRUBIAS materials are not only an added value material for conventional European agriculture, but also for the expanding organic farming sector in Europe.

Regarding the third criterion, **market demand and trade** is expected for all three STRUBIAS material groups in different segments of the EU agricultural sector. Quantitatively assessing the expected trade of STRUBIAS materials on the internal market is challenging, partly because many STRUBIAS materials are produced during integrated processes with a different primary aim (e.g. waste management and control, biogas production, manure hygienisation). It is expected that the overall share of the STRUBIAS materials will be used as fertilisers that can be used to provide nutrients, mostly P, to the conventional European agricultural sector. As a best estimate for the year 2030, the opening of the P-fertiliser market to STRUBIAS materials will result in a substitution effect of mined rock phosphate and processed P-fertilisers by fertilising products containing precipitated phosphate salts & derivates, and thermal oxidation materials & derivates of 17% to 31%. STRUBIAS materials expected to be on the market in 2030 will be mostly derived from municipal wastewaters and sludges and manure, with the former being a more important feedstock than the latter. Most P-recovery in the form of STRUBIAS materials is likely to take place in western Europe due to increasing concerns associated with the landspreading of unprocessed biogenic materials and the region-situation specificity of high livestock and population densities. As a consequence, some biogenic materials, such as sewage sludge, are increasingly being incinerated and the resulting ashes are transferred to landfills and construction materials. Moreover, possible unbalanced nutrient stoichiometry and spatial constraints on landspreading of manure often hamper sustainable circular nutrient management through enhancing P accumulation in soils and eutrophication under these regional conditions. A certain market demand is also expected for STRUBIAS materials to be used as liming materials, soil improvers or growing media in specific agricultural sectors. For specific fertiliser materials, including pyrolysis & gasification materials, the organic farming sector could be a potential trade market. The development of STRUBIAS production pathways is partly dependent on production costs. Based on the momentary evaluation, STRUBIAS production costs are for most – but not for all – pathways more expensive compared to mined rock phosphate and processed P-fertilisers. The supplementary life cycle cost of P-recycling through STRUBIAS pathways is lowest in regions characterised by nutrient surpluses. Nonetheless, feedstock availability and decisions that impact upon the use routes for eligible input materials for STRUBIAS materials will also determine the market for STRUBIAS materials. With the intention to promote a more circular and resource-efficient economy, policy targets, financial incentives or financial disincentives for the handling of biogenic materials will impact upon the STRUBIAS market outlook. Specifically, the EU and national legislative framework and policies for waste management, ground and surface water quality, and renewable energy targets are expected to greatly influence the STRUBIAS market and trade potential.

In order to better understand the trade-offs between **resource management and health and environmental protection**, life cycle analyses were performed. Scenario modelling was applied for nine selected STRUBIAS materials of a high technological readiness level that show significant market potential. The life cycle stages include transport, feedstock material transformation and STRUBIAS manufacturing stages, transport, storage at the farm, land application, and use-on-land. The analyses identified sector opportunities and mechanisms that maximise the socio-environmental benefits of emerging P-recovery pathways. Because the P contained in biogenic materials is to a large extent dissipated in nutrient-surplus regions, the greatest opportunities for P recycling through STRUBIAS pathways occur in those EU areas. The implementation of specific STRUBIAS pathways to reduce the dependence on phosphate rock can provide coinciding benefits for human health and environmental protection relative to mined rock phosphate and processed P-fertilisers. The impacts and costs are largely dependent on the local situation, and opportunities for STRUBIAS are indicated when current handling scenarios for biogenic materials show a low resource and nutrient use efficiency (e.g. incineration of sewage sludge), or are transported over long distances to areas suitable for land application (e.g. manure exports due to soil nutrient excess). In other settings, the analysis confirmed that trade-offs between impacts on phosphate rock depletion, global warming, eutrophication and human health occur for some STRUBIAS pathways. Also, those pathways for producing STRUBIAS materials thus provide opportunities for addressing critical environmental and/or human health issues, although at the expense of adverse impacts on other matters. This implies that the implementation of STRUBIAS pathways is likely to depend on the needs and priorities of local stakeholders, and thus the nexus of costs and impacts upon human health, agronomic yields, water quality, energy balances, resource depletion, climate regulation and long-term food security. The main mechanisms that contribute positively to environmental and human health savings relate to increasing the bio-availability of the nutrients contained in the biogenic input materials and reducing the metals contained therein. There are also risks of adverse and unintended negative effects, however, in processes involving the removal of other valuable materials in the biogenic input materials (e.g. nitrogen, organic carbon), reduced levels of contaminant removal from the biogeochemical cycle relative to counterfactual feedstock handling scenarios, and effects related to additional manufacturing steps that are associated with high chemical or energetic demands. Relative to the counterfactual scenarios of the direct landspreading of unprocessed and digested biogenic materials, STRUBIAS production pathways mitigate phosphorus eutrophication and may show reduced impacts on human health by effectively removing biological pathogens, pharmaceutical and personal care compounds, other persistent and emerging organic pollutants, and possibly metals.

The possible opening of the EU fertilising market to STRUBIAS materials might be associated with **economic benefits and challenges**. Harmonisation of the legislation concerning all fertilising materials at EU level will increase the transparency on product quality within the EU fertilising market and thus decrease cross-border transaction costs for STRUBIAS materials. The possibility of producing STRUBIAS materials under the EU legislative framework might prevent additional costs for producers of eligible input materials for STRUBIAS pathways, thus reducing costs for disposal and waste compliance, especially for sewage sludge and manure. The addition of STRUBIAS materials as possible component materials for EU fertilising products might also stimulate industry innovation to possibly develop new fertilising products, including those with a tailored chemical and physical composition. A major challenge lies in the fact that actors belonging to different sectors form part of the transformation cascade of biogenic materials into STRUBIAS materials (e.g. agriculture, wastewater treatment sector, waste management sector, food processing industry) and will have to establish agreements in order to synchronise material streams, considering both quantitative and qualitative aspects.

Overall it is concluded that the implementation of the manufacturing and use of STRUBIAS materials in the EU agricultural sector is closely linked to progression towards a more circular economy, and a reduced dependence on phosphate rock as a finite primary raw material that is essential to sustain food production. STRUBIAS materials that meet the proposed technical requirements offer an avenue to enhanced food security and sustainable fertilisation, while respecting the environment and its natural resources in the EU and elsewhere. **A stable legal framework for all three STRUBIAS material groups is therefore desirable to promote trade and use of these materials in the EU agricultural sector, and to provide a material quality benchmark for producers and consumers of P-fertilising products containing STRUBIAS materials**.

2 Draft proposals for STRUBIAS recovery rules

2.1 CMC XX: Precipitated phosphate salts & derivates

1. An EU fertilising product may contain precipitated phosphate salts exclusively obtained through precipitation from one or more of the following input materials:

a) wastewaters and sewage sludge from municipal wastewater treatment plants;
b) derived products referred to in Article 32 of Regulation (EC) No 1069/2009 for which an end-point in the manufacturing chain has been determined in accordance with the third subparagraph of Article 5(2) of that Regulation;
c) animal by-products, the products derived from which are referred to in Article 32 of Regulation (EC) No 1069/2009 and have an end-point in the manufacturing chain determined in accordance with the twastewaterhird subparagraph of Article 5(2) of that Regulation;
d) wastewaters from food processing, pet food, feed, milk and drink industries, unless the processing steps involved contact with biocides within the meaning of Regulation (EU) No 528/2012 other than those defined as product type 4 of main group 1 of Annex V thereto;
e) residues from the production of bioethanol and biodiesel as referred to in Directive 2009/28/EC of the European Parliament and of the Council of 23 April 2009 on the promotion of the use of energy from renewable sources and amending and subsequently repealing Directives 2001/77/EC and 2003/30/EC[5];
f) bio-waste within the meaning of Directive 2008/98/EC resulting from separate bio-waste collection at source, other than those materials included in points (b) and (c);
g) living or dead organisms or parts thereof, which are unprocessed or processed only by manual, mechanical or gravitational means, by dissolution in water, by flotation, by extraction with water, by steam distillation or by heating solely to remove water, or which are extracted from air by any means, except:
    i. materials originating from mixed municipal waste,
    ii. sewage sludge, industrial sludge or dredging sludge,
    iii. animal by-products or derived products within the scope of Regulation (EC) No 1069/2009 for which no end-point in the manufacturing chain has been determined in accordance with the third subparagraph of Article 5(2) of that Regulation, and
    iv. materials mentioned in points (a) to (f);
h) substances and mixtures, other than:
    i. those listed under points (a) to (g),
    ii. waste within the meaning of Directive 2008/98/EC,
    iii. substances or mixtures which have ceased to be waste in one or more Member States by virtue of the national measures transposing Article 6 of Directive 2008/98/EC,

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[5] OJ L 140 5.6.2009, p. 16.

iv. substances formed from precursors which have ceased to be waste in one or more Member States by virtue of the national measures transposing Article 6 of Directive 2008/98/EC, or mixtures containing such substances,
v. non-biodegradable polymers, and
vi. animal by-products or derived products within the scope of Regulation (EC) No 1069/2009.

In addition, precipitated phosphate salts shall be obtained through precipitation from any material listed in points (a) to (h), or combination thereof, processed by manual, mechanical or gravitational means, by solid-liquid fractionation using biodegradable polymers, by dissolution in water, by flotation, by extraction with water, by steam distillation or by heating solely to remove water, by thermal hydrolysis, by anaerobic digestion or by composting. The temperature of such processes shall not be raised above 275 °C.

[Note: The exclusion of a material from a lettered item does not prevent it from being an eligible component material by virtue of another lettered item.]

2. The precipitation process shall take place under controlled conditions in a reactor which only processes input materials referred to in paragraph 1 above, excluding materials that are knowingly contaminated with material streams not listed in paragraph 1 unless such contamination is unintentional, only results in trace levels of exogenous compounds, and constitutes a one-off incident. Physical contact between input and output materials must be avoided after the precipitation process in the treatment plant, including during storage.

3. Regardless of the input material applied, the precipitated phosphate salt shall contain:

a) a minimum P_2O_5 content of 16% of the dry matter content[6];
b) a maximum organic carbon content of 3% of the dry matter content[7];
c) no more than 3 g/kg dry matter of macroscopic impurities above 2 mm in any of the following forms: organic matter, glass, stones, metal and plastics;
d) no more than 5 g/kg dry matter of the sum of the macroscopic impurities referred to in point (c).
e) no presence of *Salmonella* spp. in a sample containing 25 g fresh mass; and
f) no presence of *Escherichia coli* or *Enterococcaceae* in a concentration of more than 1 000 CFU/g fresh mass.

Precipitated phosphate salts derived from materials listed under point a) of paragraph 1 shall have:

g) no more than 6 mg/kg dry matter of PAH_{16}[8];
h) no presence of *Clostridium perfringens* in a concentration of more than 100 CFU/g fresh mass; and
i) no presence of *Ascaris* sp. eggs in a sample containing 25 g fresh mass.

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[6] As measured using vacuum drying at 40 °C to avoid the loss of crystal-bound water.

[7] As measured using vacuum drying at 40 °C to avoid the loss of crystal-bound water.

[8] Sum of naphthalene, acenaphthylene, acenaphthene, fluorene, phenanthrene, anthracene, fluoranthene, pyrene, benzo[a]anthracene, chrysene, benzo[b]fluoranthene, benzo[k]fluoranthene, benzo[a]pyrene, indeno[1,2,3-cd]pyrene, dibenzo[a,h]anthracene and benzo[ghi]perylene.

By way of derogation from points e), f), h) and i), testing shall not be necessary for precipitated phosphate salts that have been subject to either of the following conditions:

i. Pressure sterilisation through the heating to a core temperature of more than 133 °C for at least 20 minutes without interruption at a pressure (absolute) of at least 3 bars. The pressure must be produced by the evacuation of all air in the sterilisation chamber and the replacement of the air by steam ('saturated steam'); or
ii. Processing in a pasteurisation/hygienisation unit that reaches a temperature of 70 °C during a time of at least 1 hour.

The derogation shall extend to precipitated phosphate salts that are exclusively derived from eligible input materials that have been subject to either of the conditions (i) or (ii).

4. An EU fertilising product may contain derivates from precipitated phosphate salts compliant with paragraphs 1 to 3 as produced through one or more chemical manufacturing steps that react precipitated phosphate salts with materials listed under point h) of paragraph 1 that are consumed in or used for chemical processing. The derivate manufacturing process shall be executed so as to intentionally modify the chemical composition of the precipitated phosphate salt.

5. Precipitated phosphate salts & derivates incorporated into the EU fertilising product shall have a maximum sum of elemental Al and elemental Fe that does not exceed 10% of the dry matter content.

6. All the precipitated phosphate salt & derivate substances incorporated into the EU fertilising product, on their own or in a mixture, shall have been registered pursuant to Regulation (EC) No 1907/2006, in a dossier containing:

(a) the information provided for by Annexes VI, VII and VIII to Regulation (EC) No 1907/2006, and

(b) a chemical safety report pursuant to Article 14 of Regulation (EC) No 1907/2006 covering the use as fertilising product,

unless explicitly covered by one of the registration obligation exemptions provided for by Annex IV to that Regulation or by point 6, 7, 8, or 9 of Annex V to that Regulation.

2.2 CMC YY: Thermal oxidation materials & derivates

1. An EU fertilising product may contain thermal oxidation materials exclusively obtained through thermochemical conversion under non-oxygen-limiting conditions from one or more of the following input materials:

a) living or dead organisms or parts thereof, which are unprocessed or processed only by manual, mechanical or gravitational means, by dissolution in water, by flotation, by extraction with water, by steam distillation or by heating solely to remove water, or which are extracted from air by any means, except:
  - materials originating from mixed municipal waste,
  - sewage sludge, industrial sludge or dredging sludge,
  - animal by-products or derived products falling within the scope of Regulation (EC) No 1069/2009, and
  - materials separately listed under points e) to j);

b) vegetable waste from the food processing industry and fibrous vegetable waste from virgin pulp production and from production of paper from virgin pulp;

c) derived products referred to in Article 32 of Regulation (EC) No 1069/2009 for which an end-point in the manufacturing chain has been determined in accordance with the third subparagraph of Article 5(2) of that Regulation;

d) animal by-products, the products derived from which are referred to in Article 32 of Regulation (EC) No 1069/2009 and have an end-point in the manufacturing chain determined in accordance with the third subparagraph of Article 5(2) of that Regulation;

e) bio-waste within the meaning of Directive 2008/98/EC resulting from separate bio-waste collection at source;

f) residues from composting, anaerobic digestion, pyrolysis or gasification as a pretreatment technique of the input materials listed under point a) to d);

g) sewage sludge from municipal wastewater treatment plants;

h) waste within the meaning of Directive 2008/98/EC with the exception of:
  - those listed under points a) to g),
  - materials which display one or more of the hazardous properties listed in Annex III to Directive 2008/98/EC,
  - materials originating from mixed municipal waste, and
  - animal by-products or derived products within the scope of Regulation (EC) No 1069/2009.

i) auxiliary fuels (natural gas, liquefied petroleum gas, natural gas condensate, process gases and components thereof, crude oil, coal, coke as well as their derived materials), when used in incineration, co-incineration or biomass combustion plants to process input materials listed under points a) to h);

j) substances which occur in nature which are used in production processes of the iron and steel industry; or

k) substances which occur in nature and chemical substances, with the exception of:
  - those listed under points a) to j),
  - waste within the meaning of Directive 2008/98/EC,
  - substances or mixtures which have ceased to be waste in one or more Member States by virtue of the national measures transposing Article 6 of Directive 2008/98/EC,
  - substances formed from precursors which have ceased to be waste in one or more Member States by virtue of the national measures

transposing Article 6 of Directive 2008/98/EC, or mixtures containing such substances,
- non-biodegradable polymers, and
- animal by-products or derived products within the scope of Regulation (EC) No 1069/2009.

[Note: The exclusion of a material from a lettered item does not prevent it from being an eligible component material by virtue of another lettered item.]

2. The thermal oxidation shall take place under non-oxygen-limiting conditions in such a way that the gas resulting from the thermochemical conversion process is raised, after the last injection of combustion air, in a controlled and homogeneous fashion and even under the most unfavourable conditions to a temperature of at least 850 °C for at least 2 seconds or 1 100 °C for at least 0.2 seconds. These conditions shall apply to all input materials, with the exemption of those listed under points a), b) and i) of paragraph 1 and derived materials resulting from their composting, anaerobic digestion or pyrolysis. For the latter exempted materials, a minimum temperature of 450 °C for at least 2 seconds shall apply.

3. The thermal oxidation unit operator shall:
- only process input materials referred to in paragraph 1 above, excluding materials that are knowingly contaminated with material streams not listed in paragraph 1 unless such contamination is unintentional, only results in trace levels of exogenous compounds, and constitutes a one-off incident;
- ensure that the input material is oxidised in such a way that the total organic carbon content of the slags and bottom ashes is less than 3%; and
- ensure that physical contact between input and output materials is avoided after the thermochemical conversion process, including during storage.

4. The thermal oxidation materials shall have:
- no more than 6 mg/kg dry matter of PAH_{16}[9], and
- no more than 20 ng WHO toxicity equivalents/kg dry matter of PCDD/F[10].

5. An EU fertilising product may contain derivates from thermal oxidation materials that have been produced from the input materials listed in paragraph 1 and compliant with paragraph 4 and that have been manufactured according to a thermochemical conversion process compliant with paragraphs 2 and 3. The derivate manufacturing process shall be executed so as to intentionally modify the chemical composition of the thermal oxidation material, and be of the following nature:

a) Chemical manufacturing: derivates as produced through one or more chemical manufacturing steps that react thermal oxidation materials with materials listed under point k) of paragraph 1 that are consumed in or used for chemical processing. Non-biodegradable polymers shall not be used.

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[9] Sum of naphthalene, acenaphthylene, acenaphthene, fluorene, phenanthrene, anthracene, fluoranthene, pyrene, benzo[a]anthracene, chrysene, benzo[b]fluoranthene, benzo[k]fluoranthene, benzo[a]pyrene, indeno[1,2,3-cd]pyrene, dibenzo[a,h]anthracene and benzo[ghi]perylene.

[10] van den Berg M., L.S. Birnbaum, M. Denison, M. De Vito, W. Farland, et al. (2006) The 2005 World Health Organization Re-evaluation of Human and Mammalian Toxic Equivalency Factors for Dioxins and Dioxin-like Compounds. Toxicological sciences: an official journal of the Society of Toxicology 93:223-241. doi:10.1093/toxsci/kfl055.

b) Thermochemical manufacturing: processes that thermochemically react thermal oxidation materials with reactants listed in paragraph 1 a) to k) that are consumed in or used for chemical processing. Thermochemical process conditions shall be compliant with paragraphs 2 and 3, and the thermal oxidation material derivate shall meet conditions listed in paragraph 4.

Thermal oxidation materials that display one or more of the hazardous properties listed in Annex III to Directive 2008/98/EC shall not be mixed or reacted, either with waste, substances or materials with the intention of reducing hazardous substances to levels below the limit values for the hazardous property as defined in that Directive. Using a mass balance approach, manufacturers that use thermal oxidation materials with hazardous properties must demonstrate the removal or transformation of the contaminants to levels below the limit values defined in Annex III to Directive 2008/98/EC.

6. Thermal oxidation materials & derivates incorporated into the EU fertilising product shall:

a) contain no more than 3% Cl^- on a dry matter basis, only applicable when Cl^- is an unintentional constituent coming from the input material(s);

b) contain no more than 400 mg kg^{-1} dry matter of total chromium (Cr), if derived from materials listed under point g), h) or j) of paragraph 1;

c) contain no more than 2 mg kg^{-1} dry matter of thallium (Tl), if derived from materials listed under point g), h), i) or j) of paragraph 1; and

d) contain no more than 600 mg kg^{-1} dry matter of vanadium (V), if derived from materials listed under point h) or j) of paragraph 1.

7. All the thermal oxidation materials & derivate substances incorporated into the EU fertilising product, on their own or in a mixture, shall have been registered pursuant to Regulation (EC) No 1907/2006, in a dossier containing:

(a) the information provided for by Annexes VI, VII and VIII to Regulation (EC) No 1907/2006, and

(b) a chemical safety report pursuant to Article 14 of Regulation (EC) No 1907/2006 covering the use as fertilising product,

unless explicitly covered by one of the registration obligation exemptions provided for by Annex IV to that Regulation or by point 6, 7, 8, or 9 of Annex V to that Regulation.

2.3 CMC ZZ: Pyrolysis & gasification materials

1. An EU fertilising product may contain materials exclusively obtained through the thermochemical conversion under oxygen-limiting conditions of one or more of the following input materials:

a) derived products referred to in Article 32 of Regulation (EC) No 1069/2009 for which an end-point in the manufacturing chain has been determined in accordance with the third subparagraph of Article 5(2) of that Regulation;

b) animal by-products, the products derived from which are referred to in Article 32 of Regulation (EC) No 1069/2009 and have an end-point in the manufacturing chain determined in accordance with the third subparagraph of Article 5(2) of that Regulation;

c) living or dead organisms or parts thereof, which are unprocessed or processed only by manual, mechanical or gravitational means, by dissolution in water, by flotation, by extraction with water, by steam distillation or by heating solely to remove water, or which are extracted from air by any means, except:
   - materials originating from mixed municipal waste,
   - sewage sludge, industrial sludge or dredging sludge, and
   - animal by-products or derived products falling within the scope of Regulation (EC) No 1069/2009;

d) vegetable waste from the food processing industry and fibrous vegetable waste from virgin pulp production and from production of paper from virgin pulp;

e) bio-waste within the meaning of Directive 2008/98/EC resulting from separate bio-waste collection at source, other than those included above;

f) residues from the production of bioethanol and biodiesel as referred to in Directive 2009/28/EC of the European Parliament and of the Council of 23 April 2009 on the promotion of the use of energy from renewable sources and amending and subsequently repealing Directives 2001/77/EC and 2003/30/EC[11]; or

g) pyrolysis/gasification additives which are necessary to improve the process performance or the environmental performance of the pyrolysis/gasification process, provided that the additives classify as intermediates within the meaning of Regulation (EC) No 1907/2006 and with the exception of:
   - those listed under points a) to f),
   - waste within the meaning of Directive 2008/98/EC,
   - substances or mixtures which have ceased to be waste in one or more Member States by virtue of the national measures transposing Article 6 of Directive 2008/98/EC,
   - substances formed from precursors which have ceased to be waste in one or more Member States by virtue of the national measures transposing Article 6 of Directive 2008/98/EC, or mixtures containing such substances,
   - non-biodegradable polymers, and
   - animal by-products or derived products within the scope of Regulation (EC) No 1069/2009.

   The total concentration of all additives must not exceed 25% of the total input material fresh weight.

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[11] OJ L 140 5.6.2009, p. 16.

In addition, an EU fertilising product may contain pyrolysis & gasification materials obtained through thermochemical conversion under oxygen-limiting conditions of any material listed in points a) to g), or combination thereof, processed by manual, mechanical or gravitational means, by solid-liquid fractionation using biodegradable polymers, by dissolution in water, by flotation, by extraction with water, by steam distillation or by heating solely to remove water, by composting, or by anaerobic digestion.

[Note: The exclusion of a material from a lettered item does not prevent it from being an eligible component material by virtue of another lettered item.]

2. The thermochemical conversion process of the input materials shall take place under oxygen-limiting conditions:
   - in such a way that a temperature of at least 180 °C for at least 2 seconds is reached in the reactor, and
   - in a pyrolysis/gasification reactor which only processes input materials referred to in paragraph 1 above, excluding materials that are knowingly contaminated with material streams not listed in paragraph 1 unless such contamination is unintentional, only results in trace levels of exogenous compounds, and constitutes a one-off incident.

Physical contact between input and output materials must be avoided after the thermochemical process, including during storage.

3. The pyrolysis & gasification materials shall have:
   - a molar ratio of H/organic C of less than 0.7, with testing to be performed in the dry and ash-free fraction for materials that have an organic C content of <50%,
   - no more than 6 mg kg^{-1} dry matter of PAH_{16}[12],
   - no more than 20 ng WHO toxicity equivalents kg^{-1} dry matter of PCDD/F[13],
   - no more than 0.8 mg kg^{-1} dry matter of dl-PCB[14], and
   - no more than 3% Cl^- on a dry matter basis,
   - no more than 2 mg kg^{-1} dry matter of thallium (Tl), in the event that more than 5% of pyrolysis/gasification additives relative to the total input material fresh weight has been applied.

4. All the pyrolysis & gasification material substances incorporated into the EU fertilising product, on their own or in a mixture, shall have been registered pursuant to Regulation (EC) No 1907/2006, in a dossier containing:

   (a) the information provided for by Annexes VI, VII and VIII to Regulation (EC) No 1907/2006, and

   (b) a chemical safety report pursuant to Article 14 of Regulation (EC) No 1907/2006 covering the use as fertilising product,

unless explicitly covered by one of the registration obligation exemptions provided for by Annex IV to that Regulation or by point 6, 7, 8, or 9 of Annex V to that Regulation.

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[12] Sum of naphthalene, acenaphthylene, acenaphthene, fluorene, phenanthrene, anthracene, fluoranthene, pyrene, benzo[a]anthracene, chrysene, benzo[b]fluoranthene, benzo[k]fluoranthene, benzo[a]pyrene, indeno[1,2,3-cd]pyrene, dibenzo[a,h]anthracene and benzo[ghi]perylene.

[13] van den Berg M., L.S. Birnbaum, M. Denison, M. De Vito, W. Farland, et al. (2006) The 2005 World Health Organization Re-evaluation of Human and Mammalian Toxic Equivalency Factors for Dioxins and Dioxin-like Compounds. Toxicological sciences: an official journal of the Society of Toxicology 93:223-241. doi:10.1093/toxsci/kfl055.

[14] Sum of congeners PCB 28, 52, 101, 138, 153, 180.

2.4 Proposals for further evaluation as end-points in the Animal By-Product Regulations

Recital (15) of the EU Fertilising Products Regulation ((EU) 2019/1009) indicates that 'For each component material category which includes derived products within the meaning of Regulation (EC) No 1069/2009, the end point in the manufacturing chain should be determined in accordance with the procedures laid down in that Regulation. Where such an end point is reached before the EU fertilising product is placed on the market but after the manufacturing process regulated under this Regulation has started, the process requirements of both Regulation (EC) No 1069/2009 and this Regulation should apply cumulatively to EU fertilising products, which means application of the stricter requirement in case both Regulations regulate the same parameter.'

Based on these provisions, the STRUBIAS recovery rules refer to eligible input materials from animal origin as 'animal by-products and derived materials referred to in Article 32 of Regulation (EC) No 1069/2009 for which an end-point in the manufacturing chain has been determined in accordance with the third subparagraph of Article 5(2) of that Regulation' (see Sections 2.1 to 2.3). It is thus implicit that any EU fertilising product of animal origin should then comply with both the technical criteria laid down in the EU Fertilising Products Regulation ((EU) 2019/1009) as well as with any criteria that will be laid down at a later stage in the Animal By-Products Regulation.

The STRUBIAS report proposes – based on robust techno-scientific evidence - alternative conditions and technical requirements for the handling, treatment, transformation, processing and storage of animal by-products or derived products and conditions for treatment of wastewater based on techno-scientific evidence. In the latter case and if considered pertinent and in line with the existing legal conditions by the Commission and the legislators, any alternative proposed method shall be assessed in line with the procedure indicated in Article 20 of Regulation (EC) 1069/2009. This involves, amongst others, an assessment by the European Food Safety Authority (EFSA) to evaluate risks associated with the food chain. EFSA collects and analyses existing research and data and provides scientific advice to support decision-making by risk managers responsible for making decisions or setting legislation about food safety.

Alternative methods should comply with the condition that sufficient evidence is available to indicate that the alternative method provides a degree of protection that is at least equivalent, for the relevant category of animal by-products, to the processing methods that are currently laid down. In this respect, category 1 animal by-product material is not further considered as input material for the present study as the co-legislator allows the production of fertilising materials from category 2 and 3 animal by-products, but not of Category 1 animal by-product material. Therefore, this report has not assessed the risks resulting from the use of unprocessed and processed (e.g. through incineration) Category 1 animal by-product material.

The following proposals are put forward for further consideration and evaluation by the Commission and other bodies as a possible end-point in the manufacturing chain in accordance with the third subparagraph of Article 5(2) of Regulation (EC) No 1069/2009:

1. Precipitated phosphate salts & derivates that have been derived from manure, non-mineralised guano, and digestive tract content pursuant to Regulation (EC) No 1069/2009 provided that they:
    - are compliant with the proposed criteria for CMC XX,
    - have no presence of *Clostridium perfringens* in a concentration of more than 100 CFU/g fresh mass, and
    - have no presence of *Ascaris* sp. eggs in a 25 g fresh mass.

2. Precipitated phosphate salts & derivates that have been derived from animal by-products and derived materials from category 2 or category 3 material as defined by Regulation (EC) No 1069/2009 provided that they:
    - are compliant with the proposed criteria for CMC XX,
    - have been hygienised in accordance with the conditions for pressure sterilisation or with other conditions to prevent risks arising to public and animal health, in accordance with the requirements laid down pursuant to Article 15 of Regulation (EC) No 1069/2009, or
    - they are digestion residues from transformation into biogas as set out in Annex V to Regulation (EU) No 142/2011.

3. Thermal oxidation materials & derivates that have been derived from animal by-products and derived materials from category 2 or category 3 material as defined by Regulation (EC) No 1069/2009 provided that they:
    - are compliant with the proposed criteria for CMC YY.

4. Pyrolysis & gasification materials that have been derived from manure, non-mineralised guano, and digestive tract content pursuant to Regulation (EC) No 1069/2009 provided that they:
    - are compliant with the proposed criteria for CMC ZZ.

5. Pyrolysis & gasification materials that have been derived from animal by-products and derived materials from category 2 or category 3 material as defined by Regulation (EC) No 1069/2009 provided that they:
    - are compliant with the proposed criteria for CMC YY,
    - have undergone one of the following treatments at a stage prior to or during the pyrolysis & gasification material production process:
        - pressure sterilisation or other conditions to prevent risks arising to public and animal health, in accordance with the requirements laid down pursuant to Article 15 of Regulation (EC) No 1069/2009,
        - transformation into biogas or compost as set out set out in Annex V to (EU) No 142/2011, or
        - Brookes' gasification process as described in point E, Section 2, Chapter IV, of Annex IV to Regulation (EU) No 142/2011.

2.5 Labelling requirements at PFC level

- All product function categories shall have a declaration when the neutralising value > 15 (equivalent CaO) or > 9 (equivalent HO-).
- EU fertilising materials derived from CMC thermal oxidation materials & derivates and pyrolysis & gasification materials that have Mn contents above 3.5% shall declare the Mn content of the product.
- EU fertilising products that contain more than 50% pyrolysis & gasification materials shall document the instructions for the intended use of the CE pyrolysis & gasification materials.

It is proposed that **the tolerance rules** as outlined in Part 3 of Annex III on labelling requirements shall, in principle, also apply to STRUBIAS materials. The STRUBIAS subgroup highlighted, however, the need to test, validate, and possibly develop new Harmonised Standards for each of the testing requirements for the STRUBIAS material groups. Therefore, it is proposed to re-evaluate the assessment of the tolerance rules in case such work indicated:
(i) a significantly higher degree in sampling variability, or
(ii) a significantly lower degree of analytical precision when analysing the nutrient content or physico-chemical characteristics,
for STRUBIAS materials relative to other CMCs included in the EU Fertilising Products Regulation ((EU) 2019/1009).

2.6 Conformity assessment procedure

Module D1 for all three different STRUBIAS CMCs, with the additional elements of the quality assurance process as indicated in Section 5.7.2 ‘Additional elements of the quality management system and auditing’.

OBJECTIVES AND METHODOLOGY

3 Objectives and methodology of the JRC STRUBIAS work

3.1 Objectives

The Joint Research Centre (JRC) of the European Commission initiated an assessment of the existing techno-scientific evidence in view of a possible inclusion of certain recovered materials as Component Material Categories (CMCs) in the EU Fertilising Products Regulation ((EU) 2019/1009)[15]. This assessment should form the basis for any technical proposals on the requirements that those candidate materials shall comply with. The acronym chosen for the project was STRUBIAS, based on the scope initially proposed at the launch of the project, namely STRUvite, BIochar and ASh materials. While this scope has been refined during the course of the project, the STRUBIAS acronym has been maintained for simplicity and recognisability.

The JRC has been supported in the process by a technical working group that constitutes a subgroup of the Commission expert group on Fertilisers (hereafter **STRUBIAS subgroup)**, which includes representatives from EU Member States, EU trade/business/professional associations, as well as from other institutions such as think tanks, research and academic institutions. The role of the subgroup was to participate in the process of **sharing knowledge and providing non-binding expert advice to the European Commission** on possible recovery rules for nutrients from eligible input materials into STRUBIAS materials. If included in the EU Fertilising Products Regulation ((EU) 2019/1009), STRUBIAS materials may be used as **component materials for the different Product Function Categories (PFCs)** listed in the EU Fertilising Products Regulation (EU) 2019/1009, more specifically fertiliser, liming material, soil improver, growing medium, agronomic additive, plant biostimulant, and fertilising product blend.

The adoption of the EU Fertilising Products Regulation ((EU) 2019/1009) aims at establishing **a regulatory framework enabling production of fertilising products from recycled bio-wastes and other secondary raw materials**, in line with the Bioeconomy strategy[16], which encompasses the production of renewable biological resources and the conversion of these resources and waste streams into value added products. In Regulation (EC) No 2003/2003 of 13 October 2003 relating to fertilisers, regulatory barriers to market rollout were noted for those materials.

In 2013, the European Commission organised a **Consultative Communication** that set out for the first time at EU level the issues around the sustainability of P use (European Commission, 2013a). The intention was to launch a debate on the state of play and the actions that should be considered. The European Institutions and all those interested – organisations or private individuals – were invited to submit their comments on the questions set out in the Consultative Communication, as well as on any other issues that they wished to raise concerning the sustainable use of P. Phosphorus recycling has also been addressed by FP7 research projects, the results of which were analysed during the 'Circular approaches to

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[15] https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELLAR:e351eb07-9713-11e9-9369-01aa75ed71a1

[16] http://ec.europa.eu/research/bioeconomy/index.cfm

phosphorus: from research to deployment' workshop, held in Berlin on 4 March 2015. Answers provided by European institutions on the questions set out in the Consultative Communication and workshop presentations indicated that promising technical progress is being made in the field of recycling of waste. Amongst others, proposed actions included removal of P from wastewater in the form of **struvite**, chemical processing of **incinerating** ashes of sewage sludge, and fertilising product production from animal by-products in the form of **biochar** through pyrolysis processes. This would boost domestic sourcing of plant nutrients which are essential for sustainable European agriculture, including P. It would also contribute to a better implementation of the waste hierarchy, by minimising landfilling or energy recovery of bio-wastes, and hence to solving related waste management problems.

Based on this information, the European Commission decided to evaluate a possible legal framework **for the production of safe and effective fertilisers from recovered, secondary raw materials, such as struvite, ashes and biochar**. These could possibly be considered CMCs in the EU Fertilising Products Regulation ((EU) 2019/1009). The acronym STRUBIAS (STRUvite, BIochar, or incineration Ashes)[17] was initially chosen as the working title for this project. Throughout the course of the STRUBIAS project, the scope of these CMCs has, however, been further refined (see Section 5.1).

STRUBIAS materials are mainly manufactured from specific secondary raw materials, including 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. The work delivered within this project should contribute to making the recovery of nutrients and organic matter from secondary raw materials a more attractive business across Europe. 'Closing the loop – An EU action plan for the **circular economy**', as adopted by the European Commission[18], has identified the Fertilisers Regulation revision as a key legislative proposal to boost the market for secondary raw materials, and the revised Waste Framework Directive[19] establishes ambitious targets for recycling.

Several STRUBIAS materials show a substantial potential to provide safe sources of **phosphorus** (P) that can constitute an **alternative for the primary raw material phosphate rock**. Both phosphorus and phosphate rock have been identified by the European Commission as critical raw materials, based on their supply risk and the economic importance for EU operators in particular.

The JRC has assessed STRUBIAS materials incorporated into CE fertilising products against the following criteria:

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[17] Note that the acronym STRUBIAS was initially chosen as the working title for this project and has been maintained for simplicity reasons, despite a refined possible scope of the different groups agreed at the STRUBIAS Kick-off Meeting (Seville, July 2016).

[18] More information on: http://ec.europa.eu/environment/circular-economy/index_en.htm

[19] Directive (EU) 2018/851 of the European Parliament and of the Council of 30 May 2018 amending Directive 2008/98/EC on waste.

I. *The use of the materials will not lead to overall adverse environmental or human health impacts*.

II. *The material shall provide plants/mushrooms with nutrients or improve their nutrition efficiency, either on its own or mixed with another material [following the definition of fertilising products in the EU Fertilising Products Regulation ((EU) 2019/1009)].*

III. *Trade on the internal market can be expected for such a fertilising material, based on the current market and the future market and trade forecasts.*

The following main benefits are expected from the introduction of EU-wide criteria for fertilising materials derived from secondary raw materials:

- A reduced **dependence on primary raw materials**, principally phosphate rock.
- Improved **functioning of the internal fertilising market** by enabling a market entry for safe fertilisers derived from secondary raw materials.
- A **stable legal framework** that provides legal certainty to the industry that manufactures fertilisers derived from secondary raw materials.
- **Reinforcing consumer confidence by ensuring high quality and safety for** secondary raw materials in accordance with the relevant articles of the EU Treaty for the functioning of the EU.
- **Reduction of administrative burdens** related to shipment, transport and trade that are redundant for environmentally safe materials.

The final JRC report resulting from the STRUBIAS project aims to support the policymaking process within the Commission, in particular the implementation of the EU Fertilising Products Regulation ((EU) 2019/1009). However, none of the proposals made in this report have any binding character whatsoever. **The final decision on a proposed incorporation of precipitated phosphate salts & derivates, thermal oxidation materials & derivates, and pyrolysis & gasification materials in the EU Fertilising Products Regulation ((EU) 2019/1009) will be taken by the Commission and could be based on the outcomes of the final JRC report.** However, the Commission is able to make further changes to the proposals in the present document for the legal requirements that those materials shall meet. As a matter of fact, changes will possibly be required to streamline the incorporation of STRUBIAS materials within the general framework of the recently approved Regulation.

3.2 Methodology applied

The JRC embarked on the STRUBIAS project by preparing a **Background Document** for the Kick-off Meeting held in July 2016, which included information related to (1) the nature of the possible input materials for the recovery of nutrients, organic matter, and acid neutralising value, (2) the quantitative share of these input materials that is currently dissipated in the EU and could potentially be used for the production of STRUBIAS materials, (3) the technical description of the different production processes as well as any applied pre- and post-processing techniques, (4) the safety and quality of materials that might be incorporated into CE fertilising products as a function of production process conditions and (5) market aspects.

The Background Document was distributed to the STRUBIAS subgroup for discussion at the Kick-off Meeting and **a written consultation round**. The STRUBIAS subgroup was asked to **correct any obsolete data** within the document, **complement** the document with additional information and to **respond to questions** where supplementary information was requested from the STRUBIAS subgroup on production process conditions and product quality. Moreover, a standardised Excel template for **data collection** was circulated to facilitate the data input from the STRUBIAS subgroup.

Based on the confidential and non-confidential data received from the STRUBIAS subgroup and complementary information found in scientific literature, the JRC drafted **a proposal for recovery rules for each of the STRUBIAS materials (STRUBIAS Interim Report released in May 2017) and a draft report on market aspects (STRUBIAS Interim Report released in December 2017)**. The STRUBIAS subgroup then had the opportunity to comment on the STRUBIAS Interim Report, including the possibility to highlight obsolete information, submit further supporting data, and to make proposals to further complement the reports. Based on the information collected, the JRC redacted the **STRUBIAS pre-final report** that was discussed at the 3-day STRUBIAS final meeting (25-27 September 2018) and subjected to an additional STRUBIAS subgroup written consultation round. The JRC has then taken all techno-scientific feedback and discussion arguments sent within the deadline into consideration for the redaction of the present document.

At all times, the option was given to provide data in a **confidential manner**, for which reason neither all the information received by the JRC has been uploaded on the CIRCABC platform, nor will it be cited in the follow-up documents of this project. Such data are interpreted in a qualitative manner with no reference to the source provider or process technology.

Each of these STRUBIAS subgroup comments has been assessed by the JRC, and **credible and relevant information** has been taken into consideration during the drafting of the pre-final STRUBIAS report. Expert judgement by the JRC and the STRUBIAS subgroup has played a key role in each of these steps and the way in which the information is presented. The information laid down in this document has been **collated and assessed by the European Commission's Joint Research Centre who led the work on the STRUBIAS project, guided by the principles of technical expertise, transparency and neutrality.** The work of the **STRUBIAS subgroup and all other contributors is gratefully acknowledged**.

3.3 Structure of the report

As for the structure of this report, **Part A** focuses on the **technical proposals** for the requirements that STRUBIAS materials shall comply with for their possible incorporation in the EU Fertilising Products Regulation ((EU) 2019/1009). The Commission has revised the original Fertilisers Regulation ((EC) 2003/2003) and expanded its scope to secondary-raw-material-based fertilising materials, including fertilisers made from waste. Section 4 outlines the **structure of the EU Fertilising Products Regulation ((EU) 2019/1009)** and indicates how STRUBIAS materials could enter into the framework of that Regulation. For each candidate STRUBIAS material, recovery rules have been proposed that cover (i) the **eligible input materials** that can be applied for the production of STRUBIAS materials, (ii) the **production process conditions**, (iii) the direct **safety and quality requirements** of the end material of the production process that can be contained in an EU fertilising product as well as labelling requirements, and (iv) the **conformity assessment procedures** that shall apply to control points (i) to (iii) (Section 5). Section 5.1 indicates the **scope** expansion from that of the original proposed candidate materials, and the associated modification of the category names. Sections 5.2.2 to 5.5 then discuss the technical proposals for each of the STRUBIAS materials for points (i) to (iii) outlined above, followed by the proposed quality management system (Section 5.6) and links to other EU legislation of interest (Section 5.8).

Part B builds on Part A and supplements it by providing for an extensive description of the **impacts of the possible incorporation of STRUBIAS materials in the EU Fertilising Products Regulation ((EU) 2019/1009)** for European agriculture, the environment and human health, and the fertilisers industry. Section 6 focuses on the **agronomic efficiency** of STRUBIAS materials to support plant yields and nutrient uptake in European agroecosystems using meta-analyses techniques. Section 7 provides an overview of the current **market for STRUBIAS materials**, as well as their potential to replace conventional fertilisers resulting from a possible opening-up of the EU fertilising market to such products. Section 8 relies on the use of life cycle assessment and life cycle costing to assess the **environmental and human health impacts** of replacing conventional fertilisers with fertilising products containing STRUBIAS materials, and estimates the production costs for STRUBIAS materials. Section 9 projects possible **impacts on the economy** of replacing conventional fertilisers with recovered fertilising products, and focuses on sales prices and compliance costs for fertilisers on the EU market.

Finally, Section 10 gives **general conclusions** on the possibility of developing STRUBIAS recovery rules to protect environmental and human health safety, the agronomic effectiveness and added value of STRUBIAS materials for the European agricultural sector, and the trade on the internal market that can be expected for each of the STRUBIAS materials.

The report is **annexed** by sections that focus on a detailed description of the **properties and the current fate of eligible input materials** for STRUBIAS production processes (Section 13) and **STRUBIAS production processes and techniques** (Section 14). These sections help to assess the suitability and technical feasibility of recovering nutrients, liming material, and organic matter from different input materials and through the different STRUBIAS production pathways. The **chemical properties of the STRUBIAS materials** as per macroelements and contaminant levels are annexed in Section 15. Finally, Section 16 provides details on specific **methods** applied in the main report.

PART A: TECHNICAL PROPOSALS

4 STRUBIAS as component material categories in the EU Fertilising Products Regulation (EU) 2019/1009

4.1 Definition and principles of recovery rules

The EU Fertilising Products Regulation ((EU) 2019/1009)[20] defines 'fertilising product' as: 'a substance, mixture, micro-organism or any other material, applied or intended to be applied on plants or their rhizosphere or on mushrooms or their mycosphere, or intended to constitute the rhizosphere or mycosphere, either on its own or mixed with another material, for the **purpose of providing the plants or mushrooms with nutrient or improving their nutrition efficiency**'. According to the EU Fertilising Products Regulation ((EU) 2019/1009), the provisions on **product criteria for EU fertilising products** contain requirements for the categories of end-products in accordance with their **intended functions ('Product Function Categories' – PFCs)**, as well for the categories of **component materials ('Component Material Categories' – CMCs)** that can be contained in EU fertilising products. Section 4.2 provides a brief introduction to the structure of the EU Fertilising Products Regulation ((EU) 2019/1009).

STRUBIAS materials may be used as *component materials* in EU fertilising products when they are compliant with the specifications contained in the 'recovery rules' of the corresponding material. The STRUBIAS recovery rules thus describe possible specific **CMC requirements that shall be fulfilled by the STRUBIAS materials which are used as ingredients in EU fertiliser products.** Such products could bear the CE mark after the relevant conformity assessment procedure has been performed, including an assessment of the compliance with the criteria laid down at PFC level. EU fertilising products could then circulate freely in the single market.

Where health and safety, the protection of consumers or of the environment, other aspects of public interest, or clarity and practicability so require, detailed technical specifications may be set out in the legislation concerned.

The STRUBIAS recovery rules shall describe the following:

i. The eligible input materials that can be applied for the production of STRUBIAS materials as well as any input materials that are ineligible.

ii. The production process conditions and parameters that shall be applied during the production phase of the STRUBIAS materials.

iii. The direct safety and quality requirements of the end material of the production process that can be contained in an EU fertilising product.

iv. The useful information, where relevant, to be incorporated in the labelling requirements for the provision of information to retailers and end users. It is noted that labelling requirements only apply at PFC level, but that the

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

framework enables cross-referencing to certain CMCs in the labelling requirements.

v. The conformity assessment procedures that shall apply to monitor points i. to iv.

The STRUBIAS subgroup proposed that recovery rules should be a set of requirements that are in line with following general principles:

- Agronomic efficacy and limits on contaminants and pathogens must be ensured for STRUBIAS materials contained in CE fertilising materials so that farmers in Europe always have access to **high-quality products**, and that the use of secondary raw materials does **not lead to overall adverse environmental impacts or human health risks**. A lack of consideration of these aspects may reduce farmers' confidence and create low market acceptance for innovative fertilisers, ultimately undermining the objective of nutrient recycling.
- A **level playing field that ensures high-quality standards for EU fertilising products** derived from primary and secondary raw materials shall offer simplicity and clarity to producers and consumers that are active on the European fertilising market.
- At the same time, requirements shall be set in a sufficiently **flexible** manner to **encourage industry to undertake nutrient recycling actions** that will contribute to achieving the policy goals set in the framework of the Circular Economy Action Plan. It is not advisable to put unnecessary restrictions that might block the emerging STRUBIAS market.
- Recovery rules shall, in principle, apply a neutral stance towards all existing and future technological systems operating on the market and input materials available (**technologically neutral recovery rules**). Such an approach stimulates competition and technological innovation, and takes into consideration that process conditions and technologies for nutrient recovery on the emergent STRUBIAS market might require further adjustments, especially if alternative input materials are used.
- Recovery rules have to **be clear, concise and enforceable**, in order to clearly **delimit** the scope of the CMC in concordance with its name, lead to reasonable compliance costs, and facilitate straightforward conformity assessments.

This proposal intends to bring forward a set of requirements that takes into account the above principles.

The CE product status shall only apply to those materials that meet all the requirements for relevant **PFCs,** in particular the **conformity assessment** requirements applicable to fertilising products. This implies that materials that meet the requirements of the CMC, but not those of the PFC, shall still have the same status as the input material from which they have been manufactured (e.g. waste status for CMCs derived from waste input materials; unless these have achieved End-of-Waste status at Member State level). This is in line with Article 19 of the EU Fertilising Products Regulation ((EU) 2019/1009) stating that ‘This Regulation lays down criteria in accordance with which material that constitutes waste, as defined in Directive 2008/98/EC, can cease to be waste, if it is contained in a compliant EU fertilising product. In such cases, the recovery operation under this Regulation shall be performed before the material ceases to be waste, and the material shall be considered to comply with the conditions laid down in Article 6 of that Directive and therefore to have ceased to be waste from the moment that the EU declaration of conformity was drawn up.’ CMC materials that are waste are thus still subject to waste legislation, including possible restrictions on, for example, transport and permit requirements for processing such materials. Therefore, the recovery rules proposed in this document **cannot be interpreted as possible ‘End-of-Waste’ criteria**, as they do not go as far as attributing PFC product status, but stop at providing CMC material status.

STRUBIAS materials have only been entering the market since relatively recently, which explains why so little information might be available for certain pollutants of concern. In some cases, it remains unclear to what extent the nature of the manufacturing process may aid the removal or selective exclusion of certain contaminants from the end material, and to what extent the nature of the input material influences the quality of the end material. For this reason, the **precautionary principle** is important when evaluating environmental and human health aspects, meaning that sufficient scientific data should be available prior to the establishment of criteria for STRUBIAS materials.

4.2 The framework and outline of the EU Fertilising Products Regulation ((EU) 2019/1009)

The EU Fertilising Products Regulation ((EU) 2019/1009) was published on 25 June 2019 and is publically available on the European Commission’s website (https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELLAR:e351eb07-9713-11e9-9369-01aa75ed71a1). For a full understanding of this report and the legal requirements that shall be met for EU fertilising materials, the reader is encouraged to review the referenced Regulation, as well as its Annexes. In this section, some specific items and articles of the Regulation will be highlighted in order to facilitate a better understanding of the design of the recovery rules.

4.2.1 Component material categories and product function categories

According to the EU Fertilising Products Regulation ((EU) 2019/1009), the provisions on **product criteria for EU fertilising products** contain requirements for the categories of end-products in accordance with their **intended function (PFC)**, as well for the categories of **component materials (CMC)**. Specific requirements for each of the CMCs apply because **different component materials warrant different process requirements and control mechanisms adapted to their different potential hazardousness and variability,** in turn dependent on the quality of the input materials applied, production process conditions, etc. Component materials for EU fertilising products should, therefore, be divided into different categories.

A fertiliser manufacturer can bring an EU fertilising product that is composed of one single ingredient, belonging to a specific CMC, onto the market. A relevant example is struvite. In this case, the EU fertiliser shall have to comply with the requirements laid down for the CMC ‘precipitated phosphate salts’ as well as for an eligible PFC, in this case PFC 1 C - inorganic fertilisers. It is also possible to put an EU fertilising product on the market that is composed of several component materials from various CMCs, where each material complies with the requirements of a certain category. A condition is, however, that **no intentional chemical reaction or transformation takes place between the different CMCs that are contained in the EU fertiliser**. Hence, an EU fertiliser producer may start from two or more substances or mixtures, provided that each of them complies with the description in one or more of the CMCs, and mix them into a final product without any intentional chemical reaction taking place. The component materials are then ‘contained’ as such in the final EU fertilising product. This follows the presumption that if different component materials do not show unacceptable risks for human health and the environment, a physical mix of them constituting the final CE marked product will also be safe, subject to compliance with certain limit values defined in Annex I (i.e. PFC level) for the final product. A relevant example is, for instance, the blending of compost (CMC 3) with a triple superphosphate (CMC 1 – virgin materials) to produce an organo-mineral fertiliser belonging to PFC 1 B.

The general framework of the EU Fertilising Products Regulation ((EU) 2019/1009) has important implications for the design of the recovery rules and the use of STRUBIAS materials used as ingredients for fertiliser production processes:

- **Treatments usually considered in recovery operations, such as intended chemical or thermochemical reactions, cannot take place on CMC materials as they would intentionally modify the chemical material properties, disqualifying such processed materials from being 'contained' as such in the CE fertilising material.** STRUBIAS CMC materials should, therefore, meet certain quality requirements **so that they can be used directly without any further processing other than normal industrial practice**. Normal industrial practice can include all steps which a producer would take for a product, such as the material being screened, sized, agglomerated, pelletised, dried solely to remove free water, or adding materials necessary for further use through physical mixing without intentionally changing the chemical composition of the material contained in the mixture. A concrete outcome of this provision is that any materials that pose a risk for human health or the environment cannot be considered CMCs (e.g. unprocessed sewage sludge incineration ash from default mono-incineration not designed for P recovery) because further chemical or thermochemical processing steps may be required before such materials could meet the PFC contaminant levels. The STRUBIAS recovery rules shall, therefore, encompass any manufacturing steps required to produce a material that can be used as a fertilising material, i.e. to be placed on the market directly without any further processing other than normal industrial practice. Therefore, **the recovery rules shall describe process and product requirements until the end stage in a manufacturing process of a fertilising material**. Concretely, this implies, for instance, that recovery rules shall include provisions for manufacturing steps on raw sewage sludge ashes, and shall place product requirements on a triple superphosphate material that is, partially or entirely, produced from incinerated sewage sludge. See Section 5.2.4 for a more detailed explanation on the incorporation of such manufacturing processes that consist of multiple steps.

- Given that CMCs shall meet quality requirements so that they can be used directly without any further processing other than normal industrial practice, operators - including those that recover fertilising materials from waste-based materials - **will have the possibility to perform the conformity assessment procedure associated with a selected PFC**. The **CE marking**, indicating the conformity of a fertilising product, is the visible consequence of a whole process comprising conformity assessment in a broad sense. Where compliance of such a fertilising product with the applicable requirements laid down for a CMC and a PFC has been demonstrated, manufacturers may affix the CE marking. In accordance with Article 19 of the EU Fertilising Products Regulation ((EU) 2019/1009), any possible waste status ceases to apply and the resulting material can be sold as a product on the internal market. A company that does not have a permit for transport or treatment of waste materials can then **buy the EU fertilising material containing the STRUBIAS material** and subsequently use it in the following ways:

i. As an input material to be used by a fertiliser blending company that **physically mixes** the product into a fertilising product blend of PFC 7, without any intentional chemical reaction taking place. The intention behind PFC 7 is to allow manufacturers to rely on conformity assessments already performed by their suppliers. If the different components are safe for human health and the environment, then the blend can also be presumed to be safe if the blending did not change the nature of the components. A relevant example is selling struvite as an EU fertilising product (thus meeting the requirements for the CMC 'precipitated phosphate salts & derivates' and the PFC inorganic fertilisers) to a fertiliser blending company to include it into an NPK-fertiliser blend.
ii. As a CE marked product to be used as an intermediate in a **chemical manufacturing process**. The CE marked product will now be used a precursor for a chemical process to make a material pertaining to CMC 1, and the new substance that comes out of this manufacturing process has to comply with the requirements for CMC 1 since an intentional chemical reaction is involved. However, it is not because the intermediate has been derived from waste that the newly produced material is excluded from CMC 1; the intermediate left the waste regime by virtue of complying with all conditions (CMC level, PFC level, labelling requirements, and conformity assessment procedure) to become an EU fertilising product. A relevant example is, for instance, the STRUBIAS material triple superphosphate (TSP) derived from bone meal ash. Triple superphosphate is not only registered pursuant to Regulation (EC) No 1907/2006 as a fertiliser (EC 232-379-5), but can also be used as an intermediate to produce a compound NPK fertiliser, i.e. a fertiliser with each nutrient contained in every granule or prill. In this case, the compound NPK producer can buy the CE marked TSP derived from bone meal ash (complying with CMC requirements for 'thermal oxidation materials & derivates' and PFC 1 C 'inorganic fertilisers') on the internal market, use it as an intermediate, and produce a compound NPK through an intended chemical reaction that then requires a REACH registration according to point 2 of CMC 1.

- **Any waste-derived material of a CMC production process that has not undergone the conformity assessment procedure (for instance, sewage sludge ashes or struvites that have not yet passed through such a procedure) will still be considered a 'waste' and must comply with the regulatory provisions for waste** (Waste Framework Directive (2008/98/EC) and Directive (EU) 2018/851, Waste Shipment Regulation (96/61/EC), etc.; unless it has achieved End-of-Waste status at Member State level). Hence, fertiliser manufacturers that receive such waste materials and turn them into EU fertilising products will be required to fulfil all obligations laid down in the legislation on waste. A relevant example is a fertiliser production plant manufacturing single superphosphate from precipitated calcium phosphates at a wastewater treatment plant. **As long as the precipitated phosphate salt has not undergone the conformity assessment procedure laid down in the EU Fertilising Products Regulation ((EU) 2019/1009), the material can be shipped under the waste legislation** to a fertiliser production plant, which will then be able to affix the CE marking following a conformity assessment procedure.

4.2.2 Status of industrial by-products and polymers

Some STRUBIAS production processes make use of specific polymers and industrial by-products as defined in Article 5 of Directive 2008/98/EC on waste. Relevant examples include:

- the solid-liquid separation of manure fractions using anionic polyacrylamide to separate a manure fraction with a higher dry matter content that can be used as an input material for a pyrolysis process;
- the use of by-product from magnesite mining and the MgO production industry to be used as a Mg source in the struvite precipitation process;
- the use of sulphuric acid (an industrial by-product from oil refining) and spent acid (e.g. a by-product from the food industry) to manufacture single superphosphate from bone meal ashes;
- the use of calcite, a by-product from the dissolution of dolomitic aggregates, as an additive for pyrolysis reactions.

Whereas industrial by-products can positively contribute to circular practices in the EU Fertilising Products Regulation ((EU) 2019/1009), criteria are required to ensure a high level of protection of the environment and human health from the use of such materials. Depending on their origin, some industrial by-products can reasonably be expected to contain particular organic and inorganic contaminants that are uncommon in the targeted input materials for the STRUBIAS CMCs. The same goes for non-biodegradable polymers that may have, for instance, a negative effect on soil organisms. Hence, a tangible risk exists that organic or inorganic contaminants present in industrial by-products and polymers could directly be transferred to the EU fertiliser that contains STRUBIAS materials.

The **JRC proposal is to transpose provisions on the possible use of by-products and polymers within the framework of the EU Fertilising Products Regulation ((EU) 2019/1009) to STRUBIAS materials** to guarantee a level playing field between STRUBIAS CMCs and other CMCs, as follows:

- by-products can be used in STRUBIAS manufacturing processes when these are consumed in or used for chemical processing, and as long as the end material of the manufacturing process is REACH-registered;
- the use of biodegradable polymers is supported.

4.2.3 The principle of optional harmonisation

In view of the very local nature of certain product markets, the EU Fertilising Products Regulation ((EU) 2019/1009) maintains the possibility that **non-harmonised fertilisers can be made available on the market** in accordance with national law, and the principles of mutual recognition of the European Union. This implies that the EU Fertilising Products Regulation ((EU) 2019/1009) will, therefore, **not restrict materials already in use in specific Member States that are covered under national legislation**.

5 STRUBIAS recovery rules

5.1 Terminology and scope of STRUBIAS component material categories

The acronym STRUBIAS was initially chosen as the working title for this project because the initial scope of the project focused on STRUvite, BIochar and ASh-based materials. As a result of the continuous refinement of the scope and a further expansion of the materials covered, a modified terminology for the possible component material categories (CMCs) covered by the STRUBIAS project is proposed. **The new CMC names are based on the principles that they (1) should make a clear reference to the production process applied to obtain the CMC material, and (2) they should clearly reflect any possible materials covered by the CMC.**

5.1.1 'Precipitated phosphate salts & derivates' instead of 'struvite'

Struvite is a phosphate mineral that can be precipitated from a liquid solution or slurry and its name has been used as the general working title for a group of possible recovered phosphate salts since the beginning of the STRUBIAS project. The precipitation process involves the **formation of a separable solid substance from a solution by converting free ions into an insoluble form through the addition of chemicals**. It is noted that mineral struvite (magnesium ammonium phosphate - $NH_4MgPO4\cdot 6H_2O$) can also be chemically synthesised from virgin chemicals, in which case it is already covered by the provisions of *Component Material Category (CMC) I - Virgin material substances and mixtures.* This production pathway for struvite is therefore excluded from the present analysis, and consequently from the newly proposed CMC.

Today struvite is the precipitated phosphate salt which is most widely commercially recovered for nutrient recycling as it is possible to isolate relatively pure minerals with a high P-content with only trace amounts of impurities, and it has a demonstrated value as a P-fertiliser. It is often assumed that precipitates harvested at a pH range between 9.0 and 10.7 are struvite-like compounds under appropriate molar ratios of magnesium, nitrogen and phosphate. On some occasions, X-ray diffraction (XRD) is used to characterise the harvested crystalline precipitates, mainly by comparing the position and intensity of peaks with the struvite reference (Hao et al., 2008). If the diffraction patterns match the struvite reference to a certain extent, precipitates are then 'confirmed' as being struvite. However, because XRD is not a quantitative method and amorphous precipitates are easily overlooked, **the harvested precipitate may be erroneously interpreted as a relatively pure struvite when, in fact, it is not** (Hao et al., 2008). The apparently fragile equilibrium of struvite in solution leads to the presence of other crystal phases as well (Andrade and Schuiling, 2001; Bhuiyan et al., 2008). The formation of other magnesium phosphates such as $MgNH_4PO_4\cdot H_2O$ (dittmarite), $MgHPO_4\cdot 3H_2O$ (newberyite), $MgKPO_4\cdot 6H_2O$ (K-struvite) and a wide variety of calcium phosphates (e.g. $CaNH_4PO_4.7H_20$ (calcium ammonium phosphate), amorphous calcium phosphates, brushite ($CaHPO_4\cdot 2H_2O$)) through precipitation or dissolution processes has been reported in the literature (Michalowski and Pietrzyk, 2006; Massey et al., 2009). Hence, it becomes clear that ‘struvite-like’ materials are in many cases not pure crystals, defined as solid particles in which the constituent molecules, atoms or ions are arranged in a fixed and rigid repeating three-dimensional pattern or lattice. In line with this observation, the production process can best be referred to using the more broad-ranging term ‘**precipitation**’ instead of ‘crystallisation’. The term precipitated phosphate salts encompasses the broad variety of materials that are targeted in recovery or production processes that are intended to be used as fertilising materials or as intermediates in manufacturing processes of P-fertilisers.

Currently, struvite is the most common precipitated phosphate salt for most industrial facilities in planned, pilot and operational facilities in Europe (see Section 7.1.1). Nonetheless, it is worth noting that, besides **struvite, other calcium phosphates and magnesium phosphates are also registered pursuant to Regulation EC No 1907/2006 (REACH) as fertilisers**. A total of 252 different phosphate salts are registered in the REACH registration system, though only some of them are registered as fertilisers.

Table 1: Examples of phosphate salts as registered pursuant to Regulation EC No 1907/2006 (REACH) as fertilisers

EC / List number	regulatory REACH process names	alternative IUPAC names (selected)	CAS number	molecular formula
232-075-2	ammonium magnesium orthophosphate	struvite	7785-21-9	NH4MgPO4·6H2O (hydrate)
231-826-1	calcium hydrogenorthophosphate	dicalcium phosphate, calcium dihydrogen phosphate	7757-93-9	CaHPO4 (anhydrous); CaHPO4·2H2O (dihydrate)
231-823-5	magnesium hydrogenorthophosphate	dimagnesium phosphate	7757-86-0	MgHPO4
235-330-6	pentacalcium hydroxide tris(orthophosphate)	hydroxylapatite, bone ash	12167-74-7	Ca5(PO4)3(OH)
231-837-1	calcium bis(dihydrogenorthophosphate)	monocalcium phosphate	7758-23-8	Ca(H2PO4)2 (anhydrous); Ca(H2PO4)2.H2O (hydrate)
236-004-6	magnesium bis(dihydrogenorthophosphate)	-	13092-66-5	Mg(H2PO4)2 (anhydrous); Mg(H2PO4)2.4H2O (quadhydrate)
231-840-8	tricalcium bis(orthophosphate)	tricalcium diphosphate, tricalcium phosphate	7758-87-4	Ca3(PO4)2
231-824-0	trimagnesium bis(orthophosphate)	trimagnesium diphosphate, trimagnesium phosphate, tribasic magnesium phosphate	7757-87-1	Mg3(PO4)2

Some P-recovery processes such as the Budenheim process, P-ROC process, and BioEcoSim deliberately target the formation of calcium phosphates, rather than struvite. Based on the information received from the STRUBIAS subgroup, there appears to be an interest in including these types of recovered P-rich salts under this CMC. More specifically, **techno-scientific information on calcium phosphates** as end-products of P-recovery processes was received from the STRUBIAS subgroup for possible inclusion. The input received enabled an assessment of the agronomic value and the environmental and human health safety aspects.

The newly proposed CMC aims at covering any acceptable form of **phosphate-based compounds** produced through a **precipitation process** that is in line with the principles of **phosphorus recovery in safe P-concentrated materials**. Therefore, the scope of this CMC has been expanded to **all P-rich mineral substances of high purity formed through the**

precipitation of dissolved phosphate ions, and their derivates, that are sufficiently effective at providing P to plants.

Precipitated phosphate salts could be **recovered** from waste-based materials (e.g. municipal wastewaters), where the precipitation process constitutes a valuable alternative to the incineration and landspreading of P-rich waste streams. Precipitated phosphate salts could also be **deliberately produced** from P-rich materials such as processed animal by-products (e.g. co-products of gelatin production from animal bone material). It is preferable that the end materials of the production processes covered are suitable for direct use as a fertiliser on agricultural land as well as for use as a source or intermediate in existing production processes for P-fertilisers and fertiliser blends.

Based on scientific literature and feedback from the STRUBIAS subgroup, the understanding of experts and users is that it is unnecessary to orient P-recovery through precipitation processes exclusively to struvites of high purity for the following reasons:

- The production of P-minerals with a high struvite content is a **technically challenging and costly process**, especially if calcium- or potassium-rich input materials are also considered (Hao et al., 2008).
- **Struvite is not necessarily superior to some other phosphate-based compounds in agronomic efficiency** (Johnston and Richards, 2003; Massey et al., 2009; Hao et al., 2013; ESPP, 2016; see also Section 6.2.2), nor does struvite of high purity have a superior fertiliser value than other, less pure compounds.
- Although there is a relationship between struvite purity and struvite contaminant levels, **precipitated phosphate salts with a low organic C content generally show low levels of contamination,** which do not pose unacceptable risks for the environment and human health (see Section 5.3.5).
- The **mineral fertiliser industry** has no strict preference for particular phosphate salts that will be used as raw materials. Wet chemical and thermochemical processes, either specifically processing phosphate salts or within existing fertiliser industry processes, can transform phosphate precipitates into water-soluble P-fertilisers (Hao et al., 2013). Rather than chemical composition, **the content of P** (preferably 30-40%, expressed as P_2O_5, similar to phosphate rock) **and organic C** (preferably as low as possible) are major factors that determine the suitability of precipitated phosphate salts for use as an intermediate raw material for the mineral fertiliser industry.

The P-recovery process could also be **a two-step process**, where phosphates are recovered in an intermediate form (e.g. hydroxyapatite or iron-phosphate complexes; P forms that typically show a reduced plant P availability), after which the second phase of the STRUBIAS production process targets the mobilisation of the precipitated P into a form that is readily available for plant P uptake (Wilfert et al., 2015). Hence, a manufacturer could decide to target the production process towards a **<u>derivate</u> from the precipitated phosphate salt.**

In conclusion, it is proposed to modify the name of this CMC to '**precipitated phosphate salts & derivates**' instead of struvite.

5.1.2 'Thermal oxidation materials & derivates' instead of 'ash-based materials'

The original project scope was limited to 'ash-based products from mono or co-incineration and combustion of biomass or resulting from industrial processes'. It is proposed to delineate the scope of the materials covered under this CMC, and more specifically to make a clear distinction between the CMCs covered by the STRUBIAS materials 'ash-based materials' and 'biochar'. Both types of materials result from the **thermo-chemical conversion processes, and no clear cut-off criteria have been defined to divide the end materials into hydrochar, biochar, or ash** (Moller, 2016) (Figure 1). The classification of the different materials produced through thermochemical conversion is not always straightforward, and at times, a combination of materials is collected. Moreover, 'ash' can have very different properties (e.g. organic carbon content depending on the degree of combustion) and be formed through different pathways that have a distinct ability to remove organic contaminants (Figure 1). Some biochars contain, for instance, a significant ash fraction. Finally, the STRUBIAS subgroup indicated that ash-based materials might not be all-encompassing as nutrient recovery can also take place in the form of '**converter slags**', a material formed through a melting process in an oxygen-rich environment (Figure 1). Therefore, focusing on the end materials of the production process might not be the best approach to delimit the scope of the CMC.

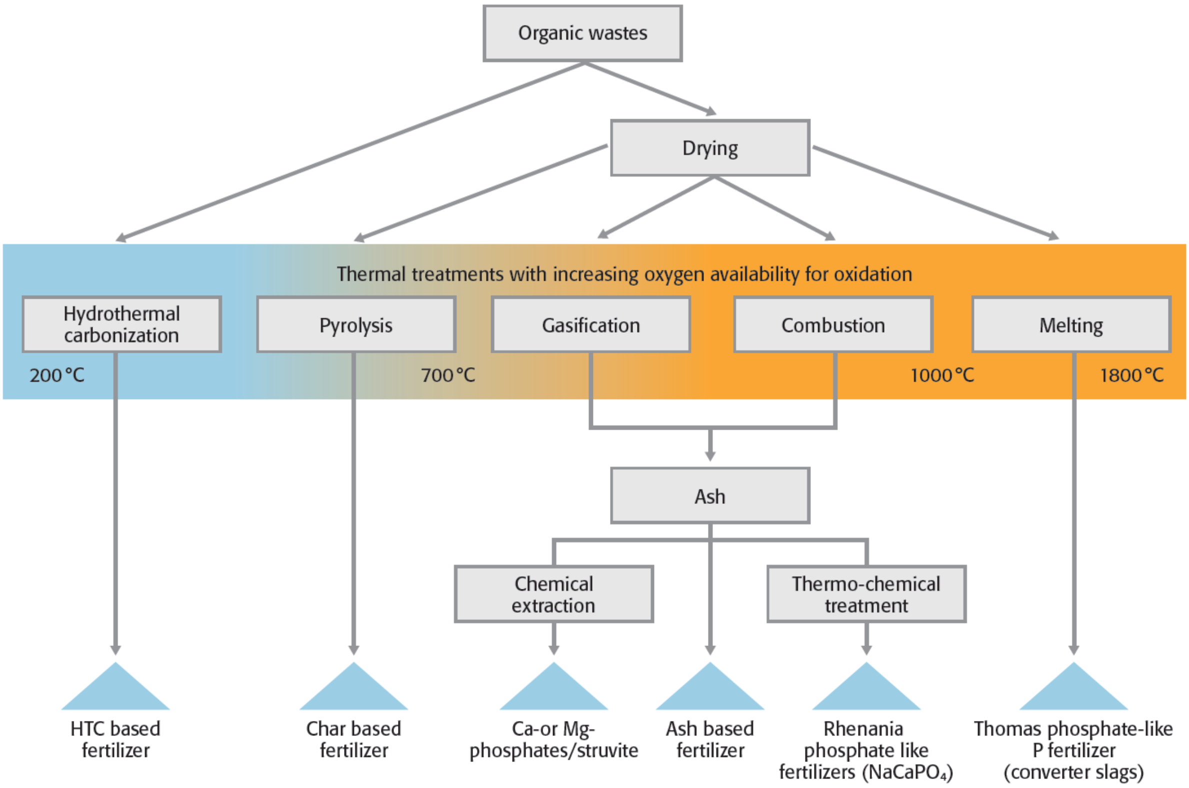

Figure 1: Schematic overview of thermo-chemical processes that can be applied for nutrient and organic matter recovery processes covered by the STRUBIAS project (adopted from Möller, 2016, with permission from the author).

A way forward for the comprehensive inclusion of the different thermochemical materials is to divide material categories based on the **oxygen profile applied in the thermochemical conversion process**, thus on the production conditions for the different materials. This is a straightforward approach as it enables the separation of materials formed under oxygen-limiting conditions from materials under non-oxygen-limiting combustion conditions. Complete oxidation in the combustion process would convert this carbon entirely to CO_2. However, in technical plants a total conversion will never be accomplished and a small amount of products of incomplete combustion are found in all residue streams (Vehlow et al., 2006). Nonetheless, combustion under a non-oxygen-limiting environment produces metals and slags with a low organic C content, typically lower than 3%. As organic carbon and soot particles show a high adsorption potential for contaminants (Mätzing et al., 2001), the organic C content of the material collected from the thermochemical conversion plant is a highly relevant parameter from an environmental and human health perspective. Combustion under non-oxygen-limiting conditions to low levels of organic C in the ashes is a well-demonstrated technique for the effective removal and thermal destruction of a broad range of organic contaminants that are inherently present in the targeted input materials for this CMC. As a matter of fact, **the thermal oxidation of waste materials is a widely applied method for the disposal or recovery of waste as outlined in the Best Available Techniques (BAT) Reference Document on waste incineration pursuant to Article 13 of Directive 2010/75/EU on industrial emissions (European Commission, 2017b).** Basically, waste incineration is here referred to as the *oxidation* of the combustible materials contained in the waste. Waste is generally a highly heterogeneous material, consisting essentially of organic substances, minerals, metals and water. During incineration, flue-gases are created that will contain the majority of the available fuel energy as heat. The organic substances in the waste will burn when they have reached the necessary ignition temperature and come into contact with oxygen (European Commission, 2017b). The actual combustion process takes place in the gas phase in fractions of seconds and simultaneously releases energy. The objective of waste incineration is to treat waste so as to reduce its volume and hazardousness, whilst capturing in solid form (and thus concentrating, for instance metals) or destroying potentially harmful substances (European Commission, 2017b). Combustion processes can also provide a means to enable recovery of the energy, mineral and/or chemical content of the waste. Also, in Regulation (EC) No 1069/2009 on animal by-products, the combustion in a non-oxygen-limiting environment is referred to as an effective measure for the hygienisation of animal by-products. Hence, it is proposed to divide the end materials from thermo-chemical conversion techniques based on the oxygen availability in the production process. **It is proposed to refer in the recovery rules for this CMC to a 'thermal oxidation process in a plant under non-oxygen-limiting conditions'. This definition provides following advantages for this CMC:**

- **a demonstrated efficiency of thermal oxidation under non-oxygen-limiting conditions to destroy a broad range of organic contaminants;**
- **the explicit reference to this technique in the existing legislative EU frameworks for the treatment of wastes and animal by-products as a method for waste hygienisation;**
- **a scope expansion to the inclusion of converter slags that are also produced as a result of a thermal oxidation process under non-oxygen-limiting conditions, more specifically basic oxygen furnace and electric arc furnace slags.**

Some thermally oxidised materials that are collected from the combustion plant can **be used directly as a fertilising material on land** (e.g. poultry litter ashes; Ehlert and Nelemans, 2015b). Materials that have undergone **a thermal oxidation process can also be subjected to further manufacturing steps** resulting in fertilising materials of a different chemical composition than the oxidised material. A relevant example is the production of triple superphosphate from the reaction of sewage sludge ashes with sulphuric acid or hydrochloric acid (see Section 5.4.3.3). This may be required for the removal of inorganic contaminants (e.g. metals/metalloids) and/or the improvement of the availability of nutrients contained in the ashes and slags. As outlined in Section 4.2.1, **the recovery rules should hence include provisions to enable a CMC status for both materials that have undergone a thermal oxidation as well as for 'thermal oxidation material derivates'.**

Altogether, it is therefore proposed to change the name of this possible CMC from 'ash-based materials' to 'thermal oxidation materials & derivates'.

5.1.3 ‘Pyrolysis & gasification materials’ instead of ‘biochar’

This material group was referred to at the beginning of the STRUBIAS project using the working title ‘**biochar**’. The European Biochar Certificate applies the following definition for biochar (EBC, 2012):

> ‘*Biochar is a heterogeneous substance rich in aromatic carbon and minerals. It is produced by pyrolysis of sustainably obtained biomass under controlled conditions with clean technology and is used for any purpose that does not involve its rapid mineralisation to* CO_2 *and may eventually become a soil amendment*’.

In general, the **organic carbon content of pyrolysed chars fluctuates between 5% and 95%** of the dry mass, depending amongst others on the feedstock and process temperature used. For instance, the C content of pyrolysed beech wood is around 85% while that of poultry manure is around 25% (EBC, 2012) and that of bone is less than 10% (3R AgroCarbon, 2016). Therefore, the European Biochar Certificate refers to pyrolysed organic matter with a C content lower than 50% as **pyrogenic carbonaceous materials**, instead of biochar. From the information received from the STRUBIAS subgroup, it is clear that there is a considerable interest in using both **C-rich** (e.g. woody biomass) and **mineral-rich** (e.g. animal bone material, different types of manure) feedstocks as **input materials** for pyrolysis/gasification processes.

Moreover, the STRUBIAS subgroup also highlighted the possibility of including **hydrochar**, the material resulting from the wet pyrolysis or hydrothermal carbonisation of plant-based or animal-based input materials, under this CMC. This proposal has been investigated and it has been concluded that such materials could possibly be covered under the scope of this CMC, as long as the end material meets the quality requirements to ensure environmental protection while providing plants with nutrients.

A single CMC should, however, cover both end material types which is why the name ‘**pyrolysis & gasification materials’** is proposed for any materials that are produced via production processes that cover the pyrolysis technology spectrum, including pyrolysis, gasification and wet pyrolysis techniques. This terminology offers the advantage that a clear reference is made to the production technology in the name of the CMC, similar to the other STRUBIAS CMCs. The materials covered under this CMC can be of a very different nature as the production process conditions may span from partial combustion to a fully reductive environment, resulting in materials with a low and high organic C content, respectively. Hence, this CMC name groups materials that have a similar potential hazardousness due to the incomplete oxidation of the feedstock materials, and does not refer to the nature of the resulting end material of the production process (similar to the other STRUBIAS CMCs).

5.2 Cross-cutting items and other major issues

5.2.1 Scope widening and a principal focus on ‘safety’ criteria

The STRUBIAS subgroup members highlighted the need to broaden the scope of the STRUBIAS CMC, expanding the eligible input materials and reduce limitations on process conditions, with the major objective to increase the recycling of nutrients, especially P.

These concerns have been addressed to the best possible extent in this document, as exemplified by following cases:

- Setting **process and material requirements to ensure that value-added materials that can contribute to nutrient recycling in a circular economy can enter the internal fertilising market**. As outlined in Section 5.1, the scope for each of the STRUBIAS CMC has been significantly expanded by incorporation, for instance, of phosphate salts other than struvite, and pyrolysis & gasification materials as produced through hydrothermal carbonisation processes.
- A detailed analysis has been performed, tracing P through the food and non-food chain. **This evaluation enabled the selection to be focused on input materials that contain P that can be recycled, where technically and economically possible**. For ‘thermal oxidation materials & derivates’, almost all non-hazardous waste materials can, for instance, be used as input materials. At the same time, other eligible input materials that show potential for the recycling of other nutrients and organic matter have been added to the input material lists of ‘thermal oxidation materials & derivates’ and ‘pyrolysis & gasification materials’, mostly in line with the proposals of the STRUBIAS subgroup (but see Section 5.2.8).
- The proposals for the recovery rules have been developed considering the principle of a **neutral stance towards existing and future technological developments**. The production process conditions have been designed with a large focus on the quality of the end materials, rather than on the production process conditions. A relevant example is the use of the H/C_{org} parameter for pyrolysis & gasification materials as a parameter for C stability instead of imposing strict time-temperature profiles.

At the same time, it should be taken into consideration that under the current proposals STRUBIAS CMCs could be derived from waste-based materials, which is why a different approach is required than for the CMCs currently covered in the EU Fertilising Products Regulation ((EU) 2019/1009). It is often the case that waste materials are associated with a gate fee; the receiver of such wastes *receives* a financial retribution for acceptance of possible STRUBIAS input materials. In contrast, a manufacturer of fertilising materials that contain other CMCs *spends* financial resources to obtain or extract possible input materials. This is a very different context. The fundamental principles of environmentally sound waste management imply that provisions are required to **avoid materials that do not meet the criteria outlined above being used in EU fertilising products**. This is especially important as the **CE mark** is **associated with the free movement of goods** with only minimal legislative and administrative procedures associated. It should be ensured that no back doors are being incorporated into the EU Fertiliser Regulation to circumvent the waste status of certain materials. A **relevant example** is the need for a criterion to restrict specific input materials (e.g. fossil fuels processed in plants other than incineration and biomass combustion plants) in thermal oxidation materials & derivates.

Moreover, the recovery rules are **constrained by the existing EU legal framework on associated matters**, such as the Animal By-Products Regulation ((EC) No 1069/2009), the Waste Framework Directive ((EC) 2008/98/EC) and Directive (EU) 2018/851, or the REACH Regulation ((EC) No 1907/2006). **Relevant examples** are the exclusion of animal by-products of category 1 for STRUBIAS production processes (see Section 5.2.6), or the compliance with the conditions for the incineration of waste as laid down in the Industrial Emissions Directive (2010/75/EU).

5.2.2 Plant nutrient availability of STRUBIAS materials

A significant share of the STRUBIAS materials show a **high nutrient content** and so might be used as ingredients for PFC 1 - Fertilisers. The **return of secondary nutrient resources to agricultural land is, however, not the same as efficiently recycling nutrients**.

If used as macronutrient fertilisers, STRUBIAS materials will be mostly considered as P-fertilisers because the P content in many STRUBIAS materials is often more aligned to plant nutrient demand than the N and K content. With the possible exception of N in pyrolysis & gasification materials, the N and K compounds in STRUBIAS materials are also generally readily available to plants and may thus contribute to plant N and K nutrition; some materials are even effective K-fertilisers (e.g. poultry litter ashes; Ehlert and Nelemans, 2015a). See Sections 5.3.4, 5.4.4.1 and 5.5.4.2 for a more detailed discussion on the plant availability of N and K in the STRUBIAS material groups.

Phosphorus, however, may be unavailable to crops when strongly bound to certain bi- and trivalent ions. A lack of consideration for the plant availability of recycled P-sources may (i) lead to the long-term accumulation of this critical nutrient in soils, which removes it from the global biogeochemical cycles, and (ii) reduce farmers' confidence and create low market acceptance for innovative P-fertilisers derived from secondary raw materials. Based on these concerns, it may be advisable to control for the **plant availability of P in P-fertilisers.**

The nutrient value of fertilising products can be determined using either **bioassay tests or chemical methods** (Camps-Arbestain et al., 2017). The bioassay tests are based on the plant response to an amendment under controlled greenhouse conditions or in field trials. Chemical methods are based on specific chemical solutions - known as extractants (e.g. water, neutral ammonium citrate - NAC, formic acid or an aqueous solution of 2% citric acid) - that are used to extract P fractions from the fertiliser. Bioassay procedures are the most reliable for predicting nutrient availability, but these methods are more time-intensive, costly and less reproducible than chemical methods. The STRUBIAS subgroup indicated the limitations of bioassay tests, from a practical and scientific point of view. In contrast, chemical methods are simpler to enforce as a criterion for plant P availability, and therefore constitute the commonly used measures in legal frameworks.

It is, however, **challenging to determine a single cut-off value that clearly and universally distinguishes between 'effective' and 'ineffective' fertilisers** because of the following factors that impact upon the nutrient availability and release dynamics of nutrients present in fertilisers:

- **The lack of extensive datasets that link results of chemical analysis to plant yield responses in laboratory and field settings**. The need for agronomic trial work is very urgent and for each of the multiple extraction procedures further assessments are required before these can be completely validated.
- **The different soil types and weather conditions** and the varying nutrient use and uptake strategies of plant species and the temporal variation in nutrient demands for different plant species. **In an agricultural sector that is increasingly evolving towards diversification, the use of targeted applications and precision-management, fertilisers derived from secondary raw materials that are only effective under specific settings (e.g. acid soil types) can effectively contribute to closing European nutrient cycles.**

Based on the feedback on the questionnaire included in the STRUBIAS Interim Report, it was observed that the **techno-scientific literature that correlates plant P-availability, plant P-responses to fertilisation and fertiliser P-solubility for STRUBIAS materials in different extractants is very limited and scattered.** No single extractant or combination of extractants seems to allow a clear distinction between 'effective' and 'ineffective' P-fertilisers. In this respect, it is relevant to note that P-fertilisers developed from secondary raw materials often have a different chemical composition to those that are currently on the market, so it remains questionable whether the relationship between extractable P and plant P availability holds. The data from Wilken et al. (2015) indicate, for instance, that specific P-fertilisers with low NAC extractable P content (6%, relative to total P) can be equally effective as phosphate-rock-derived mineral P-fertilisers. Hence, setting exclusive criteria based on the available evidence **includes a tangible risk for excluding a market entrance for innovative P-fertilisers developed from secondary raw materials,** and thus undermining possibilities for innovative products that have been developed in line with the circular economy principles. Moreover, **some P-fertilisers that have high acid-extractable P-availability actually show a low plant-available P content** (Duboc et al., 2017). Therefore, the incorporation of a minimum extractable P-content may not be of use to enforce the plant availability of P in the fertiliser. As a matter of fact, it may even be unfavourable as it may **misinform and mislead end users on the agronomic efficiency of the STRUBIAS material, and create false quality expectations for low-quality P-fertilisers.**

Several **STRUBIAS materials** fulfil specific roles to promote plant growth and increase agricultural yields, and often **stretch beyond the purpose of solely providing P to plants**.

Examples include P-rich pyrolysis & gasification materials that could address the need for the recycling of organic matter in agroecosystems, or ashes that contribute to micronutrient plant nutrition. From this point of view, **it is a more straightforward option to enforce plant P-availability for specific CE products at PFC level**. The omission of requirements on plant P availability for STRUBIAS materials would be in line with the criteria for the other CMCs. Setting supplementary requirements for STRUBIAS CMCs may, in contrast, involve **a duplication of criteria at CMC and PFC level**, which is considered undesired by most members of the STRUBIAS subgroup.

For all these reasons, it is **proposed to set no criteria to regulate plant P availability for STRUBIAS materials if the nutrient value of fertilising products is regulated at PFC level** through product quality or labelling requirements, at least for the current category of inorganic P-fertilisers. In the event that no criteria on extractable P-content for inorganic P-fertilisers are included at PFC level in the EU Fertilising Products Regulation ((EU) 2019/1009), one of following measures may be suitable to STRUBIAS materials at PFC level:

- A **labelling requirement** for STRUBIAS materials that exceed a certain P-content threshold (e.g. 3% P_2O_5). It is proposed to label the ratio of water-extractable and acid-extractable P content (e.g. NAC, formic acid, citric acid) relative to total P content.
- **Lenient P-solubility requirements for STRUBIAS materials that exceed a certain P-content threshold** (e.g. 3% P_2O_5):
  - water solubility: minimum level of 25% of total P, **or**
  - solubility in neutral ammonium citrate: minimum level of 30% of total P, **or**
  - solubility in formic acid: minimum level of 35% of total P.

Hence, lenient requirements are proposed as it is assumed that farmers enjoying a competitive fertiliser market will select the best products in line with their specific settings and needs. Although divergent views on this topic were recorded within the STRUBIAS subgroup, the JRC believes that **market competition will serve as an effective mechanism as long as the STRUBIAS technical requirements impede the access of low-quality fertilising materials to the internal market.** By setting requirements on input materials for STRUBIAS processes and quality requirements on output materials, the STRUBIAS recovery rules as provided in Section 5.3 to 5.5 should effectively impede the entry of low-quality P-fertilisers to the internal market. The internal manufacturing costs for STRUBIAS P-fertilisers meeting the proposed technical criteria is mostly more expensive than the waste management cost for their source materials, and even more expensive than for phosphate-rock-derived P-fertilisers. Hence, in view of receiving revenues from the sales of STRUBIAS materials, it is in the own benefit of the manufacturer to produce a high-quality material with a high plant P-availability. Low-cost materials (e.g. specific by-products) will be unable to achieve the CE status through compliance with the STRUBIAS technical requirements.

5.2.3 Avoided duplication of criteria at CMC and PFC level

The fundamentals underlying the two sets of requirements (i.e. PFC- and CMC-level requirements) are given by the fact that **different component materials warrant different process requirements and control mechanisms adapted to their different potential hazardousness and variability**. Component materials for EU fertilising products should therefore be divided into different categories.

In the STRUBIAS Interim Report, there was **a focus on minimum nutrient requirements as a proxy for material purity**. This approach was proposed to ensure that STRUBIAS material quality and chemical composition are in line with the intended functions of the fertilising materials. The JRC has addressed this concern of the STRUBIAS subgroup by shifting the approach for criteria development away from minimum nutrient contents **towards the direct limitation of elements that are unwanted and could even be unsafe in STRUBIAS materials and the exclusion of certain input materials** (e.g. fossil fuels used at fossil fuel power plants for thermal oxidation materials & derivates). These criteria are not expected to restrict the market for the targeted STRUBIAS materials. Nonetheless, a minimum P content for precipitated phosphate salts has been maintained in the final set of draft technical proposals as it is the most straightforward manner to delimit the scope of this CMC, and avoids compliance schemes that are more costly and complex (e.g. including XRD measurements to confirm the presence of *phosphate* minerals in the CMC precipitated *phosphate* salts & derivates).

Limit values for specific **metals and metalloids** have been retained in the criteria at CMC level, more specifically for thermal oxidation materials & derivates. The metals and metalloids regulated at PFC level are restricted to a specific set of elements, more specifically As, Cd, hexavalent Cr, Hg, Ni, Pb, Cu and Zn. Nonetheless, the production processes of STRUBIAS materials are specific in the sense that they can concentrate non-volatile metals and metalloids in the solid residues during thermal conversion and due to the inclusion of waste-based materials and industrial by-products as eligible input materials. These observations imply that a **potential risk for the environment and human health exists due to the presence of some less common metals and metalloids in STRUBIAS materials.** Therefore, a detailed assessment is required as described in Sections 5.4.5.1 and 5.5.5.1 for thermal oxidation materials & derivates and pyrolysis & gasification materials, respectively. For the final set of proposals for the STRUBIAS recovery rules, the techno-scientific comments from the STRUBIAS subgroup on the methodology applied has been reviewed. Relevant and validated techno-scientific information (e.g. on solid-liquid partition coefficients for the different metals and metalloids, application rates, percolation fluxes) has been taken into account during the revised assessment, and the results obtained have been further corroborated against environmental and human toxicity studies. This has resulted in changes in the metals and metalloids included in the compliance schemes for the CMCs thermal oxidation materials & derivates and pyrolysis & gasification materials as well as their proposed limit values.

5.2.4 ‘Two-step’ fertiliser production processes

As outlined in Section 4.2.1, the structure of the EU Fertilising Products Regulation ((EU) 2019/1009) does not enable intentional chemical reactions or transformations to take place between the different CMCs that are contained in EU fertilisers. This implies that any chemical or thermochemical processing steps required to ensure that the materials under consideration **no longer pose a risk to human health or the environment** need to **occur before assessing the possible CMC status.** For instance, unprocessed sewage sludge incineration ash from default mono-incineration not designed for P recovery **cannot, by any means, be considered a CMC** because further processing would be required before such materials could be placed on the market as a CE-marked product. The recovery rules shall, therefore, encompass any manufacturing steps required to produce a material that can be used as such as a fertilising material, i.e. directly without any further processing other than normal industrial practice. Therefore, **the recovery rules shall describe process and product requirements until the end stage in the manufacturing process of a fertilising material is reached**. Concretely, this implies, for instance, that recovery rules shall include provisions for post-combustion manufacturing steps on unprocessed mono-incineration sewage sludge ashes, and that product requirements apply to a triple superphosphate material that is (partially or entire) produced from incinerated sewage sludge.

The STRUBIAS subgroup highlighted the need to better explain the possibility of **‘two-step’ manufacturing processes** to enable further manufacturing processes for intermediate materials (e.g. precipitated phosphate salts, ashes). The principle of two-step manufacturing processes is that a first manufacturing step can be applied to reduce concentrations of specific contaminants to acceptable levels (e.g. incineration or precipitation to produce materials with a low level of organic contaminants). In a second step, the intermediate material obtained can then be further processed using an extensive set of substances/mixtures to a chemically different material on condition that the risk of re-contamination is controlled. The second manufacturing step can, for instance, improve the plant availability of the nutrient contained in the product, or remove inorganic contaminants from the fertilising material.

The draft proposals for the recovery rules took these comments into consideration, and enabled the manufacturing of ‘**derivates**’ from precipitated phosphate salts and thermal oxidation materials (i.e. ashes and slags). Because the process limitations in the second step of the processes are minimal (e.g. no positive list of chemicals that can be used), a **large degree of freedom** is given to the manufacturers to apply their processes of choice and to promote the development of innovative processes that start from intermediate materials. Examples of such processes are thermochemical manufacturing processes starting from raw ashes, or the chemical transformation of phosphates precipitates – other than Ca and Mg phosphates – into P fertilisers that are already available on the internal market (example of a possible future development as exemplified in Wilfert et al. (2015)). Note that some provisions apply to such two-step manufacturing processes in order to avoid that inert materials are added with the exclusive intention of diluting the contaminants to their established limit values in this and other EU legislation (see Section 5.2.5). This is in line with the requirements for CMC 1 (virgin materials), where no limitations are set on the use of substances and mixtures to produce the CMC material either.

5.2.5 Mixing of ashes

The technology used and the way in which the fertiliser manufacturing installation is designed, built, maintained and operated impacts upon the emissions of contaminants to the soil originating from STRUBIAS materials that will be incorporated in the EU fertilisers. For ashes, especially the metals and metalloids contained in the ashes are of main concern for contamination. In simple terms, some post-combustion manufacturing techniques effectively remove a large share of the contaminants, whereas others remove only a minor share. As some ashes - e.g. sewage sludge ashes - contain metals and metalloids (e.g. Pb, Ni) above the limit values set for inorganic fertilisers at PFC level, production processes that do not remove and discard metals typically reach the limit values through 'dilution'.

The **mixing of waste is common practice in the EU and is recognised as a treatment operation by Annex I and II to the Waste Framework Directive (2008/98/EC)** (see footnotes to operations D13 / R12) (European Commission, 2012). Provision R12 of Annex II refers, for instance, to the exchange of waste for submission to any of the operations numbered R 1 to R 11. This provision 'can include preliminary operations prior to recovery including preprocessing such as, inter alia, dismantling, sorting, crushing, compacting, pelletising, drying, shredding, conditioning, repackaging, separating, blending or mixing prior to submission to any of the operations numbered R1 to R11 [if no other R codes apply]'. For STRUBIAS materials, code R5 is especially relevant as it refers to the 'recycling/reclamation of other inorganic materials'. On the other hand, EU legislation also recognises that individual **waste streams should in principle be kept separate from other wastes and not mixed with the intention of reducing contamination levels and hazardous characteristics in order to meet limit values or to bypass legal requirements**. This holds **especially true for hazardous waste** streams as Article 18(1) of the Waste Framework Directive states that 'it shall be ensured that hazardous waste is not mixed, either with other categories of hazardous waste or with other waste, substances or materials. Mixing shall include the dilution of hazardous substances'. Although Article 18(2) of the same Directive provides for a possible derogation from this rule, the conditions for such a derogation are quite strict, and no best available techniques have been described for fertiliser manufacturing processes.

The STRUBIAS manufacturing technologies applied to process ashes can equally be applied to process phosphate rock into a P-fertiliser. Also, for the primary raw material phosphate rock, the quality of the resulting P-fertiliser is highly dependent on the production process applied, with levels of specific metals (e.g. Cd) in phosphate-rock-derived P-fertilisers varying by more than one order of magnitude depending on the process applied (Ecophos, 2018). Nonetheless, the most commonly applied process to convert phosphate rock, the acidulation process, does not remove metals to a significant extent, and the mixing of input materials of different quality and metal content is permitted for P-fertilisers derived from primary raw materials. Hence, demanding the removal of contaminants for P-fertiliser production processes derived from secondary raw materials may hinder the establishment of a

level playing field for the fertiliser sector. In the developing market for P recovery from secondary raw materials, it remains uncertain to what extent the most advanced techniques that remove metals from the ashes will become reasonably accessible under economically and technically viable conditions.

Sincere manufacturers of STRUBIAS materials will certainly **not apply mixing operations with the sole intention of contaminant dilution**; rather the mixing is a prerequisite to ensure the production of high-quality fertilising materials. The **proposed technical requirements for thermal oxidation materials & derivates contained in the EU fertilising product include provisions to ensure that no long-term increase in the accumulation of metals and metalloids in soils occurs to levels of concern.** Hence, despite a higher metal return to land for processes that apply a mixing *relative* to processes that remove all contaminants from the ashes, **human health and environmental protection is *de facto* implied through the maximum limits for metals in the EU Fertilising Products Regulation ((EU) 2019/1009).** The contaminants in the ashes originate largely from the uptake and recycling of metals in the agro-food system, and do not contribute to a further enrichment of the contaminants in the environment. Hence, recovery processes – even those relying on the principle of dilution – will not result in gross imports of new contaminants into the environment, in contrast to imports from phosphate-rock-derived P-fertilisers that are associated with an inflow of metals from outside Europe. Nonetheless, the landfilling of ashes or their accretion in construction materials removes the metals from the biogeochemical cycles, at least assuming that these routes are an everlasting sink.

Relative to business-as-usual practices for the handling of sludges in Europe, the route of producing P-fertilisers through the ash mixing process generally provides benefits through a reduced exploitation of the primary raw material phosphate rock, but the impacts on eutrophication, human health and global warming potential depend on the counterfactual use and handling scenario of the input material (i.e. landfilling versus landspreading of the sewage sludge; see Section 8.8). This example indicates that **the impacts of the mixing of the ashes relative to current handling scenarios for biogenic wastes on environmental and human health aspects are not clear-cut and are situation-dependent**.

In view of meeting the legislative requirements for waste materials, and more specifically to comply with Article 10 [‘the necessary measures shall be undertaken to ensure that waste undergoes recovery operations’], Article 13 [‘protection of human health and the environment’] and Article 18 [‘it shall be ensured that hazardous waste is not mixed, either with other categories of hazardous waste or with other waste, substances or materials. Mixing shall include the dilution of hazardous substances’] of Directive 2008/98/EC, the **following provisions** are proposed with respect to the handling and mixing of waste materials:

- The **simple physical mixing of input materials classified as waste with other non-hazardous and hazardous wastes, substances or materials – i.e. without the occurrence of an intentional chemical reaction – will not be permitted** as such practices might enable mixing with the sole intention of reducing contaminant values of waste. Therefore, it is proposed to refer in the legal requirements for the treatment of waste to the need to intentionally modify the chemical composition of the material. The use of this wording will prevent inert materials from being added to fertilising materials with the sole intention of reducing contaminant levels. Therefore, the mixing of wastes with wastes or other materials should at all times occur with the intention of improving the quality and/or properties of the resulting material and plant nutrient availability, of removing contaminants, or a combination of both. Operations aimed at lowering the contaminant concentration without lowering the contaminant to bio-available nutrient ratio in the original material are as such restricted.

- Manufacturers that use **hazardous wastes** (e.g. sewage sludge ashes with certain species of Zn; Donatello et al., 2010) within their STRUBIAS production process **should demonstrate the removal or transformation of the respective hazardous substances to levels below the limit values defined in Annex III to Directive 2008/98/EC**. This condition is especially pertinent because of the absence of best available techniques for the manufacturing processes concerned, and will thus create (i) a robust legal framework that provides legal stability for fertiliser manufacturers and (ii) a technologically neutral level playing field for manufacturers across the EU.

5.2.6 Link to Animal By-Products Regulations and alternative processing methods in STRUBIAS production processes

Animal by-products are a possible input material for all STRUBIAS material groups and STRUBIAS production pathways are already observed for precipitated phosphate salts, thermal oxidation materials & derivates, and pyrolysis & gasification materials in Europe. For instance, K-struvites are precipitated from (preprocessed) manure slurries, poultry manure is incinerated, and animal bone material is pyrolysed.

Any input material of animal origin is subject to the scope of the controls of **Regulation (EC) No 1069/2009**. Products derived from animal by-products referred to in Article 32 of Regulation (EC) No 1069/2009 for which the end point in the manufacturing chain has not been determined should not be placed on the market as component materials contained in the CE fertilising product in accordance with the provisions of the EU Fertilising Products Regulation ((EU) 2019/1009). Therefore, any input materials of animal origin should first be subject to envisaged regulatory procedures to define end points under the Animal By-Products Regulation which include among others the EFSA opinion on the risks to public and animal health of the aforementioned component materials (see below).

The possible end point for further use in the EU Fertilising Products Regulation ((EU) 2019/1009) will ultimately be laid down in the Animal By-Products Regulation, **allowing the Annexes to the EU Fertilising Products Regulation (EU) 2019/1009s to refer to animal by-products or derived products for which an end point in the manufacturing chain has been determined in accordance with the third paragraph of Article 5(2) of Regulation (EC) 1069/2009**. This will ensure automatic legal consistency in the event of any modification in the enabled use route for animal by-products as EU fertilisers. This is stated in recital (15) of the EU Fertilising Products Regulation (EU) 2019/1009:

> ‘For each component material category which includes derived products within the meaning of Regulation (EC) No 1069/2009, the end point in the manufacturing chain should be determined in accordance with the procedures laid down in that Regulation. Where such an end point is reached before the EU fertilising product is placed on the market but after the manufacturing process regulated under this Regulation has started, the process requirements of both Regulation (EC) No 1069/2009 and this Regulation should apply cumulatively to EU fertilising products, which means application of the stricter requirement in case both Regulations regulate the same parameter.’

The European Commission can thus lay down further modifications to the permitted use routes and technical requirements for the handling, treatment, transformation, processing and storage of animal by-products or derived products in the Animal By-Products Regulation. This Regulation focuses on biological hazards that may be present in the materials. In addition to these requirements, supplementary requirements can be proposed to ensure environmental and human health protection (e.g. due to the presence of animal drug residues in manures) as well to promote the safe handling and storage of EU fertilising materials derived from animal by-products. At present, Article 32(1) of Regulation (EC) 1069/2009 indicates that organic fertilisers and soil improvers may be placed on the market and used provided:

(a) they are derived from category 2 or category 3 material;
(b) they have been produced in accordance with the conditions for pressure sterilisation or with other conditions to prevent risks arising to public and animal health, in accordance with the requirements laid down pursuant to Article 15 and any measures which have been laid down in accordance with paragraph 3 of this Article;
(c) they come from approved or registered establishments or plants, as applicable; and
(d) in the case of meat and bone meal derived from category 2 material and processed animal proteins intended to be used as or in organic fertilisers and soil improvers, they have been mixed with a component to exclude the subsequent use of the mixture for feeding purposes and marked when required by measures adopted under paragraph 3.

In addition, digestion residues from transformation into biogas or compost may be placed on the market and used as organic fertilisers or soil improvers.

The STRUBIAS report can – based on robust techno-scientific evidence - **propose alternative conditions and technical requirements** for the handling, treatment, transformation, processing and storage of animal by-products or derived products and conditions for treatment of wastewater based on techno-scientific evidence. In the latter case and if considered pertinent at a later stage by the Commission and the legislators, any alternative proposed method shall be assessed in line **with the procedure indicated in Article 20 of Regulation (EC) 1069/2009**. This involves, amongst others, an assessment by the European Food Safety Authority (EFSA) to evaluate risks associated with the food chain. EFSA collects and analyses existing research and data and provides scientific advice to support decision-making by risk managers responsible for making decisions or setting legislation about food safety. Note that these proposals will not form part of the draft proposals for STRUBIAS recovery rules provided in Section 2 because the end-point in the manufacturing chain for animal by-products will be laid down in the Animal By-Products Regulation, and not in the EU Fertilising Products Regulation ((EU) 2019/1009). Instead, the STRUBIAS nutrient recovery rules will refer to animal by-products and derived materials that potentially can be used as input materials as:

a) animal by-products, the products derived from which are referred to in Article 32 of Regulation (EC) No 1069/2009 and have an end-point in the manufacturing chain determined in accordance with the third subparagraph of Article 5(2) of that Regulation;
b) derived products referred to in Article 32 of Regulation (EC) No 1069/2009 for which an end-point in the manufacturing chain has been determined in accordance with the third subparagraph of Article 5(2) of that Regulation.

Alternative methods should comply with the condition that sufficient evidence is available **to indicate that the alternative method provides a degree of protection that is at least equivalent, for the relevant category of animal by-products, to the processing methods that are currently laid down**. In this respect, the following aspects are deliberated and ensuing measures are proposed for the use of animal by-products and derived materials in the STRUBIAS technical proposals:

1) The co-legislator allows the production of fertilising materials from category 2 and 3 animal by-products, but not of category 1 animal by-product material. Therefore, this report has not assessed the risks resulting from the use of unprocessed and processed (e.g. through incineration) category 1 material. Therefore, at present, **Category 1 animal by-product material is not further considered as input material for the present study.**
2) The STRUBIAS subgroup and STRUBIAS interim reports have **indicated that a share of the STRUBIAS production pathways use animal by-products as input materials (i)** that are already processed by the rendering industry in line with the processing methods **(pressure sterilisation) or (ii) are digestion residues from transformation into biogas or compost. Both can, under the current regulatory framework, already be placed on the market as (organic) fertilisers**. Therefore, it is proposed that these processed materials can be **used, without further restrictions to control for biological hazards, as input materials for STRUBIAS production pathways**.
3) There is also an interest from the STRUBIAS subgroup to rely on three different production processes that are currently not permitted as standard use methods in the Animal By-Products Regulation provisions for international trade:
    i. unprocessed manure as input material for precipitated phosphate salts;
    ii. unprocessed manure as input material for pyrolysis & gasification materials;
    iii. unprocessed animal by-products of Categories 2 and 3, including manure, for thermal oxidation materials & derivates.

Therefore, **alternative conditions and technical requirements** for the handling, treatment, transformation, processing and storage of the materials resulting from such materials are proposed as follows:

i. Unprocessed manure as input material for precipitated phosphate salts

Manure, digestive tract content separated from the digestive tract, (raw) milk, milk-based products and colostrum classified as category 2 and category 3 materials pursuant to Regulation (EC) 1069/2009 do not require a specific treatment for hygienisation if the competent authority does not consider them a risk for the spreading of serious transmissible diseases; they can be applied to land in unprocessed form. However, **the placing on the market of processed manure, products derived from processed manure and guano from bats is subject to the requirements laid down in Annex XI (Chapter I, Section 2) to Regulation (EU) 142/2011.** The standard processing method that such materials must undergo includes a heat treatment process of at least 70 °C for at least 60 minutes and they shall have been subjected to reduction in spore-forming bacteria and toxin formation, where they are identified as a relevant hazard. Nonetheless, the competent authority may authorise the use of other standardised process parameters besides those referred to above, **provided that such parameters ensure the minimisation of biological risks**. This involves, amongst others, the identification and analysis of possible hazards, a validation of the intended process by measuring the reduction of viability/infectivity of endogenous indicator organisms, including, for instance, *Enterococcus faecalis*, thermoresistant viruses such as parvovirus, parasites such as eggs of *Ascaris sp.*, *Escherichia coli*, *Enterococcaceae*, and *Salmonella* spp.

Based on the assessment of the biological and other risks outlined in Section 5.3.5.2, it is proposed that manure, non-mineralised guano, and digestive tract content can be used as an eligible input material for production processes without prior processing on condition that precipitated phosphate salt has the following:

- A maximum organic carbon content of 3% of the dry matter content[21].
- No presence of *Clostridium perfringens* in a concentration of more than 100 CFU/g fresh mass and the absence of viable *Ascaris* eggs in a 25 g fresh mass sample of the precipitated phosphate salt.
  By way of derogation, the testing on *C. perfringens* and *Ascaris* sp. shall not be necessary for materials that have that have been subject to either of the following conditions:
    - Pressure sterilisation through heating to a core temperature of more than 133 °C for at least 20 minutes without interruption at an absolute pressure of at least 3 bars. The pressure must be produced by the evacuation of all air in the sterilisation chamber and the replacement of the air by steam (‘saturated steam’).

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[21] As measured using vacuum drying at 40 °C to avoid the loss of crystal-bound water.

    - A processing step in a pasteurisation/hygienisation unit that reaches a temperature of 70 °C during a period of at least 1 hour.

The rationale for proposing the conditions on maximum organic carbon content and supplementary microbial testing **relate predominantly to concerns associated with the release of antimicrobial residues with a high affinity for organic matter (e.g. tetracycline, sulphonamides) and the spreading of antibiotic-resistant bacteria (e.g. spore-forming bacteria and other bacterial species).** The maximal limits for organic carbon, *C. perfringens* and *Ascaris* eggs can be met in the production process, even without supplementary processing steps. However, for materials resulting from manufacturing processes that include operations that ensure the effective elimination of biological pathogens (e.g. anaerobic digestion plus pasteurisation, thermal hydrolysis), exemptions are proposed for microbial testing. We refer to Section 5.3.5.2 for a detailed discussion on the topic.

ii. Unprocessed manure as input material for pyrolysis & gasification materials

Similar to precipitated phosphate salts, there is an interest from the STRUBIAS subgroup to use unprocessed manure as an input material to pyrolysis & gasification processes. Given the legal framework on the use of processed manure outlined above for precipitated phosphate salts, an assessment has been performed of the impact of pyrolysis/gasification processes on biological and other hazards (see Section 5.5.5). It was indicated that biological hazards are typically removed through the application of dry or wet heat under the time-temperature profiles applied in pyrolysis and gasification production processes. Pyrolysis/gasification processes are likely to significantly reduce or even remove the dominant antimicrobial substances that could be present in manures, such as tetracycline. Therefore, **it is proposed that manure, non-mineralised guano, and digestive tract content can be used as an eligible input material for pyrolysis production processes without prior processing**.

As some pyrolysis and gasification processes are in agreement with the hygienisation steps to prevent risks laid down pursuant to Article 15 and any measures which have been laid down in accordance with paragraph 3 of this Article of Regulation (EC) No 1069/2009 (e.g. hydrothermal carbonisation techniques apply a similar or even more stringent pressure sterilisation process than the processing method 1 described in Chapter III of Annex IV to Regulation (EU) 421/2011), it is proposed that pyrolysis & gasification derived from animal by-products of categories 2 and 3 should undergo one of the following treatments at a stage prior to or during the pyrolysis & gasification material production process: (i) pressure sterilisation, (ii) anaerobic digestion, or (iii) Brookes' gasification process as described in point E, Section 2, Chapter IV, of Annex IV to Regulation (EU) No 142/2011.

iii. Unprocessed animal by-products, including manure, for thermal oxidation materials & derivates

The combustion at 850 °C provides an equivalent or superior degree of protection to the processing methods set out in Chapter III of Annex IV for category 2 and 3 animal by-products. Above temperatures of 120 °C, minimal thermal death times are required to inactivate biological pathogens that could be present in category 2 and 3 animal by-products, even under dry conditions. Moreover, the combustion at > 850 °C provides an effective manner to remove organic chemical pollutants that could be present in the animal by-products. **Therefore, it is proposed to place animal by-products of categories 2 and 3 on the positive list of eligible input materials and to adhere to the operating conditions as laid down in Article 50 of the Industrial Emissions Directive for the processing of these materials, without the need for an additional hygienisation step prior to combustion.** The combustion conditions are in line with Regulation (EU) No 592/2014 that amends Regulation (EU) 142/2011, indicating the processing method for poultry litter through combustion in on-farm combustion plants. It is proposed that (1) these combustion conditions could apply as a processing method for all types of animal by-products of categories 2 and 3, and (2) to all types of combustion plants, regardless of their maximum capacity and location on- or off-farm, as long as they are compliant with the necessary hygiene standards laid down in Regulation (EU) No 592/2014 and emission limit values depending on their capacity (e.g. Medium Combustion Plant (MCP) Directive ((EU) 2015/2193)).

5.2.7 Limits for bulk organic C for precipitated phosphate salts

The proposal for the 3% limit value for organic C in precipitated phosphate salts & derivates was supported by some STRUBIAS subgroup stakeholders, but questioned by others due to the lack of solid techno-scientific data to support the need for such a criterion on bulk carbon. In the final proposals, the limit value of 3% is maintained for precipitated phosphate salts & derivates.

The main reason for proposing the 3% organic carbon limit is related to (i) delimiting the scope of this CMC that focuses on P-rich mineral substances of high purity formed through the precipitation of dissolved phosphate ions (see Section 5.1.1), and (ii) the fact that the **knowledge base for environmental and safety aspects is predominantly built on precipitated phosphate salts of high purity** (Section 5.3.5), and that **solid techno-scientific evidence indicates an inverse relationship between environmental and human health risks and material purity**. The organic C is not only an impurity, but also a vector for the adsorption of impurities, and thus a proxy for the purity of the precipitated phosphate salt. The precipitation process lacks a robust treatment step to remove organic pollutants and the risks associated with the precipitated salts are thus positively correlated to the amount of impurities that are co-precipitated. Therefore, it is proposed to add organic C as a parameter, together with other particular organic and biological indicators (e.g. PAHs, *Clostridium perfringens*), in the compliance scheme for this CMC.

The precipitation process involves the formation of a separable solid substance from a solution by converting the substance into an insoluble form through the addition of chemicals. Due to the nature of the process, the probability of incorporating dissolved impurities in the precipitated phosphate salt is relatively low as the precipitate is formed through the reaction of *dissociated, free* floating ions in solution that react with one another.

For instance, Mg^{2+} and NH_4^+ react with PO_4^{3-} ions to create a struvite salt that can precipitate. Instead, the impurities become part of the precipitated phosphate salts because of the incomplete separation of the precipitate from the sludge or liquefied matrix that contains particulate or suspended impurities. Organic components have a large adsorption capacity for impurities owing to their large surface area and microporous structure. Moreover, some organics are contaminants that make up a significant share of the organic carbon. Metals, specific pharmaceuticals (e.g. sulphonamides and tetracyclines) and biological pathogens are, for instance, dominantly present in the organic matrix of the sludge (Karvelas et al., 2003; Lou et al., 2018; Ye et al., 2018). Therefore, the organic carbon content in precipitated phosphate salts is positively correlated to the accumulation of impurities in the form of antibiotics (Ye et al., 2018), and other impurities have mainly been found in precipitated phosphate salts with an organic C content of > 3% (STOWA, 2015). **The relationship between organic C content and the level of impurities is thus evident and demonstrated from a theoretical and experimental point of view.** The limit value of 3% is proposed because it is the upper limit for most of the precipitated phosphate salts that have been evaluated in the risk assessment for this CMC. The limit also imposes a reduction of one order of magnitude in organic C relative to unprocessed manure and sewage sludge.

The proposal of the 3% limit value in turn enables the proposal of a **minimal compliance scheme** for this CMC that effectively excludes the extensive and expensive testing for a broad range of organic contaminants (e.g. pharmaceutical compounds and personal care products, phthalates, surfactants), and strengthens **market confidence** in fertilising materials recovered from biogenic wastes in times of increased concerns about emerging organic contaminants in consumer products and the food chain. Hence, it is indicated that the inclusion of the proposed organic C limit of 3% can effectively contribute to a robust and stable legal framework. We further refer to Section 5.3.5.1 for a detailed discussion on this topic.

5.2.8 Organic carbon content in ashes from biomass combustion plants

The conditions for the incineration of waste as laid down in the **Industrial Emissions Directive (2010/75/EU)** contain stringent temperature requirements of **850 °C for more than 2 seconds** or more than 1 100 °C for more than 0.2 seconds. Moreover, waste incineration plants shall be operated in such a way as to achieve a level of incineration **such that the total organic carbon content of slag and bottom ashes is less than 3%** or their loss on ignition is less than 5% of the dry weight of the material. For materials other than waste within the meaning of Directive 2008/98/EC, such as plant-based materials, those strict conditions only apply to the largest combustion plants (> 50 megawatt (MWth); Industrial Emissions Directive (IED)), but not to smaller plants (i.e. those covered under Regulation (EU) 2015/1189 on ecodesign requirements for solid fuel boilers or under the Medium Combustion Plant (MCP) Directive).

Those Directives ensure the implementation of the obligations arising from the Gothenburg Protocol under the UNECE Convention on Long-Range Transboundary Air Pollution.

Therefore, the primary focus of the abovementioned pieces of **legislation is on the emissions to the atmosphere, rather than on the characteristics of the solid material (i.e. soot, fly and bottom ashes, charred materials, etc.) remaining in the boiler or combustion plant**. As a matter of fact, the current fate of these materials after combustion involves their use in non-agricultural sectors (e.g. construction materials or landfilling). For their use in an EU fertilising product that is not subject to further management controls and restrictions, a detailed assessment of the risks associated with human health and the environment should therefore be performed.

Certain materials, such as plant-based residues from agriculture and forestry, **are inherently low in organo-chemical pollutants which is why stringent time-temperature profiles are not required to ensure the destruction of pollutants in the material, or proportionate considering the limited risk of emissions of certain persistent organic pollutants**.

Most residue management regulations use the organic C as a key parameter indicating the degree of organic contaminant removal as organic carbon serves as a reactive surface for the adsorption of possible contaminants, such as volatile and persistent organic pollutants (Vehlow et al., 2006). Due to the incomplete combustion of organic matter and the possibility of using biomass with a high chloride content, **organic pollutants**, such as volatile organic carbon and polychlorinated biphenyls, can be formed and can remain in the combustion residue. In addition, information on the possible environmental risks related to the possible presence of **water-soluble and insoluble organic contaminants is lacking**. Biomass ashes can contain organic aromatic structures, condensed refractory biomass and char-like particles, and some biomass ashes thus show similarities to the materials obtained from pyrolysis and gasification processes.

The proposal is therefore the following:

- The **strict time-temperature profiles** with temperatures > 850 °C shall only apply to all eligible input materials, other than **certain plant-based materials**.
- To limit the CMC 'thermal oxidation materials & derivates' to materials that are oxidised in such a way that the **total organic carbon content of the slags and bottom ashes is less than 3%, regardless of the input material applied.** This implies that partially oxidised materials shall not be allowed for this CMC and that ashes from certain plant-based materials should also meet this criterion. Possibly, such ashes with a higher organic C content shall be subject to further re-burning to levels below < 3%. Biomass that is combusted or gasified under (oxygen-limiting) conditions that results in the presence of **unburnt organic matter** (organic C content > 3%) could possibly also make an entry in the CMC '**pyrolysis & gasification materials**'. The testing regime of the latter category is somewhat different to that for 'thermal oxidation materials & derivates' because of the need for additional testing on specific contaminants (e.g. dl-PCBs, volatile organic carbon).

5.2.9 Sewage sludge as an input material for pyrolysis & gasification materials

The proposal has been made by the STRUBIAS subgroup to include sewage sludge as an input material for pyrolysis & gasification materials.

Sewage sludge may contain a set of organic pollutants, including not only persistent organic pollutants (PAHs, dl-PCB, PCDD/F), but also a broad set of organic emergent pollutants such as phthalates (e.g. di (2-ethylhexyl) phthalate (DEHP)), surfactants present in cleaners and detergents (e.g. linear alkylbenzene sulphonates (LAS) and nonylphenols (NPE)), personal care products, pharmaceuticals and endocrine-disrupting compounds (sulphonamides, galaxolide, etc.) and polymers used to bind solid particles in solid-liquid separation processes. Given the potential risks associated with these substances, **there are significant public and governmental concerns related to the recycling of sewage sludges in the European food chain**. **The spectrum of emerging contaminants in sewage sludge is extensive (Petrie et al., 2015), and much broader than for any of the eligible input materials for pyrolysis & gasification materials.** Whereas some of the above-mentioned contaminants can certainly be degraded under oxidative conditions at high temperatures, the necessary techno-scientific evidence that demonstrates their removal under oxygen-limiting and reducing conditions is lacking. It is known that stringent time-temperature pyrolysis profiles (>550°C, > 20 min) induce a weight loss in pyrolysis & gasification materials due to burning out of organic compounds (Deydier et al., 2005a; Koutcheiko et al., 2007; Ro et al., 2010; Marculescu and Stan, 2012), but **the knowledge base of studies that assessed the proportional removal of specific organic pollutants is limited and restricted to only a few organic pollutants.** Therefore, the precautionary principle should apply. Limitations in the potential of dry and wet pyrolysis/gasification processes to remove organic pollutants have been observed for organic contaminants like nonylphenol, chlorinated aromatic fractions and specific veterinary antibiotics (Weiner et al., 2013; Ross et al., 2016; vom Eyser et al., 2016). Moreover, the mechanisms, nature and soil residence times of any decay products that could be formed remain unclear, and possibly metabolites can have differential toxicity from the parent compound (Weiner et al., 2013; Ross et al., 2016; vom Eyser et al., 2016). Whereas high temperatures can effectively transform contaminants in the gaseous phase, these could also potentially be re-adsorbed on the organic carbon and soot particles that show a high adsorption potential for contaminants (e.g. on fly ash particles present in some pyrolysis & gasification materials; Mätzing et al., 2001). As indicated in the latest draft of the Best Available Techniques (BAT) Reference Document for waste incineration under the Industrial Emissions Directive (IED, 2010/75/EU) (European Commission, 2017b), **the pyrolysis of sewage sludge is a rather new method and not a widely proven technique for the treatment of waste materials.** The limited degree of technological maturity in combination with the wide spectrum of operational pyrolysis and gasification configurations induces possible risks of solid materials escaping exposure to high temperatures for this CMC, and thus insufficient organic pollutant removal levels. Thus, there is no adequate and long-term experience that indicates the suitability of pyrolysis methods to ensure the effective removal of the broad spectrum of organic pollutants that could be present in waste materials like sewage sludge (European Commission, 2017b).

Given that the solid residue quality is dependent on the process temperature (European Commission, 2017b), **the inclusion of sewage sludge on the positive input material list would also involve a possibly complex compliance scheme for this CMC, stringent time-temperature profile conditions to ensure a breakdown of bulk organic composites, or a combination of both**. Because of the heterogeneous nature of organic compounds, the compliance cost would considerably increase (e.g. GC-MS measurements). Moreover, it may be challenging for the STRUBIAS subgroup to agree on the identity of the organic compounds that should be included in the compliance scheme as well as on safe limit values for many of these emerging organic compounds.

Research findings from the last decade indicate that pyrolysis & gasification materials derived from sewage sludge will not meet the limit values for toxic metals (especially Cd, Cu, Pb, Ni, Zn) at PFC level (He et al., 2010; Hossain et al., 2010; Gascó et al., 2012; Méndez et al., 2012; Van Wesenbeeck et al., 2014; Lu et al., 2016). Heavy metals are predominantly recovered in the solid matrix (char) during the pyrolysis/gasification process, and thus hardly any metal removal takes place during the pyrolysis/gasification process (Tomasi Morgano et al., 2018). Instead, non-volatile toxic metals become more concentrated in pyrolysis & gasification materials, and no post-pyrolysis/gasification processes are described to remove the inorganic contaminants. While this is an often reported argument to exclude sewage sludge as an input material, the JRC believes that cost-effective compliance schemes for the output material could effectively control for toxic metals/metalloids. Nonetheless, it is unlikely that pyrolysis & gasification materials derived from sewage sludge can make up an important share of the pyrolysis & gasification materials on the internal market, unless the limit values for non-volatile toxic metals are respected through the mixing with other component materials. The limited market viability of pyrolysis & gasification materials derived from sewage sludge might be further undermined by the fact that the plant bio-availability of phosphorus in such materials remains largely unknown under European agricultural settings (see Section 6.2.4). The plant nutrient availability in pyrolysis & gasification materials is controlled by the coordinated cations present in the feedstock applied (Al, Fe, Ca, Mg) (Ippolito et al., 2015). As some sewage sludges are enriched in Al and Fe, relative to other nutrient-rich input materials such as manure, a reduction in the plant nutrient availability can be expected for sewage-sludge-derived pyrolysis & gasification materials relative to their manure-derived counterparts. In view of consumers' confidence in pyrolysis & gasification materials, the uncertainty associated with the plant availability of the nutrients present in sewage-sludge-derived pyrolysis & gasification materials is a concern, especially as the STRUBIAS subgroup indicated a lack of satisfactory chemical testing methods to evaluate plant nutrient and P availability in STRUBIAS materials.

In conclusion, there are two fundamental problems that have led the JRC to take its present position of not proposing sewage sludge on the positive input material list for pyrolysis & gasification materials in this second draft report. First, **the necessary science of the impacts on human health and the environment is not in place for organic contaminants, nor is the presumption of non-adverse impacts confirmed by techno-scientific evidence collected by the STRUBIAS subgroup for sewage-sludge-derived pyrolysis &**

gasification materials. Second, in view of the limited market potential for sewage-sludge-derived pyrolysis & gasification materials, **the risk of undermining consumer confidence in pyrolysis & gasification materials in general and of increasing the complexity of the compliance scheme for the CMC group is so large that it presently distorts the evaluation of any other factors involved in the assessment.** At present, the possible benefit of adding sewage sludge on the input material list is simply too low to counterbalance any eventual loss in consumer confidence for pyrolysis & gasification materials, and, by extension, fertilising materials derived from waste. This proposal is in line with the non-acceptance of contaminated input materials, including sewage sludge, for pyrolysis & gasification materials according to voluntary standardisation schemes (EBC, 2012) and national legal frameworks (Meyer et al., 2017). Moreover, it should be noted that, in view of the very local nature of certain product markets, EU Member States can still rely on the principle of optional harmonisation to make available non-harmonised fertilisers on the market in accordance with national law. Finally, the proposals in this document provide two other avenues for the safe recovery of valuable fertilising elements from sewage sludge, via precipitation of phosphate salts or thermal oxidation.

As outlined in Article 42 of the EU Fertilising Products Regulation ((EU) 2019/1009), the European Commission has proposed to be empowered to adopt delegated acts to amend Annexes I to IV to the Regulation for the purposes of adapting them to technical progress in the light of new scientific evidence. Based on the currently collected information, it is indicated that some pyrolysis & gasification manufacturing may be candidate materials to comply with the conditions outlined in Article 42(1) of the Regulation. In view of the possible development of process and quality criteria for such materials at a later stage, the JRC therefore recommends undertaking more scientific research to build up a more robust techno-scientific database to demonstrate that those materials are effectively compliant with the conditions outlined. Specifically, more techno-scientific data are required to show that EU fertilising products derived from (specific) pyrolysis & gasification materials (i) do not present an unacceptable risk to human, animal or plant health, to safety or to the environment, and (ii) are sufficiently effective to fulfil their function as EU fertilising materials.

5.3 Precipitated phosphate salts & derivates

5.3.1 Scope delimitation and possible uses

The **recovery and recycling of phosphate through precipitation processes aims to reduce the dependence on phosphate rock as a critical raw material**, the ultimate primary raw material of all the P cycling through the food and non-food system. Precipitated phosphate salts may also contain other plant nutrients (Ca, N, Mg, etc.), but their recycling is of less concern as these elements are not present on the list of critical raw materials. The scope of the CMC is outlined in Section 5.1.1, and has been expanded to **any P-rich mineral substances of high purity formed through a precipitation process, and their derivates, that are sufficiently effective at providing P to plants.**

The current **legal framework for precipitated phosphate salts or struvite-like recovered materials varies across the different EU Member States** (Dikov et al., 2014; De Clercq et al., 2015; ESPP, 2017). Precipitated phosphate salts can be legally used as a **fertiliser** in the Netherlands, Belgium, Germany, France, Denmark and the UK. As a general rule, the material needs to comply with maximum limit values for inorganic contaminants (As, Cd, Cr, Cu, Hg, Pb, Ni, Zn), biological pathogens and minimum nutrient contents in most Member States, while some countries also have maximum limit values for organic contaminants (PAHs, PCDD/F, HCH, aldrin, dieldrin, endrin, isodrin, DDT+DDD+DDE and mineral oil) based on the dry matter or the nutrient content of the fertiliser. The legislation in the Netherlands explicitly refers to sewage sludge as an input material for precipitated phosphates, but makes no mention of the recovery of phosphate salts from other input materials. Additionally, a cross-border mutual recognition initiative for struvite between the Netherlands, Flanders (Belgium) and France is under development (North Sea Resources Roundabout). **No voluntary standards for struvite have been agreed so far**.

The framework of the EU Fertilising Products Regulation ((EU) 2019/1009) allows CMCs to be used in a PFC category of the user's choice. Nonetheless, precipitated phosphate salts & derivates are likely to be contained in CE fertilising products of **Product function Category I – Fertilisers, thanks to their high P content**.

5.3.2 Input materials and reactants

5.3.2.1 Targeted input materials

Nutrient recovery as phosphate salts is restricted to liquids and slurries and the separated fractions of those materials (e.g. the liquid digestate fraction after anaerobic digestion). As a matter of fact, pilot and operational facilities that manufacture precipitated phosphate salts are mainly installed at **municipal wastewater treatment plants** and, to a lesser extent, at **agri-food processing industry sites (mostly potato and dairy)**. A small amount of operational and pilot plants recover nutrients as phosphate salts from energy crop plants, and **chemical industry waste streams** (pharmaceutical industry). At small-scale (laboratory) installations, P-recovery from **bio-waste digestates** and **other food processing industries** (e.g. rendering industry) has been documented. These material streams comprise the overall share of the P that is dissipated in a liquid or slurrified state in the EU (see Section 13).

Also, animal by-products of categories 2 and 3 can be used for the production of precipitated phosphate salts, as follows:

- The precipitation of pure Ca and Mg phosphates from complex matrices like **manure** is challenging, although progress has been made recently (Huang et al., 2015). Therefore, the implementation of nutrient recovery processes from **manure and livestock stable slurries** is limited, with the recovery of K-struvite from veal calf manure being the only process that is operational (Stichting Mestverwerking Gelderland; Ehlert et al., 2016a). Nonetheless, there exists a substantial interest in and potential for recovering P from manure and livestock stable slurries through P-precipitation (e.g. BioEcoSim phosphate salt precipitation process). Additionally, there is an interest within the STRUBIAS subgroup to manufacture precipitated phosphate salts from fish excreta and sludges. As outlined in Section 5.2.6, **the placing on the market of processed manure, products derived from processed manure and guano from bats is subject to the requirements laid down in Annex XI (Chapter I, Section 2) to Regulation (EU) 142/2011.** These requirements indicate that manure should undergo a hygienisation treatment or, alternatively, that the processed manure material should demonstrate compliance with a set of microbial *Standards* (e.g. *Ascaris* eggs, spore-forming bacteria, viruses), if authorised by the competent authority. STRUBIAS production processes are at times not compliant with the default processing methods (‘a heat treatment process of at least 70 °C for at least 60 minutes and they shall have been subjected to reduction in spore-forming bacteria and toxin formation’). Nonetheless, some STRUBIAS production pathways may even provide an equal level of environmental protection as the standard processing method (STOWA, 2016; note that this study does not focus on manure-derived precipitated phosphate salts, but indicates low levels of biological agents in struvites derived from sewage). Therefore, it is proposed that unprocessed manure can be used as an input material for the production of precipitated phosphate salts on condition that the end material that will be incorporated in an EU fertilising product is *either* compliant with a set of microbiological requirements *or* has been subject to specific standard processing methods (see Section 5.2.6 and Section 5.3.5). If accepted by the Commission, such a proposal could possibly avoid the default implementation of energy- and cost-intensive hygienisation steps in the production process, and provide a level of environmental protection that is equal to or higher than the current requirements laid down in the implementing Regulation (EU) 142/2011.
- Other animal by-products of categories 2 and 3 can in theory also be used for the production of precipitated phosphate salts (e.g. residues from gelatin production processes, digestates of leftover materials from heat-processed animal by-products in the rendering industry). These types of materials are typically **residues from the rendering industry and have thus already undergone a treatment process in line with the requirements laid down in the Animal By-products Regulation** (e.g. pressure sterilisation). Therefore, these materials have already ‘reached the end point

in the manufacturing chain' according to Regulation (EC) 1069/2009, and under the existing legal framework can already be used for production of organic fertilisers and soil improvers.

Considering the different proposed testing requirements and compliance schemes in the second set of draft proposals for STRUBIAS materials, it is proposed to separate manure, non-mineralised guano, and digestive tract content from the other animal by-products on the list of input materials.

The following materials from the food processing industry show a significant potential for P-recovery in the form of precipitated phosphate salts:

- **Waste from potato processing facilities is suitable for phosphate recovery since the wastewater contains large amounts of phosphate.** During preparation of the prebaked frozen product, potatoes are treated with sodium acid pyrophosphate ($Na_2H_2P_2O_7$) after the blanching treatment. Sodium acid pyrophosphate is needed to complex iron (Fe^{2+}). In this way, sodium acid pyrophosphate prevents iron in the potato from reacting 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 in a greyish-coloured substance that causes after-cooking grey discoloration (Rossell, 2001). The blanching treatment also causes leaching of phosphate from the potatoes, but no known contaminants are formed during the reaction.
- Many processing plants produce sludge from the processes for extraction of the crop part of interest**. Sugar mills** produce wastewater, emissions and **solid waste from plant matter and sludges** (Hess et al., 2014). 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 content of the beet, the molasses and wastewaters generated during the sugar beet processing are also rich in N, K and P (Gendebien et al., 2001; Buckwell and Nadeu, 2016). Gendebien et al. (2001) indicated, for instance, effluent P concentrations of > 100 mg PO_4^{3-}-P L^{-1}. During the further processing and the fermentation of molasses in the **brewery industry**, vinasses and wastewater may be generated from the cleaning of chemical and biochemical reactors (for mashing, boiling, distillation, fermentation and maturation) and solid-liquid separations (separation and clarification).
- **Dairy wastewaters** contain milk solids, detergents, sanitisers, milk wastes, and cleaning waters from intermediate clean-up operations at the different processing steps (storage, pasteurisation, homogenisation, separation/clarification, etc.).

By using and producing plant and edible food materials as starting materials, certain **food processing industry** waste streams and wastewaters are also intrinsically of low risk as long as the origin and additives of the wastewater components and the processing steps that may lead to contamination of the stream are controlled. After all, Commission Regulation (EU) No 1130/2011 includes a list of authorised additives approved for use in food additives, enzymes and flavourings, and substances of concern are thus effectively prohibited by law. Nonetheless, a significant proportion of the wastewaters originates from the washing of installations. Typical **cleaning agents** used in the food processing industry sector are (European Commission, 2006c):

- 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. The use of detergents in the EU is controlled through Regulation (EC) No 648/2004 which only enables the use of biodegradable detergents. Moreover, recent research indicated that common cleaners and surfactants, such as linear alkyl benzenesulphonates are not co-precipitated to a significant extent in phosphate salts (Egle et al., 2016). Specific **disinfection products**, such as hypochlorite, can form disinfection by-products which can be carcinogenic and highly ecotoxic in their nature, and no information is available on their behaviour during precipitation processes. The authorised biocides that are used in the food and feed area are listed as product type 4 of main group 1 of Annex V to Regulation (EU) No 528/2012. It is proposed to enable the use of wastewaters from food processing industries, **unless previous processing steps involved contact with:**

- biocides within the meaning of Regulation (EU) No 528/2012 other than those defined as product type 4 of main group 1 of Annex V thereto;
- animal by-products or derived products falling within the scope of Regulation (EC) No 1069/2009 for which no end point in the manufacturing chain has been determined in accordance with the third paragraph of Article 5(2) of that Regulation.

Moreover, it is proposed to include semi-solid **waste materials from the agro-food processing industry** (e.g. molasses, vinasses) as input materials. On the proposed input material list, such materials are included as **bio-wastes** within the meaning of Directive 2008/98/EC and Directive EU 2018/851 (i.e. 'biodegradable garden and park waste, food and kitchen waste from households, offices, restaurants, wholesale, canteens, caterers and retail premises and comparable waste from food processing plants').

The precipitation process often takes place on digestates that predominantly derive from one or more of the above listed feedstocks. At anaerobic digestion plants, these feedstocks are often complemented with other organic materials. While most of these co-digested materials (e.g. plant-based materials) are proposed as target input materials in the foregoing sections, specific residues from the bio-energy sector are not listed as such. Therefore, the STRUBIAS subgroup highlighted the need to evaluate the possible inclusion of **residues from biodiesel and bio-ethanol production** (e.g. fermentation residues, glycerine/glycerol). This proposal received support from the other STRUBIAS subgroup experts because these streams do not contain contaminants at levels of concern. As a matter of fact, such residues are often used as animal feed, and appear on the positive list to be used as fertilisers in specific EU Member States (e.g. see Annex Aa. to Article 4 to the Fertilisers Regulation in the Netherlands).

Large amounts of **wastewater** are produced by the **energy production industry, pulp and paper industry, P-fertiliser, chemical and metal industry and pharmaceutical industry** (Moloney et al., 2014; Eurostat, 2016). The wood pulp and paper industry is the non-food sector that contributes most to P-losses (van Dijk et al., 2016), but the P is present in a highly diluted form (0.2-0.4 mg L^{-1}). Phosphorus losses from other non-food sectors, more specifically **chemical and metal waste streams,** are often low and diluted (van Dijk et al., 2016), which explains why P-precipitation is technically challenging It was indicated that P-recovery from the P-fertiliser industry actually involves an optimisation of a manufacturing process starting from primary raw materials, and could thus possibly be covered under a different CMC (CMC 1 – Virgin material substances and mixtures). Moreover, specific chemical waste streams **may contain contaminants that are present in large quantities** (e.g. pharmaceutical compounds). In many of these processes, the behaviour of specific contaminants during the precipitation process is unknown (e.g. chlorinated organic compounds present in treated pulp and paper sludges; Pokhrel and Viraraghavan, 2004). Hence, given that (i) these streams may possibly contain ‘rare’ contaminants that are not present in the other input materials of biogenic origin proposed for this CMC, (ii) the behaviour of some of these contaminants during the precipitation process is unknown, and (iii) the overall objective is to enable simple, low-cost and straightforward compliance schemes, it **is proposed to exclude waste from such materials as input material**. It is noted that the optional harmonisation principle will allow the placing on the market of fertilising materials originating from very specific input materials (see Section 4.2.3)

5.3.2.2 Reactants

The precipitation process is based on the addition of **chemical reactants, phosphate counter ions, and pH regulators** (chemicals, CO_2) in a reactor (Quintana et al., 2004; Le Corre et al., 2009; Rahman et al., 2014). Different reactor types and configurations exist (see Section 14.1), some of them with a **seed bed** (sand, struvite, but also poorly soluble Mg compounds (MgO), stainless steel mesh, pumice stone and borosilicate glass may be used; Kataki et al., 2016). The use of Mg-containing industrial by-products has been indicated to reduce operational costs (Quintana et al., 2004). However, as outlined in Section 4.2.2, the STRUBIAS recovery rules will adopt the provisions of the use of industrial by-products as laid down in the general framework of the EU Fertilising Products Regulation ((EU) 2019/1009) as developed by the Commission.

5.3.2.3 Proposals for input materials and reactants

Based on the information presented in Sections 5.3.2.1 and 5.3.2.2, the following proposal is put forward for input materials and reactants for the manufacturing of precipitated phosphate salts:

An EU fertilising product may contain precipitated phosphate salts exclusively obtained through precipitation from one or more of the following input materials:

i) wastewaters and sewage sludge from municipal wastewater treatment plants;
j) derived products referred to in Article 32 of Regulation (EC) No 1069/2009 for which an end-point in the manufacturing chain has been determined in accordance with the third subparagraph of Article 5(2) of that Regulation;
k) animal by-products, the products derived from which are referred to in Article 32 of Regulation (EC) No 1069/2009 and have an end-point in the manufacturing chain determined in accordance with the twastewaterhird subparagraph of Article 5(2) of that Regulation;
l) wastewaters from food processing, pet food, feed, milk and drink industries, unless the processing steps involved contact with biocides within the meaning of Regulation (EU) No 528/2012 other than those defined as product type 4 of main group 1 of Annex V thereto;
m) residues from the production of bioethanol and biodiesel as referred to in Directive 2009/28/EC of the European Parliament and of the Council of 23 April 2009 on the promotion of the use of energy from renewable sources and amending and subsequently repealing Directives 2001/77/EC and 2003/30/EC[22];
n) bio-waste within the meaning of Directive 2008/98/EC resulting from separate bio-waste collection at source, other than those materials included in points (b) and (c);
o) living or dead organisms or parts thereof, which are unprocessed or processed only by manual, mechanical or gravitational means, by dissolution in water, by flotation, by extraction with water, by steam distillation or by heating solely to remove water, or which are extracted from air by any means, except:
    v. materials originating from mixed municipal waste,
    vi. sewage sludge, industrial sludge or dredging sludge,
    vii. animal by-products or derived products within the scope of Regulation (EC) No 1069/2009 for which no end-point in the manufacturing chain has been determined in accordance with the third subparagraph of Article 5(2) of that Regulation, and
    viii. materials mentioned in points (a) to (f);
p) substances and mixtures, other than:
    vii. those listed under points (a) to (g),
    viii. waste within the meaning of Directive 2008/98/EC,
    ix. substances or mixtures which have ceased to be waste in one or more Member States by virtue of the national measures transposing Article 6 of Directive 2008/98/EC,
    x. substances formed from precursors which have ceased to be waste in one or more Member States by virtue of the national measures transposing Article 6 of Directive 2008/98/EC, or mixtures containing such substances,
    xi. non-biodegradable polymers, and

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[22] OJ L 140 5.6.2009, p. 16.

xii. animal by-products or derived products within the scope of Regulation (EC) No 1069/2009.

In addition, precipitated phosphate salts shall be obtained through precipitation from any material listed in points (a) to (h), or combination thereof, processed by manual, mechanical or gravitational means, by solid-liquid fractionation using biodegradable polymers, by dissolution in water, by flotation, by extraction with water, by steam distillation or by heating solely to remove water, by thermal hydrolysis, by anaerobic digestion or by composting. The temperature of such processes shall not be raised above 275 °C.

[Note: The exclusion of a material from a lettered item does not prevent it from being an eligible component material by virtue of another lettered item.]

Related to animal by-products, the following proposals are put forward for further consideration and evaluation by the Commission and other bodies as a possible end-point in the manufacturing chain in accordance with the third subparagraph of Article 5(2) of Regulation (EC) No 1069/2009:

1. Precipitated phosphate salts & derivates that have been derived from manure, non-mineralised guano, and digestive tract content pursuant to Regulation (EC) No 1069/2009 provided that they:
   - are compliant with the proposed criteria for CMC XX,
   - have no presence of *Clostridium perfringens* in a concentration of more than 100 CFU/g fresh mass, and
   - have no presence of *Ascaris* sp. eggs in a 25 g fresh mass.

2. Precipitated phosphate salts & derivates that have been derived from animal by-products and derived materials from category 2 or category 3 material as defined by Regulation (EC) No 1069/2009 provided that they:
   - are compliant with the proposed criteria for CMC XX,
   - have been hygienised in accordance with the conditions for pressure sterilisation or with other conditions to prevent risks arising to public and animal health, in accordance with the requirements laid down pursuant to Article 15 of Regulation (EC) No 1069/2009, or
   - they are digestion residues from transformation into biogas as set out in Annex V to Regulation (EU) No 142/2011.

5.3.3 Production process conditions

It is proposed that phosphate salts can be recovered at plants that are specifically designed for the **purpose** of producing **fertilising materials** or be the by-product resulting from a process aimed at **producing different primary outputs** (e.g. energy and treated water) as long as end material quality conditions are fulfilled. For this reason, operational facilities can be a **stand-alone** installation or be **integrated** into another system.

5.3.3.1 Pre-processing

Phosphate salts are precipitated from PO_4^{3-} ions present in liquids and slurries, but pretreatments exist that transform organic-bound P into solution as phosphates (e.g. combined aerobic treatment followed anaerobic digestion). **Solid-liquid separation** techniques are then applied to remove interfering ions, colloidal and suspended particles from a phosphate-containing liquid solution. Also, the concentration of P-rich precipitates together with the other colloidal and suspended particles during P-recovery processes are facilitated through solid-liquid separation processes.

In the event that phosphate is recovered from liquid and slurry fractions, pretreatments are applied to increase the content of orthophosphate ions (PO_4^{3-}) present in the liquid. **A pretreatment is often a prerequisite to increase the P-recovery efficiency and is directly applied on input materials**. Based on the scientific literature (Alonso Camargo-Valero et al., 2015; Bamelis et al., 2015; Camargo-Valero et al., 2015) and the information received from the STRUBIAS subgroup, applied pretreatments include **acidification and liming**, **thermal hydrolysis (at temperatures of 150-180 °C)**, **pressure** modifications, the circulation of wastewater in **enhanced biological phosphorus removal** (EBPR) tanks, chemical P-removal processes, and **anaerobic digestion**. These techniques are applied in existing municipal wastewater treatment plants or at operational pilot P-recovery facilities.

Coagulants, pH regulators, chemical and biological stabilisers, detergents and flocculants are commonly used to increase the efficiency of removal of waste fractions from food processing facilities.

Solid-liquid separation techniques (e.g. centrifuge, sieve belt, filter press, screw press, rotation liquid sieve, vibration screen, sedimentation tank, dissolved air flotation, lamella separator, filtration by means of a straw bed, ultrafiltration using semi-permeable membranes, and reverse osmosis), possibly after the application of polymers, are generally applied at some stage during the pre-processing stages of the input material preceding the precipitation of phosphate salts. **Organic or inorganic coagulants** are sometimes used to achieve a good separation between solid and liquid phases (Hjorth et al., 2010; Schoumans et al., 2010). Usually, coagulants, flocculants and polymers are polyelectrolytes, aluminium and iron sulphates and chlorides, calcium oxides and hydroxides, polyacrylamide or also magnesium oxide and magnesium hydroxides. The above-mentioned techniques are all based on **mechanical separation,** possibly complemented by the addition of chemical substances, **mild temperature treatment** and membrane technologies. No limitations on the use of such techniques are proposed as long as the polymers applied have no adverse effects on animal or plant health, or on the environment (see requirements proposed for CMC 8 and 9 in the EU Fertilising Products Regulation ((EU) 2019/1009). Hydrothermal carbonisation and other possible pretreatment methods are presently not listed due to (i) the unknown behaviour of possible contaminants (e.g. organic acids, oil-like substances, phenols, furfurals, and their derivatives) formed *de novo* by these pretreatments during the precipitation process, and (ii) the limited applicability of these pretreatment methods so far, which is why no comprehensive evaluation of the additional risks associated with these pretreatment methods is possible at present.

As phosphate salt precipitation can take place on materials obtained after applying the above-mentioned techniques, it is proposed to make a reference to chemicals that can be used (see point g of Section 5.3.2.3) and pretreatments in the paragraph related to the input materials as follows (see also Section 5.3.2):

h) substances and mixtures, other than:
   i. those listed under points a) to g),
   ii. waste within the meaning of Directive 2008/98/EC,
   iii. substances or mixtures which have ceased to be waste in one or more Member States by virtue of the national measures transposing Article 6 of Directive 2008/98/EC,
   iv. substances formed from precursors which have ceased to be waste in one or more Member States by virtue of the national measures transposing Article 6 of Directive 2008/98/EC, or mixtures containing such substances,
   v. non-biodegradable polymers, and
   vi. animal by-products or derived products within the scope of Regulation (EC) No 1069/2009.

In addition, precipitated phosphate salts shall be obtained through precipitation from any material listed in points a) to h), or combination thereof, processed by manual, mechanical or gravitational means, by solid-liquid fractionation using biodegradable polymers, by dissolution in water, by flotation, by extraction with water, by steam distillation or by heating solely to remove water, by thermal hydrolysis, by anaerobic digestion or by composting. The temperature of such processes shall not be raised above 275 °C.

The limit of **275 °C** is proposed based on the upper temperature limit for thermal treatments investigated and applied, i.e. the thermal hydrolysis processes (Barber, 2016). There is no risk of the *de novo* formation of persistent organic compounds such as PAHs, PCDD/Fs or dl-PCBs within the proposed temperature range (Vehlow et al., 2006; Van Caneghem et al., 2010).

5.3.3.2 Core process

According to the conditions on production process provisions, a wide range of materials could theoretically be precipitated, some of which clearly fall beyond the scope of this CMC. Examples of materials that could be produced include for instance precipitates other than those rich in phosphates (e.g. calcium sulphate), or sludge-like insoluble precipitates formed in chemical wastewater treatment plants by the application of chemicals ($FeCl_3$, $Fe_2(SO_4)_3$), alum ($Al(SO_4)_3$) or lime ($Ca(OH)_2$). Obviously, these materials are not within the scope of this CMC. These materials have a low P content (e.g. 1-5% P for sewage sludge; Janssen and Koopman, 2005; Herzel et al., 2016). **Hence, setting a criterion on the minimum P_2O_5 content will delimit the scope of the CMC category, in line with the objective of**

providing an avenue for the recovery of P-rich materials that can directly be used as fertilisers on land or as intermediates for fertiliser material production processes. The JRC is aware that a criterion for minimum P-content has been proposed at PFC level (PFC 1 – Fertilisers), but, at the same time, it is believed that P-content is the preferred manner to ensure that materials fit within the scope of this CMC. Alternative approaches based on the measurement of the mineral composition (e.g. X-ray diffraction (XRD), mineral liberation analyses (MLA) using a Scanning Electron Microscope (SEM) - Mineral Energy Dispersive Spectra (MEDS), etc.) are less straightforward and will lead to higher compliance costs (EUR 2 000-4 000 per sample). Measuring the P_2O_5 content will not lead to additional compliance costs as this parameter will be measured as part of the conformity assessment procedures for PFC 1 – Fertilisers.

The targeted precipitated phosphate salts typically have a P_2O_5 content of more than 25% (Table 2). Nonetheless, the P_2O_5 content could be lower for specific phosphate salts, especially for hydrated salts. Sodium phosphate dibasic 12-hydrate ($Na_2HPO_4.12H_2O$) was identified as the phosphate salt with the lowest P_2O_5 content (20%). Since a limited amount of impurities might be co-precipitated together with the pure crystals without major environmental and human health risks, a minimum P_2O_5 content that corresponds to 80% of that mineral is proposed as a criterion to delimit the scope of this CMC:

The precipitated phosphate salt shall have a minimum P_2O_5 content of 16% of the dry matter content[23].

Current end materials of operational plants typically have a P_2O_5 content > 20% (see Section 15.1.1), indicating that this criterion should be not a restricting element in the compliance scheme. **Together with the criterion on maximum organic C content (< 3%, see Section 5.3.5.1), these requirements will clearly differentiate precipitated phosphate salts from materials that fall beyond the scope of this CMC.** Based on the reported values from operational plants, the 20% threshold value for the P_2O_5 content of precipitated phosphate salts is an achievable target for industrial P-recovery processes (Section 15.1.1).

There are other processes (e.g. *flocculation* of suspended P present in sludges, *adsorption* of P) that could lead to the production of P compounds. Because the nature of such processes is different to *precipitation* (as outlined in Section 5.1.1), the non-selective inclusion of compounds other than phosphates occurs, leading to the production of materials that have a lower P content, and possibly a higher contaminants content. Assessing such materials and production processes other than precipitation processes falls beyond the scope of the CMC precipitated phosphate salts & derivates. Note that the proposed scope expansion for this CMC from struvite to different types of salts impedes the use of P_2O_5 content as a sole measure for purity. Due to the varying P content of the different salts (Table 2), the minimum salt purity can vary considerably, with a minimum purity of < 30% for specific salts (e.g. magnesium hydrogen phosphate, having 59% P_2O_5 in pure form). Therefore, a limit value for organic C has been proposed to delimit the scope of this CMC (see Section 5.3.5.1).

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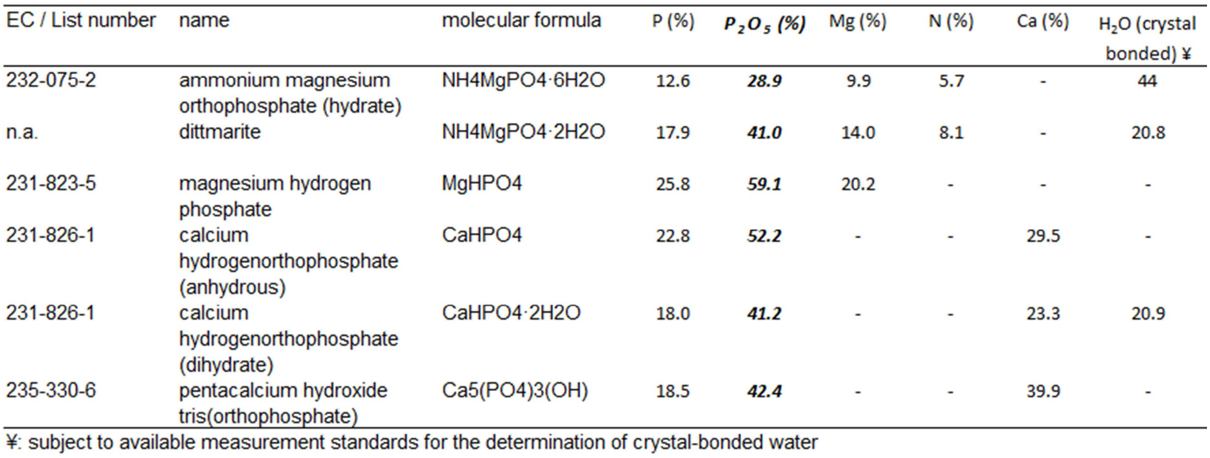
[23] As measured using vacuum drying at 40 °C to avoid the loss of crystal-bound water.

Table 2: Elemental composition of different types of precipitated phosphate salts in pure form as dried at a temperature of 40 °C until constant weight



EC / List number	name	molecular formula	P (%)	***P_2O_5 (%)***	Mg (%)	N (%)	Ca (%)	H_2O (crystal bonded) ¥
232-075-2	ammonium magnesium orthophosphate (hydrate)	NH4MgPO4·6H2O	12.6	***28.9***	9.9	5.7	-	44
n.a.	dittmarite	NH4MgPO4·2H2O	17.9	***41.0***	14.0	8.1	-	20.8
231-823-5	magnesium hydrogen phosphate	MgHPO4	25.8	***59.1***	20.2	-	-	-
231-826-1	calcium hydrogenorthophosphate (anhydrous)	CaHPO4	22.8	***52.2***	-	-	29.5	-
231-826-1	calcium hydrogenorthophosphate (dihydrate)	CaHPO4·2H2O	18.0	***41.2***	-	-	23.3	20.9
235-330-6	pentacalcium hydroxide tris(orthophosphate)	Ca5(PO4)3(OH)	18.5	***42.4***	-	-	39.9	-

¥: subject to available measurement standards for the determination of crystal-bonded water

The chemicals, pH regulators and seed beds required for the precipitation process are discussed and identified together with the proposals for the list of eligible input materials and reactants in Section 5.3.2.2.

5.3.3.3 Post-precipitation manufacturing steps

Nutrients in most recovered Ca and Mg phosphate salts show a high plant availability (Section 5.3.4) and the material has no adverse effects on the environment and human health during the handling and use phase as a fertiliser (see Section 5.3.5). Therefore, precipitated phosphate salts that meet the proposed criteria of this project **can be used directly as a fertiliser** or as an ingredient in the tested physical fertiliser blends. As indicated during the STRUBIAS Kick-off Meeting and by Six et al. (2014), there is considerable interest on the part of the fertiliser blending and mineral fertilising industry in using precipitated phosphate salts as **an intermediate raw material in its production processes:**

- Given that the P in most pure precipitated phosphate salts is already in plant-available form, there is no need for acidulation and further chemical reactions of the precipitated phosphate salts. Hence the materials may be ready for use in **physical fertiliser blends**, for instance together with acidulated phosphate rocks of CMC 1 (*Virgin material substances and mixtures)* (Six et al., 2014). Nonetheless, the different components should meet certain criteria, with respect to purity and granulometry (Formisani, 2003). Certain combinations of molecules should be avoided due to possibly occurring chemical reactions in the granulator that cause nutrient loss or reduce the water-solubility of specific elements in the blend. A potentially limiting factor for the further direct use of precipitated phosphate salts could also be the moisture content of the (hydrated) precipitated phosphate salts and the chemical compatibility with other selected fertilising compounds in physical blends.

- Some precipitates may not be suitable as fertilising material due to their lower plant P-availability, but may be appropriate **intermediates for chemical manufacturing processes of water- or acid-soluble P-fertilisers**. This especially holds true for P-recovery processes leading to the production of end materials such as hydroxyapatite (Kabbe, 2017), a phosphate salt with a lower plant P-availability, especially at alkaline pH (Arai and Sparks, 2007). Moreover, forthcoming P-recovery processes could also rely on similar two-phase processes, i.e. through the precipitation of other materials with low plant P-availability (e.g. Fe-phosphates) that can be used as an intermediate in a P-fertiliser production process (Wilfert et al., 2015).

Hence, it is desirable to enable the further chemical processing of precipitated phosphate salts in the recovery rules to permit the production of fertilising materials of the producer’s choice. Given the framework of the EU Fertilising Products Regulation ((EU) 2019/1009), such chemical processing steps should be covered under this CMC and thus be included in the recovery rules (see Sections 4.2.1 and 5.2.4).

As outlined in Section 5.2.4, **‘two-step’** manufacturing processes enable the further processing of intermediate materials. The principle of the two-step manufacturing processes for this CMC is that the precipitation process isolates a material that can be considered ‘safe’ due to the low values of organic contaminants, biological pathogens, and inorganic contaminants. The safety of this material is assured by a combination of process requirements and parameter testing on the precipitate (e.g. biological pathogens, maximal organic C contents). Therefore, subsequent chemical manufacturing steps can be applied to produce a fertilising material of a preferred chemical composition, as long as no new ‘risk materials’ are introduced. Risk materials are, in this context, defined as materials which can introduce biological contamination or other unintentional organic contaminants. Hence, it is proposed to enable only virgin materials (and possibly *safe* industrial by-products as permitted within the framework of the EU Fertilising Products Regulation ((EU) 2019/1009), see Section 4.2.2) for such post-precipitation manufacturing steps, and **to exclude waste materials, materials which have ceased to be waste and substances formed from precursors which have ceased to be waste, waste animal by-products** (similar to the provisions for CMC 1). Because the process limitations on the second step of the processes are minimal, a high degree of **sovereignty is allowed for manufacturers to apply processes of their choice** and to promote the development of innovative processes that start from intermediate materials. The proposed provisions cover all wet chemical processes that involve the removal of P along with other elements from the precipitate by elution, after which the dissolved elements are recovered by solidification, precipitation, ion exchange or membrane technologies. **The end material from the whole manufacturing process (‘precipitated phosphate salts & derivates that will be incorporated in the EU fertilising product’) will then be subject to further testing on dry matter content and Al and Fe contents** (see Section 5.3.4).

Therefore, it is proposed to include the following point in the CMC recovery rules:

An EU fertilising product may contain derivates from precipitated phosphate salts compliant with paragraphs 1 to 3 as produced through one or more chemical manufacturing steps that react precipitated phosphate salts with materials listed under point h) of paragraph 1 that are consumed in or used for chemical processing. The derivate manufacturing process shall be executed so as to intentionally modify the chemical composition of the precipitated phosphate salt.

5.3.3.4 Finishing steps

No specific requirements for 'finishing' techniques that relate to the agglomeration or washing of materials have to be included at CMC level. Post-processes (e.g. modification of size or shape by mechanical treatment, washing with water) are normal industrial practice and any materials required are included on the input material list.

Hence, precipitated phosphate salts & derivates may undergo further post-processing steps with the following intentions:

- Improve the purity of the material and to remove any physical and organic impurities by **washing** with substances that do not change the chemical structure of the crystalline phases of the precipitate.
- **Agglomerate** the product as pellets or granules using a variety of equipment including rotating pans and drums, fluidised beds and other specialised equipment. It should be noted that granulation processes might cause the heating of the precipitated phosphate salts, which could alter the chemical composition of the product due to dehydration.

5.3.4 Agronomic value

The objective for materials from the CMC precipitated phosphate salts & derivates is to supply plants with soluble phosphates as a macronutrient. **Recovered Ca and Mg phosphate salts show generally good plant P-availability, with plant responses to fertilisation being similar to mined and synthetic P-fertilisers currently on the market (see Section 6.2.2).**

Some members of the STRUBIAS subgroup also formulated requests to include recovered Fe phosphates in this category, thus as CMC materials that can be incorporated into the EU fertilising product (e.g. KREPRO process). **Aluminium and iron phosphates are, however, not registered as fertilisers pursuant to Regulation EC No 1907/2006 (REACH).** The material properties of the ferric phosphates (24-29% Fe) obtained through the KREPRO process that were proposed as end materials to be included in this CMC showed high organic C contents (6-29%), low to moderate P contents (6.6-30.6%, expressed as P_2O_5), and molar Fe/P ratios in the range of 1.3-5.1. As only limited testing has been performed on these materials, their agronomic value remains uncertain due to concerns over the plant availability of Fe-complexed phosphates (Lindsay and De Ment, 1961; Ghosh et al., 1996; Wilfert et al., 2015). Kahiluoto et al. (2015) indicated good plant availability for sludges with moderate molar Fe/P ratios of 1.6, but not for materials with higher molar Fe/P ratios of 9.8. Moreover, if the P in Fe and Al phosphates were plant-available, there would be a substantial risk of the soluble aluminium or iron forms inducing plant toxicity as the liberation of P from such complexes involves a breakup of the chemical bonds, and thus the liberation of free Al and/or Fe in the soil solution. Both Fe and Al can be toxic if supplied in excessive concentrations to plants (Connolly and Guerinot, 2002). **Hence, the direct use of Al and Fe phosphates as CMC materials is not desirable as (i) the agronomic value of such materials of a P source remains unknown, thus leading to a tangible risk for the accretion of P in the soil, and (ii) liberated phosphate counter-ions in the form of Al or Fe could cause potentially toxic plant effects.** Therefore, the following criterion is proposed:

Precipitated phosphate salts & derivates contained in the EU fertilising product shall have a maximum of 10% of the sum of elemental Al and elemental Fe of the dry matter content[24].

The proposed cut-off value allows Fe and Al to be present in relatively low amounts in the precipitated phosphate salts. This is important as eligible input materials, such as sewage sludge, are often rich in Al or Fe, some of which will be transferred as impurities to the precipitate. Typical values for Al and Fe in precipitated phosphate salts are in the range of 0-4% and 0-2% for Al and Fe, respectively (González-Ponce et al., 2009; Gell et al., 2011; Antonini et al., 2012; Uysal et al., 2014; Vogel et al., 2015; Siciliano, 2016). Assuming a minimum P_2O_5 content of 20% and a maximum Fe content of 10% in the precipitated phosphate salt, the molar Fe/P ratio in the material would be ~0.65, a value at which plant P availability would not be compromised (Kahiluoto et al., 2015). The proposal on maximal Al/Fe contents is also in line with the technical report of Ehlert et al. (2016a) which evaluated the possible inclusion of 'recovered phosphates' in the Dutch fertiliser legislation, and recommended to constrain the category to Ca and Mg phosphates.

In line with Wilfert et al. (2015), there may be *potential* for P-recovery from sludges containing Al-P and Fe-P complexes as *input materials or intermediates* for the production of precipitated phosphate salt fertilisers. This is the reason why they have been included in this document as eligible input materials (Section 5.3.2) and no criterion for the maximum Fe and Al content of the intermediate precipitate (Section 5.3.3.2) has been proposed. Proposed limits on Fe and Al only apply to the finished precipitated phosphate salts & derivate materials contained in the EU fertilising product. Altogether**, it is possible to use Al/Fe-rich compounds as an input material to produce an end material pertaining to this CMC. Moreover, a two-step precipitation process can be applied, where Fe is used to precipitate phosphate into Fe phosphates (with a P_2O_5 content of > 20% and an organic C content of < 3%) as long as this precursor is later transformed into a material that meets the proposed requirement on maximum Al and Fe content (as described in Section 5.3.3.3).**

It is noted that some precipitated phosphate salts could also contain the macronutrients N and/or K (e.g. struvite, K-struvite). Although these elements are typically readily available to plants in STRUBIAS materials, the ability of precipitated phosphate salts to contribute

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[24] As measured using vacuum drying at 40 °C to avoid the loss of crystal-bound water.

significantly to plant N and K uptake is minor because plant N and K demands are much higher than the N/K quantities that are applied under good fertiliser management conditions.

5.3.5 Environmental and human health and safety aspects

Certain input materials that are targeted for nutrient recovery through P-precipitation have high contents of inorganic and organic pollutants (Boxall, 2012), which may potentially be transferred to the precipitated phosphate salt. Some of these pollutants can be monitored by chemical analysis (e.g. polyaromatic hydrocarbons (PAHs), metals like Cd and Hg) in the resulting materials. However, in recent years concerns have been raised on a broad variety of compounds including natural toxins, human pharmaceuticals, phthalates, veterinary medicines, pesticides and derivates, nanomaterials, personal care products, paints and coatings, etc. (Boxall, 2012). These organic contaminants are of particular concern as analytical methods to trace these - by nature heterogeneous - compounds are complex and costly. Moreover, risk assessments on these pollutants are often lacking, which makes it challenging to derive limit values.

It is important to recognise that precipitated phosphate salts **are a new type of industrial material, and, compared to better-known materials, relatively few samples have been tested for contaminants, especially of organic origin.** As already outlined in Section 5.3.2, most laboratory, pilot and operating P-precipitation plants from which information on environmental and human health safety aspects is available are reliant on municipal wastewaters as inputs (both for struvite and calcium phosphates). Nevertheless, data for different food processing industries and manure and livestock stable slurries are also available (Section 15.1.2). **Data on contaminants, especially organics, are mainly available for precipitated phosphate salts of high purity with a low C content** (especially for struvites, but also for dicalcium phosphates – confidential data) and limited information is available for precipitated phosphate salts with relatively higher levels of organic C.

5.3.5.1 Organic chemical contaminants

Identification of specific contaminants

Given that there is no thermal destruction phase during the production of precipitated phosphate salts, it is **relevant to assess the environmental and human health impacts of the presence of specific organic contaminants in phosphate salts.** Possible pre-processing techniques such as anaerobic digestion and wet digestion, pasteurisation, and thermal hydrolysis (Section 5.3.3.1) might cause a substantial reduction in the risk of organic contaminants (Lukehurst et al., 2010), but are not always applied in the production process and do not secure the removal of the wide variety of organic pollutants that can be found in some input materials. Therefore, a hazard exists for the absorption and inclusion of organic contaminants in the end material of the recovery process.

It is pertinent to evaluate hazards according to the **probability of occurrence** in the framework of a risk assessment. In this context, relevant frameworks for comparison are the direct land application of sewage sludge, manures, and digestates from manure slurries and bio-waste on land (Langenkamp and Part, 2001; Smith, 2009; WCA environment, 2014; Petrie et al., 2015; Ehlert et al., 2016b). Such comparisons are useful as the precipitation process **is a separation technique, rather than a transformation process**. Therefore, only contaminants present in the input materials can be transferred to the precipitate. In contrast to thermal oxidation materials & derivates and pyrolysis & gasification materials, there is no *de novo* formation of contaminants in this separation process.

Although emerging pollutants require supplementary screening (see below), risk assessments for sewage sludge, manures and designated bio-wastes from food and feed industry and residues from agriculture and landscape management indicate that organic contaminants are not expected to pose major health problems to the human population when those are directly reapplied on agricultural land (Langenkamp and Part, 2001; Smith, 2009; WCA environment, 2014; Ehlert et al., 2016b). This view is based on a technical evaluation of the situation, which acknowledges the concentration of organic contaminants in sewage sludge in relation to their behaviour and fate in the soil. It was concluded that the biodegradation and behaviour of organic compounds in the soil together with the low levels of crop uptake minimise the potential impacts of most organic pollutants on human health (Langenkamp and Part, 2001; Smith, 2009).

Nevertheless, the **risk assessments also indicated that certain substances present in input materials** like sewage sludges, manures, digestates and (industrial) wastewaters **require further investigation** (UMK-AG, 2000; Langenkamp and Part, 2001; Smith, 2009; WCA environment, 2014; Ehlert et al., 2016b): (i) phthalates, (ii) surfactants present in cleaners and detergents, (iii) PAHs, PCDD/Fs and dl-PCBs, (iv) plant protection products and biocides, and (v) antibiotic and other drug residues, personal care products, and endocrine-disrupting compounds. Therefore, it is relevant to evaluate to what extent the above-mentioned substances can be transferred to the precipitated phosphate salts:

- From the database compiled by Egle et al. (2016) and the confidential information received from the STRUBIAS subgroup, it could be observed **that phthalates, surfactants and cleaning substances** (as measured by nonylphenol and nonylphenol ethoxylates with 1 or 2 ethoxy groups (NPE) and linear alkylbenzene sulphonates (LAS)) are generally present in low quantities in phosphate salts that are recovered from municipal wastewaters, and several orders of magnitude below the limit values for these compounds established in different EU Member States and the provisions of Directive 86/278/EEC (Langenkamp and Part, 2001).
- Data on **PAHs, PCDD/Fs and dl-PCBs** in precipitated phosphate salts are very limited. Kraus and Seis (2015) found very low quantities of these persistent organic pollutants in three struvites. Dioxin-like PCBs and PCDD/F contents in precipitated phosphate salts were also well below levels of concern

according to other studies (Uysal et al., 2010; confidential information provided by the STRUBIAS sub-group; Kraus and Seis, 2015; Egle et al., 2016). Confidential data provided by the STRUBIAS subgroup indicated a PAH content of 2.5 mg kg^{-1} fresh matter for struvite recovered from digested sludge, although it is noted that the organic C of the respective sample was unknown. Given that (i) the current dataset is limited to seven samples, and (ii) that sewage sludge as well as derived composts and digestates may contain PAH contents in the range of 6-20 mg kg^{-1} (2008; Salado et al., 2009; Saveyn and Eder, 2014), **it is - based on the precautionary principle - proposed to monitor and limit PAHs in precipitated phosphate salts when those are derived from sewage, an input material that is possibly rich in PAHs (Langenkamp and Part, 2001)**.

- Limited information is available for **plant protection products and biocides**. This issue is especially relevant when digestates from plant-based and animal products are used as input materials for P-recovery. However, the use of known potentially unsafe plant protection products (e.g. aldrin, dieldrin, HCHs, HCBs, DDT/DDD/DDE) has been largely phased out, which is why the risk is inherently low. In their study on the safety of designated bio-wastes from the food and feed industry and residues from agriculture and landscape management, Ehlert et al. (2016b) indicated that data on organic micropollutants in crop digestates are largely missing, but that such compounds do not generally restrict the use of digestates on land as there is no major risk for the environment and human health.

- The use of **pharmaceutical products and personal care products** has caused concerns about the presence of pharmaceutical compounds in precipitated phosphate salts derived from municipal wastewaters, and more specifically separately collected urine, and stable manure and livestock slurries (Ronteltap et al., 2007; Ye et al., 2017). Residual antibiotics can affect the soil and aquatic microbiome, resulting in differential inhibition of certain microorganisms and in perturbations in community composition. The increasing use of antibiotics in medicine, veterinary medicine, and agricultural production systems has coincided with increasing development of high levels of antibiotic resistance and novel antibiotic resistances (Popowska et al., 2012).
- **Antibiotics have been widely applied in the livestock industry** and veterinary antibiotics in manures have generated significant concern; these residues have a great capacity to disturb the natural ecological balance, persist in soils, be transported towards water bodies, and trigger an increase of resistant bacteria in the environment (Tong et al., 2009). There are concerns that the presence of co-contaminants may have synergistic or additive ecotoxicological impacts upon soil functions (Horswell et al., 2014). Moreover, the uptake of antibiotics by children and pregnant women should be constrained, and some compounds have been reported to cause an array of

animal and human diseases including nephrotoxicity, hepatotoxicity, and hypersensitivity reactions, hypouricemia, hypokalemia, and proximal and distal renal tubular acidosis. The most common antibiotics in swine wastewater are tetracyclines, sulphonamides, and fluoroquinolones (Li et al., 2013). Residual concentrations of tetracyclines from fresh animal wastes have been reported to range from 11 ng g^{-1} to 880 ng g^{-1} (Daghrir and Drogui, 2013). Ye et al. (2018) indicated that 21-98% of the tetracyclines and 0-68% of the fluoroquinolones present in a separated liquid pig manure fraction could be retained in struvite, leading to tetracycline concentrations in the granules that range from 0.2 µg g^{-1} to 2.0 µg g^{-1}. It was indicated that the tetracycline entrapment in the precipitated phosphate salt was linearly correlated to the total organic carbon content in the salt (R^2 = 0.72, n = 15) because tetracyclines have a high affinity for organic matter through cation bridging and cation exchange (Luo et al., 2011). Hence, during solid-liquid separation, the overall share of the tetracyclines present in the unprocessed manure will adsorb to the solid manure fractions that are rich in organic C (Wallace et al., 2018). Therefore, the maximal total retention of the antibiotics in precipitated phosphate salts will be reduced as, by definition, precipitation is a process that forms solid materials from dissolved substances present in the liquid fraction. Nonetheless, in order to avoid the entrapment of tetracyclines adhered to soil organic matter, it is recommendable to limit the organic C content in the precipitate (see ‘total organic carbon’ below). This holds particularly true as common pretreatments and hygienisation steps such as mesophilic anaerobic digestion only result in a limited removal of these veterinary antibiotics (Massé et al., 2014; Montes et al., 2015; Liu et al., 2018; Wallace et al., 2018).

- Ronteltap et al. (2007) reported that common **pharmaceutical compounds present in sewage and municipal wastewaters** (e.g. propranolol, ibuprofen, diclofenac and carbamazepine) transfer into the precipitated materials in only very small quantities, i.e. at rates ranging from 0.01% (diclofenac) to 2.6% (propranolol) in the precipitated phosphate salt versus their amounts in urine. Escher et al. (2006) found that less than 1-4% of the spiked hormones and pharmaceutical compounds in the urine feedstock were present in struvite. This was a better removal performance than could be achieved via alternative approaches such as bioreactor treatment, nanofiltration, ozonation or UV. Similarly, Ye et al. (2017) reported that 0.3-0.5% of the tetrazines from spiked solutions were found in struvites. Kemacheevakul et al. (2012) also found traces of some pharmaceutical products (tetracycline, erytromycine and norfloxacine, and other spiked compounds were not traced back in the end material) that were supplied as spikes to artificial urines, but here also the accumulation was negligible. In the study of STOWA (2015), metopropol was found in detectable concentrations in one out of the four struvites (only in an unwashed sample with an organic C content of 3.7%), but not in the remaining samples which were lower in organic C. Butkovskyi (2016) investigated the transfer of pharmaceutical compounds (diclofenac, naproxen, ibuprofen, metformin, and paracetamol), biocides (triclosan, benzalkonium chloride, 2,4-dichlorophenol), fragrance (galaxolide or HHCB), and parabens in struvites. It was indicated that only 2,4-dichlorophenol was found in minimal concentrations (0.5 µg g^{-1}) in the precipitated phosphate salt. The STRUBIAS subgroup also provided analyses of a wide range of pharmaceutical compounds for struvites from digested sludges; it was found that the concentration of two compounds (carbamazepine and carvedilol) was minimally elevated above detection limits, but that the precipitation processes reduced the concentrations of all other compounds investigated to below detectable levels. **It is concluded that pharmaceutical compounds can possibly accumulate in precipitated phosphate salts, but that the concentrations measured in relatively pure precipitated phosphate salts are low. Therefore, no major risk in terms of safety of recovered struvite from eligible input materials has been identified for material handling, the environment or the food chain.** This conclusion is in agreement with the risk assessment performed by de Boer et al. (2018) who indicated that the risk to human health from eating crops that were fertilised with struvites of high purity is insignificant. The average person would have to eat approximately 750 kg of dry food per day to reach the maximum acceptable daily intake limit, defined as the amount of a specific foodstuff that can be ingested (orally) on a daily basis over a lifetime without an appreciable health risk. It is noted that the overall share of the studies that assessed the risk originating from pharmaceutical compounds present in sewage have used precipitated phosphate salts of a relatively high purity and low organic carbon content, mostly originating from already operating struvite reactors.

In general, data **indicate that precipitated phosphate salts are generally safe with respect to organic contaminants**. The safe use of precipitated phosphate salts has also been indicated in a **bioassay** that assessed ecotoxicity for plants and aquatic organisms after the application of recovered struvite (ADEME - Naskeo Rittmo Timab, 2016). Theoretical and experimental evidence indicates that the **organic C level of the phosphate salts could be a critical factor to control the possible transfer of pollutants** from the input material to the fertilising products.

Total organic carbon

The section above indicated that no unacceptable risks are present for particular organic pollutants in precipitated phosphate salts of high purity that were targeted for this CMC. However, **it should be noted that this assessment was almost entirely based on precipitated phosphate salts of a high purity**, as indicated by XRD measurements, nutrient contents (P, Mg, N; and their ratios), or organic carbon contents. Therefore, **the conclusion of reduced risks associated with organic chemical contaminants is only demonstrated for high-quality precipitated phosphate salts, and no conclusions can be drawn for**

materials of a lower quality. Evaluating environmental and human health risks and other agronomic aspects for P-rich materials that are not produced through a precipitation process as outlined in Section 5.3.3.2 falls beyond the scope of the project mandate.

Different proxies can be used to characterise the purity of precipitated phosphate salts, with nutrient contents (P, N, Mg, and their relative ratios) and crystallographic measurements (XRD and other) being the most common. Nonetheless, these proxies are less suitable for defining purity because (i) the different precipitated phosphate salts that are encompassed in this CMC (e.g. struvite, K-struvite, dittmarite, dicalciumphosphate, anhydrous calcium hydrogenorthophosphate) vary widely in their nutrient content and elemental ratios, and (ii) crystallographic measurements are expensive. **Therefore, it is proposed to use organic carbon as a proxy that is inversely related to material purity.** The organic carbon content of the precipitated phosphate salt is a basic indication of the level of organic contamination and purity. The salts can contain both natural and synthetic organic matter. Examples of natural organic matter include undecomposed organic matter and pathogenic bacteria, whereas synthetic organic matter includes contaminants like pesticides, antibiotics and detergents. **Some functional groups in dissolved organic matter (e.g. carboxyl, hydroxyl, carbonyl) facilitate electrostatic association with contaminants and enhance migration by co-transport** (Polubesova et al., 2006). As a matter of fact, the specific surface of organic matter is about one to two orders of magnitude higher than for minerals (Horowitz, 1991), providing an adsorptive surface for natural and synthetic contaminants. Thus, organic matter originating from possibly contaminated input materials like sewage sludge and manure slurries can be the vehicle for the transportation of a variety of organic pollutants and biological pathogens in precipitated phosphate salts. Organic matter might thus not only contain contaminants that were present in the input material (e.g. heat-resistant plant pathogens), it is also often a vector for the selective adsorption of synthetic contaminants that were present in the liquefied matrix from which the precipitated phosphate salt was precipitated. It has been indicated that the organic C content is inversely related to some specific pharmaceutical compounds such as tetracycline (Lou et al., 2018; Ye et al., 2018), a major organic chemical contaminant of interest.

Manure and sewage sludge shows an organic C content of around 30%, and a P content of 1-3%, expressed on a dry matter basis. In order to achieve an improvement in contaminant levels of one order of magnitude for compounds that are adsorbed to organic matter, **it is proposed to limit the organic C content in precipitated phosphate salts to 3% of the dry matter content.** Additionally, the risk is also considerably lower because phosphate salts are concentrated in nutrients, resulting in lower field application rates compared to the sewage sludge. The STOWA study indicated that PAHs (PAH_{10}: 9.5 mg kg^{-1} dry matter), pharmaceutical compounds (metoprolol, 0.4 mg kg^{-1}) as well as spore-forming bacteria (spores of sulphite-reducing clostridia: 4.5-860 colony-forming units g^{-1} struvite) were, for instance, present in ‘struvites’ with an organic C content of 3.7% derived from digested sludge, but not in struvites with an organic C content below 1%. The proposed maximum organic C level of 3% for precipitated phosphate salts should thus further result in a major reduction of the risk of organic contaminants relative to the most contaminated input material on the eligible input material list.

Based on the documented values for organic C, it is believed that the 3% organic C limit is an achievable target for salts that are produced through the precipitation of phosphates from manure and municipal wastewaters. When materials have an organic C content > 3%, these organic compounds are often present as larger recognisable organic fractions (e.g. twigs, seeds; see STOWA, 2015) which can be easily removed via a material washing procedure (STOWA, 2015). It is noted that some P-recovery processes might recover P that is organically bound, but such processes fall beyond the scope of this CMC which focuses on the precipitation of dissolved phosphate ions from solution. Hence, **the overall share of the operational P-recovery facilities meet the proposed limit value of 3% for organic C and techniques are available to achieve the proposed limits**.

In addition, the limit value of 3% for organic carbon should provide the following benefits:

- **Minimal compliance costs and administrative burdens for operators** in the context of the conformity assessment procedures in the EU Fertilising Products Regulation (EU) 2019/1009. The **analytical procedures to trace** and quantify individual natural (e.g. plant pests and heat-resistant pathogens) and synthetic (e.g. pharmaceutical compounds) organic contaminants **are complicated and expensive,** with **costs** typically largely exceeding those for the determination of inorganic metals and metalloids (Langenkamp and Part, 2001). Although the relationship between organic C content and the abundance of contaminants is based on a limited dataset, specific contaminants have only been found at levels of concern for precipitated phosphate salts with an organic C content > 3%. Setting a limit value of 3% for organic C could enable a testing regime with a minimum of parameters, thus avoiding costly measurements of inorganic and organic compounds (metals and metalloids that are not regulated at PFC level, pharmaceutical compounds and personal care products, pesticides, plant protection chemicals and their decay products, heat-resistant pathogens, agronomic efficacy parameters, PCDD/F, dl-PCB, etc.). Setting a higher maximum limit for organic C would, conversely, be associated with complex and costly conformity assessment procedures as well as with further research and time delays required to derive safe limit values and to establish measurement *Standards* for the broad range of contaminants.

- **Market confidence and acceptance** is a critical aspect for fertilisers derived from secondary raw materials. The majority of the literature information that shows the agronomic efficacy and the product safety for precipitated phosphate salts is based on materials of high quality and low organic matter content. Moreover, the public, the media and governments are increasingly concerned about the presence of a broad spectrum of emerging organic pollutants of biogenic origin in consumer products and the food chain (Petrie et al., 2015). For many of such compounds, solid risk assessments are lacking to assess the validity of these concerns in precipitated phosphate salts. Setting a limit on organic C may further help support the proposed inclusion of fertilisers derived from secondary materials as CMCs in the EU Fertilising Products Regulation ((EU) 2019/1009) and their uptake by farmers and the broader public.

Experts from the STRUBIAS subgroup proposed the use of CEN - EN 15936 ‘Sludge, treated bio-waste, soil and waste - Determination of total organic carbon (TOC) by dry combustion’ as the preferred method for the determination of the organic C content in precipitated phosphate salts (see also Section 5.7.3). It is noted, however, that the testing and possible development of new *Harmonised Standards* falls beyond the scope of the JRC mandate.

5.3.5.2 Biological pathogens

Examples of pathogens that could be present in eligible input materials, especially manure and municipal wastewaters, include **bacteria** (e.g. *Salmonella, Legionella, Shigella, Clostridium, Vibrio cholera, Campylobacter, E. coli*), **fungi and plant pests, and viruses** (e.g. Hepatitis A and E virus, norovirus, rotavirus, enterovirus, reovirus, astrovirus, calicivirus), **protozoa** (e.g. *Cryptosporidium, Giardia, Entamoeba, Toxoplasma gondii*) and **worm eggs** (e.g. *Ascaris, Toxocara*) that can cause a broad array of animal, plant and human diseases. Although **specific EU legislation** (EU plant health legislation (2000/29/EC (actual) and 2016/2031/EC (replacing 2000/29/EC as of 15 December 2019)) has been put in place to prevent the introduction and spread of harmful and relevant organisms (e.g. *Synchytrium endobioticum*) within the entire Union territory; specific quality control measures for this CMC may further control for risks associated with the possible presence of biological pathogens.

The proposed 3% organic C limit will promote the precipitation of salts of a high purity, and as such reduce the risk of a possible transfer of biological pathogens to the precipitate. Moreover, during the production process of precipitated phosphate salts, some – but not all - pathogens are killed during drying or precipitation at moderately high pH. **Especially for anaerobic spore-forming bacteria and parasitic nematodes, the elimination could be incomplete resulting in their accumulation in the precipitated phosphate salts** (Decrey et al., 2011; STOWA, 2015; Ehlert et al., 2016a). To a minor extent, viruses and other biological agents could also be transferred to the precipitated phosphate salt, but become rapidly deactivated as the precipitated phosphate salt is dried (Decrey et al., 2011). Therefore, the presence of viruses is not expected to be an issue of concern on condition that the end material is dried after its production. Also, the presence of organic toxins in precipitated phosphate salts should be far below levels of concern for human and environmental health protection (Gell et al., 2011).

The adequate elimination of biological pathogens from precipitated phosphate salts is an important measure of the risk of further developing **antimicrobial resistance in the agro-food chain**. Antibiotic resistance can disseminate readily among microbial populations through horizontal gene transfer facilitated by the mobile genetic elements in antibiotic-resistant bacteria, which can compromise the efficacy of antibiotics in animal and human medicine. In addition to the release of antibiotic residues in the environment (discussed in Section 5.3.5.1), antimicrobial resistance is also directly affected by the development of **antibiotic-resistant bacteria** in the intestines of animals and humans, which end up in the excreta and eventually in the environment through the application of materials derived from biogenic wastes (Looft et al., 2012). Amongst others, *Clostridium* spp. might be resistant to multiple antimicrobial agents, and thus contribute to antimicrobial resistance in the food chain (Frieri et al., 2017). The application of low-quality struvite derived from pig manure (no hygienisation steps applied, with an unknown presence of biological pathogens present in the struvite; no data on antibiotic use in the livestock production or antibiotic concentration in the manure) has been shown to increase both the abundance and diversity of antibiotic-resistant bacteria in soil and phyllosphere (Chen et al., 2017).

The presence of biological pathogens is also dependent on the **pre-processing techniques that are applied for the hygienisation of the eligible input materials** (Wallace et al., 2018). Contrasting evidence exists on the capability of mesophilic anaerobic digestion (36 °C or 42 °C) (Bagge et al., 2005; Xu et al., 2015) to remove pathogens, but methods that apply (thermophilic) anaerobic digestion after pasteurisation pretreatment and pressure sterilisation techniques result in a significant decrease or effective removal of spore-forming bacteria and *Ascaris* eggs (Sahlstrom et al., 2008; Bagge et al., 2010; Fröschle et al., 2015). Nonetheless, many different production routes for precipitated phosphate salts do not apply such a hygienisation step.

Hence, for reasons of environmental and human protection, it is important to restrict the presence of biological pathogens in precipitated phosphate salts, and to provide as such substantial improvements relative to the landspreading of unprocessed manure and sewage sludge for sustainable agriculture, routes that are well known for their contribution to pathogen spreading and antimicrobial resistance (Udikovic-Kolic et al., 2014; Chen et al., 2016; Singer et al., 2016). The requirements for the hygienisation of animal-derived materials are laid down in the **Animal By-Product Regulation ((EC) 1069/2009)**. The placing on the market of **processed manure, products derived from processed manure and guano from bats is subject to the requirements laid down in Annex XI (Chapter I, Section 2) of Regulation (EU) 142/2011**. The standard processing method that such materials must undergo includes a heat treatment process of at least 70 °C for at least 60 minutes and they shall have been subjected to reduction in spore-forming bacteria and toxin formation, where they are identified as a relevant hazard. Nonetheless, the competent authority may authorise the use of other standardised process parameters besides those referred to above, provided that such parameters ensure the minimisation of biological risks. This involves, amongst others, the identification and analysis of possible hazards, a validation of the intended process by measuring the reduction of viability/infectivity of endogenous indicator organisms, including, for instance, *Enterococcus faecalis*, thermoresistant viruses such as parvovirus, parasites such as eggs of *Ascaris* sp., *Escherichia coli*, *Enterococcaceae*, and *Salmonella* spp. National legislation on biological pathogens in fertilising materials varies across EU Member States, with **some EU countries having strict limits on pathogens (e.g. France) and others**

having a more generic description on the need to restrict human health effects related to fertiliser management practices (e.g. the Netherlands).

An approach is proposed for the microbial testing of precipitated phosphate salts that is dependent on the input material applied. **Standard microbial testing is proposed to involve *Salmonella* spp. and *Escherichia coli* or *Enterococcaceae,* but including additional requirements on spore-forming bacteria (*Clostridium perfringens* as an indicator organism) and *Ascaris* eggs when manure or municipal wastewater are used as input material for the production process**. Moreover, it is proposed that microbial **testing is not required when certain pretreatments (conditions for anaerobic digestion as specified in Annex V to Regulation (EU) No 142/2011, pressure sterilisation, thermal hydrolysis, etc.) are applied that result in the hygienisation of the precipitated phosphate salt**. The limit values for the different biological pathogens are proposed to be in line with the values as laid down in the French legislation (Norme NFU 44-095), as follows:

Regardless of the input material applied, the precipitated phosphate salt shall meet all of the following requirements:

a) … [not related to biological pathogens];
b) …[not related to biological pathogens];
c) … [not related to biological pathogens];
d) … [not related to biological pathogens];
e) no presence of *Salmonella* spp. in a 25 g sample; and
f) no presence of *Escherichia coli* or *Enterococcaceae* in a concentration of more than 1 000 CFU/g fresh mass.

Precipitated phosphate salts derived from materials listed under points a) and b) [i.e. municipal wastewaters and manure] shall meet the following requirements:

g) … [not related to biological pathogens];
h) no presence of *Clostridium perfringens* in a concentration of more than 100 CFU/g fresh mass; and
i) no presence of viable *Ascaris* sp. eggs in a 25 g fresh mass.

By way of derogation from points e), f), h) and i), testing shall not be necessary for precipitated phosphate salts that have undergone either of the following conditions:

i. Pressure sterilisation through heating to a core temperature of more than 133 °C for at least 20 minutes without interruption at a pressure (absolute) of at least 3 bars. The pressure must be produced by the evacuation of all air in the sterilisation chamber and the replacement of the air by steam (‘saturated steam’).
ii. Processing in a pasteurisation/hygienisation unit that reaches a temperature of 70 °C during a time of at least 1 hour.

The derogation shall extend to precipitated phosphate salts that are exclusively derived from eligible input materials that have undergone either of the conditions (i) or (ii).

Related to animal by-products, the following proposals are put forward for further consideration and evaluation by the Commission and other bodies as a possible end-point in the manufacturing chain in accordance with the third subparagraph of Article 5(2) of Regulation (EC) No 1069/2009:

1. Precipitated phosphate salts & derivates that have been derived from manure, non-mineralised guano, and digestive tract content pursuant to Regulation (EC) No 1069/2009 provided that they:
    - are compliant with the proposed criteria for CMC XX,
    - have no presence of *Clostridium perfringens* in a concentration of more than 100 CFU/g fresh mass, and
    - have no presence of *Ascaris* sp. eggs in a 25 g fresh mass.

2. Precipitated phosphate salts & derivates that have been derived from animal by-products and derived materials from category 2 or category 3 material as defined by Regulation (EC) No 1069/2009 provided that they:
    - are compliant with the proposed criteria for CMC XX,
    - have been hygienised in accordance with the conditions for pressure sterilisation or with other conditions to prevent risks arising to public and animal health, in accordance with the requirements laid down pursuant to Article 15 of Regulation (EC) No 1069/2009, or
    - they are digestion residues from transformation into biogas as set out in Annex V to Regulation (EU) No 142/2011.

5.3.5.3 Metals and metalloids

Metals and metalloids (semimetals) have been associated with contamination and potential toxicity or ecotoxicity. The group includes essential microelements that are required for the complete life cycle of an organism, but the establishment of **safe environmental levels** must consider the intake-response relations for both deficiency and toxicity. The degree of toxicity of metals and semimetals varies greatly from element to element and from organism to organism and depends on its concentration in soil, plant, tissue, ground water, etc.

Data on inorganic metals and metalloids (As, Cd, Cd, Cu, Cr (VI), Hg, Ni, Pb, and Zn) are mainly available for struvites and Ca phosphates obtained from municipal wastewaters, but information was also collected for manure, separately collected urine, and livestock stable slurries and particular food processing industries (potato industry and dairy industry) (Section 15.1.2). Nevertheless, municipal wastewaters are the input material that is most enriched in inorganic metals and metalloids (Eriksson, 2001). Materials from certain food processing industries (Gendebien et al., 2001) and (digestates) of vegetable waste from agriculture and forestry (Valeur, 2011; Al Seadi and Lukehurst, 2012; Ehlert et al., 2016b) contain significantly lower amounts of inorganic metals and metalloids.

As indicated in Section 15.1.2, **precipitated phosphate salts show low levels of inorganic metals and metalloids**, both for P-salts that have been derived from municipal wastewaters (precipitated from sludge liquor and digested sludge), manure, and other eligible input materials. Also for Ca phosphates with a low organic C content, confidential information confirms that inorganic metals and metalloids in precipitated phosphate salts are not a major issue of concern. This is in conformity with the mechanism of precipitation that involves the formation of a separable solid substance from a *solution* by converting the substance into an insoluble form through the addition of chemicals. Metals are mostly associated with organic matter in sludges (Karvelas et al., 2003), which is why the proposed maximum limit of 3% for organic C may help to limit the metal content in the precipitates.

Given that the metal/metalloid levels are **generally one to two orders of magnitude lower than the limits for inorganic contaminants at PFC level** for inorganic macronutrient fertilisers in the EU Fertilising Products Regulation ((EU) 2019/1009) (Cd: 60 mg kg^{-1} P_2O_5; Cr (VI): 2 mg kg^{-1}; Ni: 100 mg kg^{-1}; Pb: 120 mg kg^{-1}; As: 40 mg kg^{-1}), it is not proposed to add **any specific limits for inorganic metals and metalloids, independent of the input material applied.** Moreover, Zn and Cu are not an issue of concern as the reported concentrations are generally low. Cd contents (on average < 1.8 mg Cd kg^{-1} P_2O_5, with a maximum documented value of 3.7 mg Cd kg^{-1} P_2O_5) are about one to two orders of magnitude lower than those encountered in phosphate rock (20 mg to more than 200 mg per kg P_2O_5; Oosterhuis et al., 2000) and one order of magnitude lower than those of mined and synthetic P-fertilisers (Kratz et al., 2016).

5.3.5.4 Emissions

Emissions from the application of fertiliser are generally attributed to four different mechanisms during material handling and application (Midwest Research Institute, 1998): (1) reactions between the soil and the applied fertiliser generating increased gaseous emissions to air including NOx, N_2O, NH_3, and SO_2, (2) soil disturbance generating particulate matter emissions where soil particles and other materials in the soil become airborne, (3) volatilisation of the fertiliser immediately above and behind the application vehicle generating gaseous emissions (e.g. NH_3), and (4) particulate matter emissions from the fertiliser itself during handling or application.

The elements that underlie the mechanisms 1, 2 and 3 are complex and depend on a number of material properties, soil and climatic properties via complex relationships that have not been characterised quantitatively. Similar to mined and synthetic fertilisers, the best form of emission control identified for precipitated phosphate fertilisers to date **is through appropriate ‘nutrient management**’ (Midwest Research Institute, 1998). Here, nutrient management is defined as the form, placement, and timing of the fertiliser application relative to the crops’ need for fertiliser. Therefore, **it is proposed to support the general labelling requirements for all PFC materials in the EU Fertilising Products Regulation ((EU) 2019/1009)**, including instructions for intended use, intended application rate, timing and frequency, recommended storage conditions, and any other relevant information on measures recommended to manage risks.

The fourth mechanism may generate **airborne dust and particulate matter emissions**, which could penetrate into the pulmonary alveolar region of the lungs. Moreover, particulate emissions could arise during specific handling operations during fertiliser production processes (e.g. blending). Airborne dusts are of particular concern because they are known to be associated with classical widespread occupational lung diseases such as pneumoconioses, especially at higher levels of exposure. Industrial handling and application of precipitated phosphate salts could thus represent potential physical hazards with regards to health hazards related to the inhalation of dusts, as well as environmental pollution. The *European Standard* EN 15051 describes the measurement of the dustiness of a powder by using a rotating drum or continuous drop method. Nonetheless, the dustiness of a powder product, defined as the propensity of a material to generate airborne dust during its handling (Lidén, 2006), not only depends on the **intrinsic physical properties of the material but also on the handling scenario.**

The European Regulation (EC) No 1272/2008 on classification, labelling and packaging (CLP) of substances and mixtures aligns the European Union system of classification, labelling and packaging of substances and mixtures to the UN Globally Harmonized System of Classification and Labelling of Chemicals (GHS). The CLP Regulation requires manufacturers, importers, and downstream users to classify substances or mixtures according to the harmonised classification criteria for physical, health, or environmental hazards. CLP Articles 5, 6 and 8.6 clearly point out that available and new information on substances and mixtures shall relate to the form or physical state(s) in which the substance or mixture is placed on the market and in which it can reasonably be expected to be used. Furthermore, CLP Title V required that by 1 December 2010, substances that met the criteria for classification as hazardous according to the CLP Regulation or substances subject to registration under REACH (Regulation (EC) No 1907/2006 concerning the Registration, Evaluation, Authorisation and Restriction of Chemicals) must have been notified to the classification and labelling inventory of the European Chemicals Agency. Safety data sheets are effective and accepted tools to communicate the safety information of products in the supply chain (Pensis et al., 2014). In addition, the ECHA Guidance to the CLP Regulation published on 13 July 2009 mentions that 'for human health, different forms (e.g. particle sizes, coating) or physical states may result in different hazardous properties of a substance or mixture in use' and therefore they may be classified differently. Hence, **correct classification and labelling allows downstream users to assess the risk associated with airborne dust during the handling and application of the heterogeneous materials within the CMC precipitated phosphate salts, and to take the necessary measures to prevent any potential adverse impacts in the event that a risk is identified.**

Directive **2010/75/EU** of the European Parliament and of the Council on industrial emissions (the **Industrial Emissions Directive - IED**) is the main EU instrument regulating pollutant emissions from industrial installations. This Directive effectively controls for gaseous and particulate matter emissions to the environment during production processes of fertilising materials, among others. Moreover, **Directive 2008/50/EC** of the European Parliament and of the Council of 21 May 2008 on ambient air quality and cleaner air for Europe also establishes air quality objectives, including for fine particulate matter.

It is concluded that the review of the literature and comments from the STRUBIAS subgroup provided no information on specific measures required for precipitated phosphate salts to control for emissions. **It is indicated that the provisions in the EU Fertilising Products Regulation ((EU) 2019/1009) on labelling and existing EU legislation are sufficiently effective to control for any adverse impacts associated with emissions during the production, handling and application of precipitated phosphate salts.**

5.3.5.5 Occupational health

Council Directive 89/391/EEC on the introduction of measures to encourage improvements in the safety and health of workers at work seeks to adequately protect workers and encourages improvements in occupational health and safety in all sectors of activity, both public and private. The Directive also promotes workers' rights to make proposals relating to health and safety, to appeal to the competent authority and to stop work in the event of serious danger. No further legal requirements are therefore proposed.

5.3.6 Physico-chemical properties

5.3.6.1 Physical impurities

It has been demonstrated that washed struvites may contain physical impurities including gravels and organic matter such as seeds, twigs, etc. (STOWA, 2015).

Figure 2: Isolated impurities from (low-quality) struvite precipitate samples (adopted from STOWA, 2015 - © STOWA, 2015)

Given that these impurities are often vectors for the adsorption of contaminants (STOWA, 2015), it is proposed to **limit visually detectable physical impurities (e.g. recognisable organic materials, stones, glass and metals) greater than 2 mm to < 0.3% and the total**

macroscopic impurities to < 0.5%, in line with the provisions laid down for other CMCs in the EU Fertilising Products Regulation ((EU) 2019/1009).

5.3.6.2 Dry matter content

A high water content during the storage of precipitated phosphate salts may possibly induce biological re-contamination of precipitated phosphate salts prior to application on land. Measures on a minimum dry matter content to impede biological re-contamination were initially proposed in previous versions of this report, but the STRUBIAS experts indicated that such a criterion was obsolete due to (i) the proposed 3% organic C limit that will effectively reduce the presence of biological pathogens in the precipitate to low levels, and (ii) the fact that moist precipitated phosphate salts could be used as an intermediate in mineral P-fertiliser production processes. Therefore, **no criterion on dry matter content has been proposed in this document**.

5.3.6.3 pH

Precipitated phosphate salts typically have a neutral to slightly basic pH. Hence, pH shocks for the soil microorganisms and fauna are not expected, and no specific requirements are proposed for pH.

5.3.6.4 Granulometry

Agglomeration is used as a means of improving product characteristics and enhancing process conditions. In addition to these benefits, agglomeration also solves a number of problems associated with material particle sizes:

- significant dust reduction/elimination and mitigation of product loss;
- improved handling and transportation;
- improved application and use;
- increased water infiltration in the soil as there is no risk of the blocking of soil pores by small water-insoluble particles.

It is noted that the particle form (granule, pellet, powder or prill) of the product shall be indicated on the **label** of solid inorganic macronutrient fertilisers (see labelling requirements in the EU Fertilising Products Regulation ((EU) 2019/1009)). According to the STRUBIAS subgroup, it is not considered relevant to set a criterion on granulometry or particle size distribution, and hence it is proposed to leave this aspect up to the market.

5.3.7 Handling and storage

The storage of hydrated precipitated phosphate salts struvite and hydrated dicalcium phosphates at high temperatures can cause the gradual loss of ammonia and water molecules, ultimately transforming the precipitated phosphate salt into different mineral phases (e.g. amorphous magnesium hydrogen phosphate). The storage of precipitated phosphate salts under dry conditions promotes the inactivation or removal of bacterial and viral pathogens, and prevents possible re-contamination (Bischel et al., 2015). **It is proposed that physical contact between input and output materials shall be avoided in the production plant after the precipitation process, and that finished precipitated phosphate salts shall be stored in dry conditions**.

5.4 Thermal oxidation materials & derivates

5.4.1 Scope delimitation and possible uses

This CMC comprises materials that have **undergone thermal oxidation in a non-oxygen-limiting environment**, as well as derivates that are (partially) manufactured from those materials (see Section 5.1.2). Hence, the CMC includes both **ashes and slags** as collected from the combustion reactor, as well as materials with a different chemical composition derived from those ('**derivates**'; e.g. triple superphosphate derived from sewage sludge ashes). Section 5.2.4 gives a detailed explanation of the technical provisions for dealing with such 'precursors' or 'intermediates' in the framework of the EU Fertilising Products Regulation ((EU) 2019/1009) and this CMC.

Thermal oxidation materials & derivates may have a variety of applications as fertilising products in agriculture and forestry (Insam and Knapp, 2011; Vassilev et al., 2013a). Primarily, they can be used as ingredients in products that are intended for use as a fertiliser or a liming material as follows:

- Ashes and slags resulting from the combustion or melting of solid biomass can contain valuable plant macronutrients such as K, P, S, Ca and Mg, with most of them in relatively soluble forms (Vesterinen, 2003; Obernberger and Supancic, 2009; Haraldsen et al., 2011; Insam and Knapp, 2011; Brod et al., 2012) (see Section 5.4.4.1). Ca, Mg and K contents are usually present in the form of oxides, hydroxides, carbonates and silicates, associated with basic properties; therefore, some ashes can serve as liming agent (Demeyer et al., 2001; Saarsalmi et al., 2010; Ochecova et al., 2014). Phosphorus occurs as phosphates of Ca, K, Fe and Al (Tan and Lagerkvist, 2011), and certain thermal oxidation materials (e.g. poultry litter ash) can have P-contents that are equivalent to those of straight macronutrient P-fertilisers. Hence, thermal oxidation materials & derivates may serve **as a component material for the production of solid macronutrient inorganic fertilisers and organo-mineral fertilisers**.

- When ash comes into contact with soil water, the pH of the soil solution increases as the oxides and hydroxides in the ash dissolve and hydroxide ions are formed. Thus, the ash has **a liming effect** when added to the soil as an amendment and can be used to neutralise acidity. The chemical constituents that determine the liming effect are essentially the same as for lime. However, ash is a more complex chemical mixture and the liming effect is lower than for lime products when expressed per unit weight (Karltun et al., 2008). The ash that comes directly from the thermal oxidation process is not chemically stable in the presence of moisture and CO_2 from the atmosphere. The oxides in the ash react with water and CO_2 and form hydroxides and carbonates. During this process, the ash increases in weight (Karltun et al., 2008). As outlined by the STRUBIAS subgroup, there is a clear need to label the liming equivalence of ashes as negative effects on productivity may also arise when the liming effect on soil pH is greater than the normal acidification of agricultural soils. Therefore, it is proposed that the neutralising value shall be labelled on PFC products when it is > 15 (equivalent CaO) or 9 (equivalent HO-).

- Any fertilising product that has a minimum macronutrient content should be marketed as a macronutrient fertiliser in the EU Fertilising Products Regulation ((EU) 2019/1009), independent of the quantity of micronutrients present in the fertiliser. Considering the content of macronutrients (N, P, K, Mg, Ca, S, Na; as defined in the EU Fertilising Products Regulation ((EU) 2019/1009)) in thermal oxidation materials & derivates (Section 15.2.1), a possible entry in the EU Fertilising Products Regulation ((EU) 2019/1009) **for thermal oxidation materials & derivates as micronutrient fertilisers is unlikely**.

- Some studies have indicated the potential of ashes, often coal ashes with a low content of plant-available nutrients, to improve the physical properties of the soil, including bulk density, porosity and water-holding capacity, and/or to cause a shift in soil texture classes (Jala and Goyal, 2006; Basu et al., 2009; Pandey and Singh, 2010; Blissett and Rowson, 2012; Yao et al., 2015). Therefore, ashes are sometimes promoted as an inorganic soil improver. Nevertheless, beneficial increases in physical soil properties are only observed in applications of large ash quantities (often 5-20% or a higher weight percent of the receiving soil; application rates of 70-500 tonnes ha^{-1}) (Chang et al., 1977; Buck et al., 1990; Khan et al., 1996; Prabakar et al., 2004). Such application rates are associated with a huge environmental footprint for transport, and a substantial dilution of nutrients in the receiving soil when nutrient-poor ashes are applied. Moreover, laboratory incubation studies found that addition of fly ash to sandy soils has a variable impact upon soil biota, with some studies documenting a severe inhibition of microbial respiration, enzyme activity and soil nitrogen cycling processes such as nitrification and N mineralisation (Jala and Goyal, 2006). The STRUBIAS subgroup indicated that **no market or demand exists for the use of thermal oxidation materials, such as coal ash, that exclusively target soil-improving functions in the EU Fertilising Products Regulation ((EU) 2019/1009)**. Instead, any soil-improving function of thermal oxidation materials is perceived as a potential **side-benefit** of those ashes acting as a macronutrient fertiliser or a liming agent. Therefore, the recovery rules shall include the necessary provisions to ensure that the EU Fertilising Products Regulation ((EU) 2019/1009) shall not be used to enable a CE status for waste materials and by-products that have no added value for agriculture (e.g. ashes from fossil fuel combustion) (see Section 5.4.2 on input materials).

Note that the section above on possible uses is only informative to introduce thermal oxidation materials & derivates in view of their possible intended uses and associated application rates. The proposed framework of the EU Fertilising Products Regulation ((EU) 2019/1009) however, in principle, allows all CMCs to be used in all PFC categories.

For thermal oxidation materials & derivates, **national legislation** related to the material properties and their use exists in different EU Member States. Moreover, national End-of-Waste protocols have been described for certain thermal oxidation materials (e.g. UK poultry litter ash quality protocol). These initiatives mostly focus on inorganic metals and metalloids (e.g. Cd, Hg) and persistent organic pollutants (e.g. PAHs, PCDD/Fs), which is why a clear reference to these initiatives will be made in the respective sections of this document.

5.4.2 Input materials and reactants

5.4.2.1 Targeted input materials

a) Waste incineration at 850 °C for > 2 seconds is generally considered an effective technique to remove biological pathogens and volatile pollutants from non-hazardous waste streams, which is why in principle a wide-ranging list of waste input materials is acceptable. Therefore, the proposal is to include **waste and (industrial) by-products** within the meaning of Directive 2008/98/EC, with the following input materials being **excluded**:

   i. <u>*Waste classified as hazardous*</u> according to Annex III to Directive 2008/98/EC (Waste Framework Directive). This exclusion is justified as (1) all non-hazardous substances of the European List of Waste cover the most relevant input materials that can be used for nutrient recovery in a techno-economically feasible manner, and (2) some residues from hazardous waste could still be associated with risks after transformation in ashes and slags.

   ii. <u>*Mixed municipal waste*</u>. The residual ash fraction after incineration of this type of waste should normally have a total organic C content of < 3%, but can potentially contain high concentrations of hazardous residues originating from the input waste (Zhang et al., 2004). Occurrences of hazardous chemicals such as herbicides, dioxins and furans and their decay compounds in leachate from ashes disposed at municipal waste landfills have been reported (Priester et al., 1996; Römbke et al., 2009). Moreover, the nutrient content of mixed municipal solid waste is relatively low (Section 15.2.1).

In addition to the waste materials, the following input materials are also proposed for inclusion:

- **Living or dead organisms or parts thereof**, which are unprocessed or processed only by manual, mechanical or gravitational means, by dissolution in water, by flotation, by extraction with water, by steam distillation or by heating solely to remove water, or which are extracted from air by any means, except:
    - materials originating from mixed municipal waste,
    - sewage sludge, industrial sludge or dredging sludge,
    - animal by-products or derived products falling within the scope of Regulation (EC) No 1069/2009 for which no end point in the manufacturing chain has been determined in accordance with the third paragraph of Article 5(2) of that Regulation,

    - materials separately listed under other points.

- **Animal by-products of categories 2 and 3** pursuant to Regulation (EC) No 1069/2009 (Animal By-Products Regulation). The main objective of this Regulation is to control for animal health, including the possible transmission of prion diseases (Paisley and Hostrup-Pedersen, 2005a; Saunders et al., 2008). Prion diseases, or transmissible spongiform encephalopathies, are fatal neurodegenerative diseases impacting a number of mammalian species, including cattle (bovine spongiform encephalopathy, BSE or 'mad cow' disease), sheep and goats (scrapie), deer, elk and moose (chronic wasting disease) and humans (Creutzfeldt-Jakob disease, and others). Scrapie and chronic wasting disease are of particular environmental concern as they are horizontally transmissible and remain infectious after years in the environment. It is likely that the environment serves as a stable reservoir of infectious chronic wasting disease and scrapie prions (Saunders et al., 2008). Johnson and colleagues have shown that prions bound to soil minerals are more infectious than unbound prions (Johnson et al., 2007). In addition, the disposal of carcasses during BSE outbreaks, both in past and potential future disposal events, serves as another environmental source of prions with the potential to infect humans. Therefore, the possible presence of prions, the main contaminant of interest that distinguishes category 1 from category 2 and 3 animal by-products, poses a significant environmental concern (Saunders et al., 2008). Prions present in category 1 material are associated with the highest risk from a human and animal health perspective for thermal oxidation materials & derivates; this type of contaminants shows the highest resistance against thermal degradation and adsorbs in an irreversible manner to soil particles (Saunders et al., 2008). This report has not assessed the risks resulting from the use of incinerated category 1 animal by-product material (see Section 5.2.6). The **prion infectivity risks associated with the use of incinerated category 1 animal by-products for animals are about two orders of magnitude higher than the infectivity risk for humans through the fertiliser-soil-food exposure pathway** (Paisley and Hostrup-Pedersen, 2005b), implying that possible measures to control for animal health will effectively control for human health issues. Other possible contaminants that may be exclusively present in category 1 materials, but not in category 2 and 3 animal by-products, including those listed in Group B(3) of Annex I to Directive 96/23/EC will be effectively removed during the combustion process (e.g. mycotoxins (De Saeger et al., 2016), dl-PCBs (see Section 5.4.5.2).
- **Bio-waste** within the meaning of Directive (EU) 2018/851 amending Directive 2008/98/EC resulting from separate bio-waste collection at source.
- **Residues from the composting, anaerobic digestion, pyrolysis or gasification of living and dead organisms, bio-waste, and animal by-products** as listed above.
- **Auxiliary fuels** (natural gas, liquefied petroleum gas, natural gas condensate, process gases and components thereof, crude oil, coal, coke as well as their derived materials), when used in incineration, co-incineration or biomass combustion plants to process input materials listed above.

- **Substances which occur in nature which are used in production processes of the iron and steel industry**.

Moreover, the **addition of combustion additives and reactive agents that are required for thermo-chemical conversion processes** that aim at the production of higher-quality ashes and slags is **permitted as outlined in Sections 5.4.3.2 and 5.4.3.3**.

The proposed list of eligible input materials takes into consideration the following elements indicated by the STRUBIAS subgroup:

- Incineration is a well-demonstrated technique to remove many organic and organo-chemical pollutants from waste-based materials. Therefore, the **CMC 'thermal oxidation materials & derivates' offers unique possibilities for the production of fertilising materials from a broad range of biogenic and industrial waste streams**.
- **All major streams that contain dissipated P should preferentially be included** as eligible input material for this CMC as most of these streams are also C-rich and can be combusted.
- Also, the technical proposals should include ashes that contain plant resources, other than P (e.g. Ca, Mg, micronutrients, etc.). Often these materials are derived from biomass and specific industries (e.g. pulp and paper industry).
- Nonetheless, there is a need **to exclude certain input materials** that might introduce a risk of the presence of additional contaminants in the ashes and slags, and that **could lead to more complex compliance schemes** (e.g. mixed municipal waste fractions separated through mechanical, physico-chemical, biological and/or manual treatment, hazardous chemical wastes, etc.);
- **Fossil fuels are sometimes used in small quantities** (e.g. during start-up, after maintenance, as combustion additives, as a reducing agent) **in waste (co-)incineration plants and biomass combustion plants**. At the same time, the basis of support for using the EU Fertiliser Regulation as a route to provide a CE status to **residues from fossil-fuel-based thermal plants is lacking,** even if minimum quantities of biomass or sewage sludges are used as an input material. Therefore, reference is made to the auxiliary nature of the fossil fuels, and to the type of plants where fossil fuels can be used (i.e. incineration, co-incineration or biomass combustion plants).
- **Steel and iron slags** are produced through a thermal oxidation process, so industrial by-products from the sector could be classified as thermal oxidation materials & derivates. Moreover, there are ongoing developments from the sector to further increase the quality of such slags for their exploitation in agriculture (e.g. adding P-rich input materials such as sewage sludge or animal bones in the reactors).

5.4.2.2 Reactants

Reactants are added in the production process of thermal oxidation materials & derivates (i) to facilitate the operational conditions of the combustion process in the case of **fuel additives**, (ii) to increase the **quality** of the resulting fertilising materials in **thermochemical processes**, **and** (iii) as part of **post-combustion manufacturing processes.**

Fuel additives

Some biomass fuels have high K contents, which react with other ash-forming elements (i.e. Cl, Si, P and S) and lead to different ash-related operational problems (Wang et al., 2012a). Biomass ash sintering causes different negative effects in the combustion plants: (a) formation of ash agglomerates that obstruct the air-biomass contact, which may cause an inhibition of the fluidisation in the fluidised bed equipment; (b) formation of sintered ash deposits in the heat exchangers resulting in a reduced heat exchange capacity, difficulty in cleaning the deposited ash and, occasionally, mechanical failure in the heat exchangers. The ash-related operational problems thus reduce the efficiency of the combustion systems, cause extra costs for boiler cleaning and maintenance, and hinder further utilisation of biomass materials as combustion fuels. Ash-related operational problems are especially severe during combustion of biomass fuels derived from the agricultural sector, contaminated waste materials and residues from bio-refinery and food processing plants. Utilisation of natural and chemical additives to abate these problems has been studied and tested for several decades. **Various additives can mitigate ash-related issues via the following mechanisms**: 1) capturing problematic ash species via chemical adsorption and reactions, 2) physical adsorption and removal of troublesome ash species from combustion facilities, 3) increasing the biomass ash melting temperature by enhancing inert elements/compounds in ash residues, and 4) limiting biomass ash sintering by diluting and pulverising the effects of the additives.

Additives are grouped according to the reactive compounds contained, including Al-silicate-based additives, sulphur-based additives, calcium-based additives, and phosphorus-based additives. Additives with strong chemical adsorption and reaction capacities can minimise K-related ash sintering, deposition and slagging during biomass combustion processes. As observed in Table 3, **most additives are natural materials and minerals** that are on the list of proposed permitted input materials (see Section 5.4.2). Also, chemicals such as ammonium sulphate, aluminium sulphate, iron sulphate, ammonium phosphate, phosphoric acid and DCP are listed (Table 3).

Table 3: List of common additives used during the combustion process (adopted from Wang et al., 2012a)

suspected effects	additives
Chemical adsorption and interaction	alloysite, cat litter, emathlite, clay minerals, clay sludge, illite, detergent zeolites, ammonia sulphate, aluminum sulphate, iron sulphate, ammonia phosphate, phosphoric acid, limestone, lime, marble sludge, sewage sludge, paper sludge, peat ash, coal fly ash, dolomite, bauxite, quartz, titanium oxide
Physical adsorption	kaolin, zeolite, halloysite, clay minerals, clay minerals, clay sludge, limestone, lime, sewage sludge, paper sludge, dolomite, calcined dolomite, bauxite, gibbsite
Restraining and powdering effects	lime, limestone

Reactants for thermochemical P-recovery processes to produce better quality fertilising materials

Some **thermochemical P-recovery approaches** rely on the addition of reactants to improve the quality of the resulting ashes and slags or the transformation of P-compounds into the gaseous phase. Section 14.2.2 gives a detailed description of such production processes in the planning, pilot or operational phase (RecoPhos, AshDec, Mephrec, EuPhoRe, etc.). These processes commonly use **alkaline and/or earth alkali salts, coke and chlorination agents** to reduce and volatilise compounds of interest for their subsequent removal (e.g. metals in AshDec process) or isolation (e.g. elemental P in thermo-reductive RecoPhos process). Considering the emerging nature of P-recovery through thermal processes and the evolving legal framework for fertilisers in the EU, it is not unlikely that more processes will develop in the near future. Therefore, the STRUBIAS subgroup indicated that the list of reactants should not be exhaustive and should remain as unrestricted as possible. This is in line with the provisions for CMC 1 (‘Virgin material substances and mixtures’) in the EU Fertilising Products Regulation ((EU) 2019/1009), where minimal provisions on the use of virgin materials as intermediates in production processes have also been defined.

Chemical reactants for derivate manufacturing processes

As outlined in Section 5.2.4, **‘two-step’** manufacturing processes enable the further processing of intermediate materials, such as incineration ashes. The principle of the two-step manufacturing processes for this CMC is that the thermal oxidation process removes specific contaminants (e.g. organic contaminants, biological pathogens), after which a subsequent chemical manufacturing step can be applied to produce a fertilising material of a preferred chemical composition and with a low metal content. In the proposals for the technical requirements, the thermal oxidation process should meet the conditions for waste incineration and will be tested for specific organic contaminants (e.g. PAHs, see Section 5.4.5.2). Ashes that meet these requirements will thus have low levels of organic contaminants.

Therefore, it is **proposed that such materials can be further processed with (chemical) intermediates to shape a high-quality material that can be incorporated into an EU fertiliser.** Specifically, it is proposed that the intermediate material obtained can be further processed using an extensive set of substances/mixtures, as long no new ‘risk materials’ are introduced in the process. Risk materials are, in this context, defined as materials which can introduce biological contamination or other unintentional organic or inorganic contaminants. Hence, it is proposed to enable only virgin materials (and possibly *safe* industrial by-products as permitted within the framework of the EU Fertilising Products Regulation ((EU) 2019/1009), see Section 4.2.2) for such post-incineration manufacturing steps, and **to exclude waste materials, materials which have ceased to be waste and substances formed from precursors which have ceased to be waste in one or more Member States, and animal by-products** (similar to the provisions for CMC 1). Because the process limitations on the second step of the processes are minimal, a high degree of **sovereignty is allowed for manufacturers to apply processes of their choice** and to promote the development of innovative processes that start from intermediate materials. The proposed provisions cover all wet chemical processes that involve the removal of P along with other elements from the ashes by elution, after which the dissolved elements are recovered by solidification, precipitation, ion exchange or membrane technologies. The elution process predominantly involves the use of strong acidic solvents, though, on occasion, alkaline substances have been used or a combination of the two. The list of solvents includes, amongst others, sulphuric acid (H_2SO_4), hydrochloric acid (HCl), nitric acid (HNO_3), phosphoric acid (H_3PO_4), oxalic acid ($H_2C_2O_4$), and sodium hydroxide (NaOH). Also, the addition of other substances, such as steam (used for instance in the EcoPhos process), will be permitted in the post-combustion manufacturing process. Finally, intermediates used to produce chemical fertiliser blends (e.g. NPK) will be permitted. **The end material from the whole manufacturing process (‘thermal oxidation materials & derivates that will be incorporated in the EU fertilising product’) will then be subject to further testing for inorganic contaminants** (see Section 5.4.5.1).

5.4.2.3 Proposals for input materials and reactants

Based on the information presented in Sections 5.4.2.1 and 5.4.2.2, the following proposal is put forward:

1. An EU fertilising product may contain thermal oxidation materials exclusively obtained through thermochemical conversion under non-oxygen-limiting conditions from one or more of the following input materials:

  l) living or dead organisms or parts thereof, which are unprocessed or processed only by manual, mechanical or gravitational means, by dissolution in water, by flotation, by extraction with water, by steam distillation or by heating solely to remove water, or which are extracted from air by any means, except:
    - materials originating from mixed municipal waste,
    - sewage sludge, industrial sludge or dredging sludge,
    - animal by-products or derived products falling within the scope of Regulation (EC) No 1069/2009, and
    - materials separately listed under points e) to j);

  m) vegetable waste from the food processing industry and fibrous vegetable waste from virgin pulp production and from production of paper from virgin pulp;

n) derived products referred to in Article 32 of Regulation (EC) No 1069/2009 for which an end-point in the manufacturing chain has been determined in accordance with the third subparagraph of Article 5(2) of that Regulation;

o) animal by-products, the products derived from which are referred to in Article 32 of Regulation (EC) No 1069/2009 and have an end-point in the manufacturing chain determined in accordance with the third subparagraph of Article 5(2) of that Regulation;

p) bio-waste within the meaning of Directive 2008/98/EC resulting from separate bio-waste collection at source;

q) residues from composting, anaerobic digestion, pyrolysis or gasification as a pretreatment technique of the input materials listed under point a) to d);

r) sewage sludge from municipal wastewater treatment plants;

s) waste within the meaning of Directive 2008/98/EC with the exception of:
    - those listed under points a) to g),
    - materials which display one or more of the hazardous properties listed in Annex III to Directive 2008/98/EC,
    - materials originating from mixed municipal waste, and
    - animal by-products or derived products within the scope of Regulation (EC) No 1069/2009.

t) auxiliary fuels (natural gas, liquefied petroleum gas, natural gas condensate, process gases and components thereof, crude oil, coal, coke as well as their derived materials), when used in incineration, co-incineration or biomass combustion plants to process input materials listed under points a) to h);

u) substances which occur in nature which are used in production processes of the iron and steel industry; or

v) substances which occur in nature and chemical substances, with the exception of:
    - those listed under points a) to j),
    - waste within the meaning of Directive 2008/98/EC,
    - substances or mixtures which have ceased to be waste in one or more Member States by virtue of the national measures transposing Article 6 of Directive 2008/98/EC,
    - substances formed from precursors which have ceased to be waste in one or more Member States by virtue of the national measures transposing Article 6 of Directive 2008/98/EC, or mixtures containing such substances,
    - non-biodegradable polymers, and
    - animal by-products or derived products within the scope of Regulation (EC) No 1069/2009.

[Note: The exclusion of a material from a lettered item does not prevent it from being an eligible component material by virtue of another lettered item.]

Related to animal by-products, the following proposals are put forward for further consideration and evaluation by the Commission and other bodies as a possible end-point in the manufacturing chain in accordance with the third subparagraph of Article 5(2) of Regulation (EC) No 1069/2009:

Thermal oxidation materials & derivates that have been derived from animal by-products and derived materials from category 2 or category 3 material as defined by Regulation (EC) No 1069/2009 provided that they:

- are compliant with the proposed criteria for CMC YY.

5.4.3 Production process conditions

Thermal oxidation materials can be obtained from combustion plants that are specifically designed for the **purpose** of producing fertilising materials or they can be a production residue resulting from a process aimed at **disposing of waste** or **producing a different primary product** (e.g. steel). The thermal oxidation plant can be a **stand-alone** installation or be **integrated** into another system.

5.4.3.1 Pre-processing

Input materials with a high **moisture content** are typically subject to mechanical treatments such as thickening, dewatering or drying. Occasionally, treatments combining an increase in the dry matter content of the input material and energy recovery (e.g. anaerobic digestion, hydrothermal carbonisation) may be applied. Hot gases exiting the furnace could also pass through an energy recovery system at the thermal oxidation plant whereby the energy can be (partly) recovered in the form of heat or electricity. The heat can be used for maintaining the combustion temperatures or for the pre-drying of the input material prior to combustion.

No limitations are proposed on any possible pre-processing steps as long as the input material list is respected. This implies that the input materials, and a combination thereof, may be physically mixed, screened, sized and chemically reacted. Also, any materials obtained from material transformation processes such as digestion, composting, pyrolysis, hydrothermal carbonisation, etc. will be permitted as long as the final thermal oxidation materials & derivates meet the product quality requirements and the minimum conditions for the core process.

5.4.3.2 Core process

Combustion conditions and carbon contents in ashes and slags

The **chemical composition and contaminant levels** present in thermal oxidation materials are not only largely influenced by the characteristics of the biomass input materials, but also by the **operating conditions** during thermal oxidation, including the type of furnace (grate firing versus fluidised bed combustion), the combustion temperature and the residence time of ashes (Steenari et al., 1999; Obernberger and Supancic, 2009; Tan and Lagerkvist, 2011; Pöykiö et al., 2014).

For large combustion plants (> 50 megawatt (MWth)) and waste incineration plants in Europe, the **combustion conditions are determined in the Industrial Emissions Directive** (2010/75/EC, IED). The IED prescribes that waste combustion plants shall be designed, equipped, built and operated in such a way that the gas resulting from the incineration of waste is raised, after the last injection of combustion air, in a controlled and homogeneous fashion and even under the most unfavourable conditions, to a temperature of at least 850 °C for at least 2 seconds (or 1 100 °C for 0.2 seconds), and that the total organic carbon content of slag and *bottom* ashes is less than 3% or their loss on ignition is less than 5% of the dry weight of the material. The legislation for medium combustion plants (between 1 MWth and 50 MWth; Directive (EU) 2015/2193) and for smaller appliances (heaters and boilers < 1 MWth) covered by Regulation (EU) 2015/1189 (on ecodesign requirements for solid fuel boilers) do not specify the combustion time-temperature conditions or the quality of the resulting ashes and slags; they focus on the emissions into the air. **Regulation (EU) No 592/2014** amending Regulation (EU) No 142/2011 as regards the use of **animal by-products and derived products** as a fuel in combustion plants adheres to the specifications as indicated in the Industrial Emissions Directive as described above.

At the **Kick-off Meeting of the STRUBIAS subgroup, there was widespread support among the participants to refer to the criteria for the thermochemical conversion of thermal oxidation materials & derivates from the Industrial Emissions Directive and the Animal By-Products Regulation**.

Organic carbon is one of the principal parameters for determining the ash and slag quality (Vehlow et al., 2006). Complete oxidation in the combustion process would convert this carbon entirely to CO_2. However, in real-world conditions a total conversion will never be accomplished and a small amount of products of incomplete combustion are found in all residue streams. In the gas phase, this is first of all CO. Other organic compounds are typically present as traces only. In the solid residues, the carbon speciation ranges from PAHs through soot or types of activated carbon to almost graphitic carbon (Ferrari et al., 2002). Most residue management regulations use the organic C as a key parameter indicating the degree of organic contaminant removal. Organic C serves as a reactive surface for the adsorption of possible contaminants, such as persistent organic pollutants (Vehlow et al., 2006). Modern incinerators show a tendency towards improved burnout and thus to lower organic C concentrations in all residue streams. An evaluation of available data indicates that such plants easily reach a > 99% conversion of carbon and its compounds in the waste to CO_2 (European Commission, 2006b; Vehlow et al., 2006). **No specific limit value is proposed for the loss on ignition (LOI)** as the combination of organic C, temperature, time and other product quality parameters (e.g. electrical conductivity, maximum levels for polyaromatic hydrocarbons) should be sufficient to delimit the scope of thermal oxidation materials & derivates. The organic C content is measured by default in many fertilising materials and forms part of the testing parameters for different PFC classes. Therefore, measuring the loss on ignition might be redundant and would only lead to a further increase in compliance testing requirements for thermal oxidation materials & derivates.

The IED and Regulation (EU) No 592/2014 on poultry litter only focus on the **carbon content of the bottom ashes.** The burnout of the particulate matter in the flue-gas and fly ashes is typically higher than that of the bottom ashes, and hence the former two materials usually show a lower organic C content than bottom ashes (Vehlow et al., 2006). Therefore, the C content is only measured in the bottom ash and slags. Even though some thermal oxidation materials & derivates are produced from *fly* ashes (e.g. sewage sludge fly ashes from fluidised bed systems), it is proposed that operators of incineration and biomass combustion plants shall perform testing on the bottom ash and slags residue fraction because (1) it is specified as such in the above-mentioned Regulations for the incineration of waste and animal by-products, (2) biomass plant operators typically discard the fly ash fraction due to its high content of metals and other contaminants, and (3) the bottom ashes typically show a higher C content.

It is proposed not to impose strict time-temperature profiles **(> 850 °C for more than 2 seconds or 1 100 °C for 0.2 seconds)** for **certain plant-based materials** in the technical requirements (i.e. those listed under point a) of the input material listed). Those materials **are inherently low in pollutants which is why stringent time-temperature profiles are not required to ensure the destruction of pollutants in the material, or proportionate considering the limited risk of emissions of certain persistent organic pollutants**. For the latter materials, a minimum temperature of 450 °C for at least 2 seconds has been proposed to ensure that the thermal oxidation material and derivates incorporated into the CE fertilising product have undergone a thermal oxidation process as intended in the scope of this CMC.

It is also proposed that the end-quality of the resulting ashes should meet the material quality requirements, including the organic carbon limits in the bottom ashes and slags. The incomplete combustion of uncontaminated biomass may lead to high levels of unburnt carbon in the ashes (Demirbas, 2005; James et al., 2012). In general, concentrations of CO, toxic volatile organic compounds such as acrolein, formaldehyde, and benzene, gaseous and particulate PAHs, and other organic species are enriched in emissions from incomplete biomass combustion (Rohr et al., 2015) and trace metals tend to accumulate in the organic ash fraction (Kabata-Pendias, 2011). High carbon contents in ash can also reduce ash stabilisation, increase the risk of spontaneous ignition after application, and significantly increase the ash volume. A complete combustion of the fuel also ensures low levels of persistent organic pollutants, such as dl-PCBs and PCDD/Fs (see Section 5.4.5.2). Finally, a high content of unburnt organic matter presents challenges for post-processing treatments such as pelletisation, briquetting and hardening as it decreases the binding properties of the ashes (James et al., 2012; Lövren, 2012). In line with the product definition of ashes as mostly inorganic compounds, it is **proposed to also limit the organic C in the bottom ashes to 3% (dry matter basis)**. Alternatively, biomass that is not completely oxidised in a thermal conversion process under oxygen-limiting conditions can be classified as a pyrolysis & gasification material (see also Section 5.2.8).

Additives and reactive agents

Section 5.4.2.2 provides a description of the reactants and additives used in thermal oxidation processes and specific P-recovery processes that aim at the production of better quality fertilising materials in an integrated, modified thermochemical process (e.g. RecoPhos, AshDec, Mephrec, EuPhoRe).

5.4.3.3 Post-combustion manufacturing steps

Raw ashes as obtained after thermal oxidation may undergo further manufacturing steps with the intention of **reducing levels of metals or metalloids to acceptable, safe levels** and/or **of increasing the plant availability of the phosphorus present in the ashes**. Therefore, raw ashes can be further processed as part of a '**two-step manufacturing process**', as outlined in Section 5.2.4 and Section 5.4.2.2. Such processes are typically of a chemical nature (e.g. Ecophos, RecoPhos, acidulation) or thermochemical nature (e.g. AshDec where ashes are re-incinerated together with other materials to achieve a chemical transformation/reordering of the ash compounds).

The reactants that can be applied in such processes principally include **virgin materials and other materials that can be used for the production of fertilisers derived from primary raw materials**, similar to those comprised under CMC 1 in the EU Fertilising Products Regulation ((EU) 2019/1009) (see Section 5.4.2.2 for details; substances excluding biomass, animal by-products, wastes, former wastes that do not meet EU End-of-Waste criteria, and their transformation products, fossil fuels, and raw materials for the steel and iron industry, and non-biodegradable polymers).

It is proposed to refer in the legal requirements to the need to intentionally modify the chemical composition of the material. The use of this terminology will prevent inert materials from being added to in the manufacturing of CE fertilising products with the sole intention of reducing the contaminant levels of the final CE product. Therefore, the addition of materials and substances to ashes and slags should at all times occur with the intention of improving the material quality and plant nutrient availability, of removing contaminants, or a combination of both. Operations aimed at lowering the contaminant concentration without lowering the contaminant to nutrient ratio in the original material should not be allowed. No further restrictions on the use of intermediates are proposed. Nonetheless, in order to comply with the legal requirements laid down in Directive 2008/98/EC, it is proposed that manufacturers that use **hazardous** ashes/slags (e.g. sewage sludge ashes with certain species of Zn; Donatello et al., 2010) within their STRUBIAS production process **should demonstrate the removal or transformation of the respective hazardous substances to levels below the limit values as defined in the EU Fertilising Products Regulation (EU) 2019/1009 and other EU policy documents on the classification of waste** (see Commission notice on technical guidance on the classification of waste - 2018/C 124/01). Although no best available techniques have been defined for STRUBIAS production processes, it is believed that – at present – such provisions might lead to the development of chemical and thermochemical techniques that enable the simultaneous compliance with the requirements laid down in Article 10 [‘the necessary measures shall be undertaken to ensure that waste undergoes recovery operations’] and Article 13 [‘protection of human health and the environment’] of Directive 2008/98/EC. Note that, in unlikely cases, hazardous substances (e.g. hazardous bio-waste) might also be used to react with ashes or slags in thermochemical post-combustion process, possibly classifying the process as a dilution of hazardous waste. However, under such conditions, manufacturers should still respect the conditions of the Waste Framework Directive, and any such operations could only be allowed in case of compliance with Article 18 of Directive 2008/98/EC, including the defined best available techniques for waste incineration. Therefore, no supplementary conditions to impede such practices are required.

Some stakeholders have argued that by setting no further restrictions on the use of intermediates, manufacturers are given the possibility to dilute contaminants present in ashes (e.g. metals in sewage sludge) to below the limit values for contaminants established at CMC and PFC level, **resulting in increased emissions of contaminants present in the input materials to the soil relative to techniques that effectively remove the contaminants by diverting them into a separate waste stream**. Under the current nutrient recovery proposals, such practices could indeed take place and the following arguments were raised to support this approach:

- Constraining the mixing of secondary raw materials in the EU Fertilising Products Regulation ((EU) 2019/1009) **would hamper the creation of a level playing field for fertiliser manufacturers**. After all, the mixing of input materials of different qualities is allowed for fertiliser manufacturers that use phosphate rock as input/source materials.
- The mixing of non-hazardous waste with other substances and mixtures is not prohibited in the existing EU legislation on waste.
- The mixing of input materials with intermediates is not exclusively performed for the dilution of waste, but also (1) to transform the P in the ashes into a more bio-available form and thus **decrease the ratio of contaminants to bio-available P in the end material**, and (2) to produce a chemical fertiliser blend (e.g. NPK fertiliser of the highest quality). For some micronutrients (e.g. Zn), the **addition of intermediates is in fact a good practice** as removing micronutrients from the ashes would require the addition of those elements afterwards to provide a balanced plant nutrition, involving their production from primary raw materials and associated environmental impacts.
- **Life cycle assessments** (Section 8) indicated that **producing P-fertilisers through the ‘mixing’ process provides benefits through the reduced need to extract the primary raw material phosphate rock, but that impacts on human health and the environment depend on the counterfactual use and handling scenario of the waste material that contained the sludges**.

For further discussion on this aspect, see Section 5.2.5.

For all these reasons, the following proposal is put forward for the implementation of post-combustion manufacturing processes:

5. An EU fertilising product may contain derivates from thermal oxidation materials that have been produced from the input materials listed in paragraph 1 and compliant with paragraph 4 and that have been manufactured according to a thermochemical conversion process compliant with paragraphs 2 and 3. The derivate manufacturing process shall be executed so as to intentionally modify the chemical composition of the thermal oxidation material, and be of the following nature:

a) Chemical manufacturing: derivates as produced through one or more chemical manufacturing steps that react thermal oxidation materials with materials listed under point k) of paragraph 1 that are consumed in or used for chemical processing. Non-biodegradable polymers shall not be used.

b) Thermochemical manufacturing: processes that thermochemically react thermal oxidation materials with reactants listed in paragraph 1 a) to k) that are consumed in or used for chemical processing. Thermochemical process conditions shall be compliant with paragraphs 2 and 3, and the thermal oxidation material derivate shall meet conditions listed in paragraph 4.

5.4.3.4 Finishing steps

No specific requirements for ‘finishing’ techniques that relate to the agglomeration or washing of materials have to be included at CMC level. Post-processes (e.g. modification of size or shape by mechanical treatment, washing with water) are normal industrial practice and any materials/processes required are included on the input material list. Hence, thermal oxidation materials & derivates may undergo further post-processing steps with the intention of increasing the chemical stability of the ashes or of agglomerating ashes as pellets or granules (Vesterinen, 2003).

5.4.4 Agronomic value

5.4.4.1 Nutrient contents and element ratios

Based on the characterisation of ash properties in relation to their potential utilisation (Vassilev et al., 2010; Vassilev et al., 2013a; Vassilev et al., 2013b), ashes are classified according to their **elemental composition** (Figure 3).

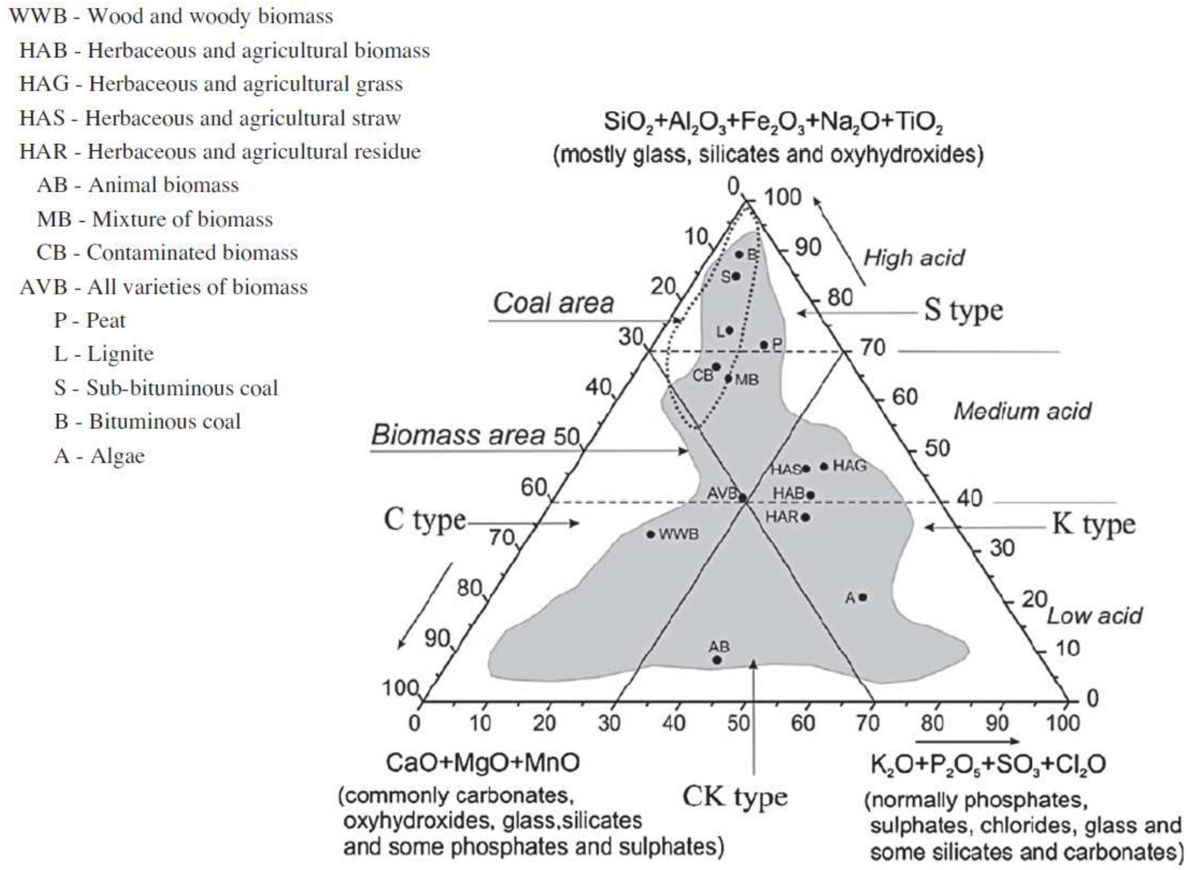


Figure 3: The classification system of ashes from fossil fuels and biomass based on the composition of their major elements (adopted from Vassilev et al., 2013b - © Elsevier Ltd, 2012)

Most raw ashes (**'K type'**, right-hand side of the triangle) are relatively **rich in one or more of the essential plant macronutrients P, K, and S**. Ashes that will be applied with the intention **of increasing the soil pH** (liming materials) are characterised as '**C type** ashes', and show high Ca and Mg contents. The high nutrient contents of ashes derived from the eligible input materials is confirmed in Section 15.2.1, and most of these ashes will be classified as C type, K type or CK type. The macronutrients K, Ca, Mg and S are relatively easily leached from ashes and thus available to plants, especially in the plant rhizosphere where plants may create a relatively acidic micro-environment through the release of root exudates (Freire et al., 2015). Phosphate (PO_4^{3-}), however, may be unavailable to plants when strongly bound to particular bi- and trivalent ions. Therefore, it is proposed to establish minimum requirements or enforce labelling of plant-extractable P fractions for thermal oxidation materials & derivates as outlined in Section 5.2.2.

'**S type**' ashes are dominated by **glass, silicates and oxyhydroxides** (mainly of the elements Si, Al and Fe), but fail to have a significant amount of carbonates, phosphates or sulphates, making them **unsuitable as liming materials or macronutrient fertilisers**. These ashes cover a relatively small zone at the top of the triangle, and are mostly produced from lignite, sub-bituminous coal and bituminous coal. As indicated by the STRUBIAS subgroup, there is no basis of support for including these materials as thermal oxidation materials & derivates in the EU Fertilising Products Regulation ((EU) 2019/1009). Therefore, the use of fossil fuels as input materials is limited to their role as auxiliary fuels in incineration, co-incineration, and biomass combustion plants as indicated in Section 5.4.2. Hence, fossil fuel ashes formed at fossil fuel power stations cannot be considered thermal oxidation materials. The direct consequence of this restriction of input materials is that **no further criteria are required to exclude such 'S type' ashes, dominated by glass, silicates and oxyhydroxides.**

Also, thermal oxidation material derivates that are produced through post-combustion manufacturing steps show a high content of plant-available nutrients. All these materials have a minimum P_2O_5 content of 15-20%, and are specifically intended to be used as a P-fertiliser. Therefore, **no additional requirements are proposed on minimum nutrient or neutralising value for thermal oxidation materials & derivates**.

5.4.4.2 Salinity

Salinity is a generic term used to describe elevated concentrations of **soluble salts** in soils and water. Comprised primarily of the most easily dissolved ions - sodium (Na) and chloride (Cl), and to a lesser extent calcium, magnesium, sulphate, and potassium - salinity in the environment adversely impacts water quality, soil structure, and plant growth (Pichtel, 2016). Although minimal accumulations (some in trace amounts) are required for normal biological function, excess salinisation might constrain crop productivity and threaten the presence of salt-intolerant plant and epiphyte species in natural ecosystems, as high dissolution rates of salts may impact upon the vegetation community. Excess sodicity can cause clays to deflocculate, thereby lowering the permeability of soil to air and water. Sodium (Na) and Cl are often present in thermal oxidation materials as inorganic salts such as sylvite (KCl) and halite (NaCl) in relatively high concentrations (Freire et al., 2015).

a. **Chloride** contents in thermal oxidation materials & derivates can be very high (e.g. in ashes from cereal and straw combustion; up to 35% of the total dry matter content), especially when expressed relative to other micronutrients (Section 15.2.1). On average, 67% of the chlorides present in ashes are water-soluble (Vassilev et al., 2013b). Hence, in specific settings and ecosystems, a significant risk is present for crops, natural vegetation and long-term soil quality when thermal oxidation materials are applied during prolonged periods of time. The Finnish legislation on the use of ashes in forest ecosystems contains a limit value of 2% for chloride (Haglund and Expertsgroup, 2008). In the EU Fertilising Products Regulation ((EU) 2019/1009) (Annex III to the proposal – Labelling requirements), it is stated that the phrase 'poor in chloride' or similar may only be used if the chloride (Cl-) content is below 3%. It is agreed that a labelling requirement cannot prevent a product high in chloride from causing adverse impacts on the environment, but, at the same time, the STRUBIAS subgroup indicated that this CMC offers the possibility to recover KCl, a macronutrient fertiliser, from ashes (e.g. 'Ash2Salt' process; Easymining Sweden). Therefore, it is proposed to set a limit value of 3% for

Cl-, only applicable when Cl^- is an unintentional constituent coming from the starting material(s).

b. **Sodium** contents in thermal oxidation materials & derivates are generally low (< 1%), although some residues such as olive husks can have higher contents. Moreover, Na plays a role as a 'functional nutrient', with a demonstrated ability to replace K in a number of ways for vital plant functions, including cell enlargement and long-distance transport, and its presence is even a requirement for maximal biomass growth for many plants (Subbarao et al., 2003). **Therefore, no limits on the Na content of thermal oxidation materials & derivates are proposed**.

c. At present, reliable methods other than leaching tests to characterise ash with regard to the speed of salt dissolution in the field are missing. One way of estimating the stability of thermal oxidation materials & derivates is to measure the conductivity in water extracts. This gives a total measurement of the dissolution of salts from the ash and indicates the risk of acute damage to vegetation, especially mosses and lichens. Given the labelling provisions for the closely related parameter Cl-, it is, however, proposed not to add any further criteria or labelling requirements for **electrical conductivity**.

5.4.4.3 Boron toxicity

Boron (B) is a very common element that may be present **in coal and some biomass ashes**, and some boron phases may be **readily water-soluble** (Pagenkopf and Connolly, 1982; Basu et al., 2009). Boron phytotoxicity is a major potential problem associated with the use of fresh fly ash as a fertilising material. Although boron is an essential nutrient in plants at low concentrations, it becomes toxic in many plants at concentrations only slightly higher than the optimal range (Ayers and Westcot, 1985; Sartaj and Fernandes, 2005). A number of studies have indicated that the solubilisation of B in coal ashes may lead to B toxicity in plants and aquatic organisms (Adriano et al., 1978; Straughan et al., 1978; Zwick et al., 1984; Aitken and Bell, 1985) and could cause B-induced inhibition of microbial respiration (Page et al., 1979) depending on the form and concentration of boron, type and characteristics (e.g. life stages) of the organism, and period and type of exposure to boron (acute or chronic). Recent evidence indicates that human B intake from food and water in the EU are below the tolerable upper intake level (EFSA, 2004), and that increased human B uptake is even promoted to enhance health due to the beneficial effects at low B concentrations (Nielsen, 2014; Pizzorno, 2015). Moreover, the most extensive and most recent dataset for rivers/catchments or regions in the EU contains consistently low B values (Heijerick and Van Sprang, 2004).

Fertilising products derived from thermal oxidation materials, including those derived from sewage sludge ashes, animal manures and wood bottom ashes, contain B contents well below the limits applicable in Lithuania and Sweden for ash-based fertilising products (500 mg B kg^{-1}) (see Section 15.2.2), and typically well below the B concentrations encountered in mineral P fertilisers (on average 1 291 mg B kg-1 for superphosphates; Kratz et al., 2016). Moreover, potentially boron-rich coal ashes are excluded from thermal oxidation materials & derivates because of the limitations on eligible input materials (see Section 5.4.2.1).

Therefore, it is concluded that negligible risks are associated with thermal oxidation materials & derivates for aquatic organisms, plants and humans. Hence, **it is proposed not to set a limit for the B content of thermal oxidation materials & derivates at CMC level.**

5.4.5 Environmental and human health and safety aspects

5.4.5.1 Metals and metalloids

This section considers concerns associated with the exposure **to alkali, alkaline earth metals, transition metals and other metals.** Whereas some of them are plant micronutrients, the potential **dissolution and accumulation to toxic levels of these inorganic metals and metalloids present in thermal oxidation materials & derivates requires a more in-depth risk assessment.** Metal or metalloid species may be considered ‘contaminants’ if their presence is unwanted or occurs in a form or concentration that causes detrimental human or environmental effects.

The primary response **of plants upon exposure to high levels of metals and metalloids in soils is the generation of reactive oxygen species and oxidative stress** (Mithöfer et al., 2004). The **indirect mechanisms** include their interaction with the antioxidant system (Srivastava et al., 2004), disrupting the electron transport chain (Qadir et al., 2004) or disturbing the metabolism of essential elements (Dong et al., 2006). One of the most deleterious effects induced by metals in plants is lipid peroxidation, which can directly cause biomembrane deterioration.

Living organisms require varying amounts of metals and metalloids. Iron, cobalt, copper, manganese, molybdenum, and zinc are required by humans, but all are **toxic** at higher concentrations (Singh et al., 2011a). Other heavy metals such as Cd, Hg and Pb are toxic elements that have no known vital or beneficial effect on organisms, and their accumulation over time in the bodies of animals can cause serious illness. **The ingestion of metals and metalloids by humans may disrupt metabolic functions, as they can accumulate in vital organs and glands such as the heart, brain, kidneys, bone, liver, etc. and could displace the vital nutritional minerals from their original binding sites, thereby hindering their biological function (Singh et al., 2011a).**

Aluminium, iron and manganese

Aluminium (Al) is the most commonly occurring metallic element, comprising 8% of the earth's crust (Press and Siever, 1974). It is a major component of almost all common inorganic soil particles, with the exception of quartz sand, chert rock fragments, and ferromanganiferous concretions. The typical range of Al in soils is from 1% to 30%, with naturally occurring concentrations varying over several orders of magnitude. The available data on the environmental chemistry and toxicity of Al in soils to plants, soil invertebrates, mammals and birds indicate that total Al in soil is not correlated with toxicity to the tested plants and soil invertebrates (EPA, 2003a). However, **aluminium toxicity is associated with soluble Al** and thus dependent upon the chemical form (Storer and Nelson, 1968). Insoluble

Al compounds such as Al oxides are considerably less toxic compared to the soluble forms (aluminium chloride, nitrate, acetate, and sulphate), and only moderately toxic effects of insoluble Al for humans have been observed at extremely high intake ratios that are unrealistic through fertiliser-based exposure pathways (Krewski et al., 2007). Aluminium in ashes is predominantly present in stable forms, and the **Al content in the soluble and exchangeable forms is extremely low** (~ 0.2%) (Lapa et al., 2007; Ibrahim, 2015). Aluminium from ashes is mainly released as $Al(OH)_4^-$ (99%) (Ibrahim, 2015). Although $Al(OH)_4^-$ is considered to be non-toxic, phytotoxic effects could occur due to the gradual formation of toxic Al species in the bulk nutrient solution, resulting from the acid soil pH or the acidification of the alkaline nutrient solution by the plant roots (Kopittke et al., 2005). Ecological risks associated with the addition of Al could therefore be present, especially in acidic soils ($pH_{H2O} < 5.5$). The mechanism that underlies Al phytotoxicity is that Al displaces Ca from the apoplast and thus reduces the number of exchange sites for Ca uptake (Godbold et al., 1988). Therefore, not the concentration of Al in itself is critical, but rather the molar ratio of Ca/Al. Hence high Ca concentrations may reduce the toxic effects of Al (Godbold et al., 1988). Ashes derived from the eligible input materials are typically rich in Ca which is easily leached (Vassilev et al., 2013b). The leaching of Ca from the ashes is much more pronounced than the leaching of soluble Al (Neupane and Donahoe, 2013; Ibrahim, 2015). Hence, it is concluded that most Al in (the neutral to basic) ashes is present as insoluble aluminium oxides and hydroxides, and that the availability of Ca in the ashes and slags derived from the eligible input materials far exceeds that of Al.

A STRUBIAS subgroup expert highlighted that, in the case of long-term continuous applications of Al-rich ashes such as wood ashes, the P sorption capacity of the soil matrix could be increased, possibly rendering P unavailable to plants. Phosphorus availability is controlled by several factors such as soil organic matter levels, soil pH, and soil aluminium and iron contents, making it a challenge to estimate how much P will be supplied to the crop. In this respect, it is important to note that most of the Al-rich ashes are also characterised by a high pH, and may act as a liming material. The acid-neutralising capacity of the ashes may therefore also have a positive impact by making soil P more available in acid and neutral soils. In combination with the often higher bulk Al contents in soils than in the thermal oxidation materials (Section 15.2.1), large uncertainties are therefore associated with the possible impact of the use of thermal oxidation materials & derivates as fertilising materials on the soil P sorption capacities in the EU. Moreover, in several EU Member States where the application of raw (wood) ash is common practice, requirements on maximum application rates over long-term periods are in place (Haglund and Expertsgroup, 2008). Therefore, **no specific criterion is proposed for Al content in thermal oxidation materials & derivates.**

Iron (Fe) is also a commonly occurring metallic element, with typical soil concentrations ranging from 0.2% to 55%. Iron can occur in either the divalent (Fe^{+2}) or trivalent (Fe^{+3}) valence states under typical environmental conditions. The valence state is determined by the activity of the hydrogen cation (pH) and the activity of electrons (Eh) of the system, and the chemical form is dependent upon the availability of other chemicals. Iron is essential for plant growth, and is generally considered to be a micronutrient. Iron is considered the key metal in energy transformations needed for syntheses and other life processes of the cells

(Merchant, 2010). The main concern from an ecological risk perspective for iron is not direct chemical toxicity per se, but the effect of iron as a mediator in the geochemistry of other (potentially toxic) metals (EPA, 2003b). Similar to Al, Fe in ashes is predominantly present in stable forms, and the **Fe content in the soluble and exchangeable forms is low** (~ 0.3%) (Vassilev et al., 2013b; Ibrahim, 2015). Moreover, in well-aerated soils between pH_{H2O} 5 and 8, **iron is not expected to be harmful to plants** (Römheld and Marschner, 1986), but under specific conditions it can become toxic (e.g. in rice plants). Therefore, **no specific criterion is proposed for Fe content in thermal oxidation materials & derivates.**

Regulatory interest in the assessment of the potential risks to soil from **manganese (Mn)** exposure has increased with increasing anthropogenic activity and industrial development. Not only can Mn be **toxic for plants and animals**; toxicity for **humans** has been reported as well from occupational (e.g. welder) and dietary overexposure. Toxicity has been demonstrated primarily in the central nervous system, although lung, cardiac, liver, reproductive and foetal toxicity have also been detected (Crossgrove and Zheng, 2004). In contrast to Al and Fe, Mn concentrations in ashes might be up to 10 times higher than the soil background Mn concentrations, hence potentially substantial risks are associated with the application of Mn-rich thermal oxidation materials & derivates. Moreover, up to 46% of the Mn present in ashes may be water-soluble (Vassilev et al., 2013b). The limit values for soil Mn concentrations associated with toxic effects on organisms are below the background concentrations of most soils, thus making their use in the assessment of potential risks impossible (EPA, 2003c; ESDAT, 2017). Also, little is known about the toxicity of colloidal, particulate, and complexed manganese, though the toxicities of metals bound into these forms are assumed to be less than those of the aqua-ionic forms (World Health Organization, 2004), Hence, there are some important challenges when it comes to deriving limit values to address potential terrestrial risks, including the variability of ambient soil background concentrations, the changing form and subsequent ecotoxicology of Mn with changing soil conditions, as well as the poor relationship between standard ecotoxicity test data for all trophic levels and the reality in the field (International Manganese Institute, 2012). As a matter of fact, it has been acknowledged by the WHO that, due to the highly variable natural background concentrations and the influence of transient waterlogging and pH changes on manganese speciation**, deriving a single guidance value for the terrestrial environment is inappropriate** (World Health Organization, 2004). Therefore, existing national legislative frameworks do not contain limit values for maximal Mn contents in thermal oxidation materials, with the exception of the UK poultry litter ash quality protocol (limit of 3.5% on a dry matter basis). The values observed for thermal oxidation materials & derivates are typically below 3.5%, with the highest Mn concentrations observed for fly and bottom wood ashes (up to 1.3% and 2.9% observed in data collected, respectively; Section 15.2.2). Ashes and slags derived from other eligible input material typically show Mn concentrations that are one to two orders of magnitude lower than the limit value of 3.5% of the UK poultry litter ash quality protocol (Section 15.2.2). In order to inform end users, **it is proposed that EU fertilising materials derived from CMC thermal oxidation materials & derivates that have Mn contents above 3.5% should label their Mn content.**

Assessment of the potential accumulation of trace metals/metalloids in soil

The pathways that lead to the presence of **metals and metalloids in eligible input material for thermal oxidation materials** often start within the food chain via plant and water uptake by roots, and by adsorption from the air. Additionally, fossil fuels and ore concentrates can have high concentrations of particular toxic metals, such as Tl, Cr and V, that were present in specific geological substrates (Karbowska, 2016). Wood contains generally higher amounts of metals than short-lived biomass sources, because of the accumulation during the long rotation period of forests, the higher deposition rates in forests and possibly the lower pH value of forest soils (Vamvuka and Kakaras, 2011) (Section 15.2.2). Because of the transfer from one link in the chain to another, some heavy metals may end up being accumulated by humans (Hapke, 1996). The recycling of metals and metalloids in the environment is evident as metals being taken up by plants used as animal feed or food end up in excreta, which are spread on land, and can ultimately lead to increasing concentrations in agricultural soils over time. Even relatively small additions to the cycle may thus lead to high soil concentrations over time (van der Voet et al., 2010). This indicates that not only ecotoxicity associated with the dispersion of metals in the atmosphere or towards freshwater bodies should be taken into account, but also the vulnerability of the soil ecosystem. Root exudates, particularly organic acids, are able to increase metal mobility, solubility and bioavailability in soil and consequently enhance the translocation and bioaccumulation of metals (Ma et al., 2016). To avoid an increase of unwanted toxic heavy metals in food for human consumption, **it is necessary to limit the concentrations upstream in the food chain.**

In line with the objective of the STRUBIAS subgroup, a **broad range of eligible input materials** has been proposed for the CMC thermal oxidation materials & derivates. Amongst others, non-hazardous waste and non-hazardous industrial by-products, animal by-products, ore concentrates and fossil fuels could be used as eligible input materials. This is possible because the combustion process enables the effective removal of most organic contaminants present in the eligible feedstocks, and **on the singular condition that the adverse effects associated with the presence of metals and metalloids is carefully evaluated**. Thermal oxidation processes result in the losses of organic matter and several volatile nutrients (e.g. N and S), whereas metals and metalloids are only partially removed. The temperature of between 800 °C and 1 050 °C in the combustion chamber of a fluidised bed boiler is high enough to vaporise some of the elements. In addition to element volatilisation characteristics, element retention through other processes in fly ash (primarily condensation processes) determines the final fate of volatilisable elements (Álvarez-Ayuso et al., 2006; Kuokkanen et al., 2006). Most of these species form compounds that condense on the surface of particles in the flue-gas, leading to the enrichment of some elements in the fly ash fraction. The bottom ash has an enhanced content of non-volatile components, and frequently contains sintered or melted particles. **The direct result is the significant concentration of metals/metalloids in thermal oxidation materials (Demirbas, 2003).**

Unlike organic contaminants, which may be oxidised by microbial action, most metals do not undergo microbial or chemical degradation, and they remain in the soil for a long time after their introduction. Changes in their chemical forms (speciation) and bioavailability are, however, possible. **Metal and metalloid contamination of soil may pose risks and hazards to humans and the ecosystem through direct ingestion or contact with contaminated soil, the food chain (soil-plant-human or soil-plant-animal-human), drinking of contaminated ground or surface water, effects on aquatic organisms, reduction in food quality (safety and marketability) via phytotoxicity, reductions in soil quality and soil faunal biodiversity, and the reduction in land usability for agricultural production causing food insecurity** (World Health Organization, 1996; Demirbas, 2003; Wuana and Okieimen, 2011).

It has been reported that metals such as cobalt (Co), copper (Cu), chromium (Cr), magnesium (Mg), molybdenum (Mo), nickel (Ni), selenium (Se), vanadium (V) and zinc (Zn) are essential nutrients that are required for various biochemical and physiological functions, but the inadequate supply of these micronutrients results in a variety of deficiency diseases or syndromes (described in detail in World Health Organization, 1996). Other metals such as aluminium (Al), antimony (Sb), arsenic (As), barium (Ba), beryllium (Be), bismuth (Bi), cadmium (Cd), gallium (Ga), germanium (Ge), gold (Au), indium (In), lead (Pb), lithium (Li), mercury (Hg), nickel (Ni), platinum (Pt), silver (Ag), strontium (Sr), tellurium (Te), thallium (Tl), tin (Sn), titanium (Ti), and uranium (U) have no established biological functions and are considered non-essential metals (Chang et al., 1996). In humans and biological systems of soil and aquatic organisms, these metals/metalloids have been reported to affect cellular organelles and components such as cell membranes, mitochondriae, lysosomes, endoplasmic reticula, and nuclei, and to inhibit some enzymes involved in metabolism, detoxification, and damage repair (World Health Organization, 1996; Wang and Shi, 2001).

Trace elements found in ashes from the eligible input materials that could accumulate in soils include **As, Ba, Be, Cd, Cr, Co, Cu, Hg, Mo, Ni, Pb, Sb, Se, Tl, V and Zn** (Pitman, 2006; Vassilev et al., 2013a; Vassilev et al., 2013b; Rohr et al., 2015; Karbowska, 2016). Most studies and risk assessments have primarily focused on inorganic elements of major environmental concern, such as As, Cd, Cr (VI), Pb, Ni or Hg among others, while overlooking other constituents (e.g. Ba, Be, Mn, Mo, Sb, Tl and V are poorly studied) which, inaccurately, have been considered as generally posing little risk to the environment.

Some inorganic metals and metalloids are already regulated for different PFCs in the EU Fertilising Products Regulation ((EU) 2019/1009). Specifically, limit values for As, Cd, Cr (VI), Hg, Ni and Pb have already been formulated in the EU Fertilising Products Regulation ((EU) 2019/1009) for the different PFCs where thermal oxidation materials & derivates could be used as ingredients. Also, Zn and Cu are elements that may be regulated at PFC level, which is why these elements are not included in this assessment at CMC level. Therefore, the present assessment will be restricted to **Ba, Be, Co, Cr, Mo, Sb, Se, Tl and V** (see Section 16.1)**.** The methodology of the assessment is centred on a three-step approach:

1) In a first step, soil screening values are collected for the different EU Member States. **Soil screening values** are generic quality standards that are used to regulate land

contamination and are adopted in many Member States in Europe in order to protect the environment and human health (Carlon, 2007). The soil screening values were compiled for the different EU Member States, and it was assessed whether the list of elements covers all relevant hazards associated with metals/metalloids based on the techno-scientific literature for thermal oxidation materials derived from the eligible input materials. If not, complementary evidence was sought in scientific literature, as was the case for thallium (see details in Section 16.1, as well as the specific section below on thallium).

2) In a second step, a maximal permissible concentration of the element in the CMC derived fertilising material is calculated based on the principle that predicted metal/metalloid accumulation as a result of the long-term application of the fertilising material and the atmospheric deposition in the soil shall not exceed the so-called soil screening value *(‘soil screening acceptable limit concentration’*). A mass balance approach is applied assuming that the **non-soluble fraction of metals and metalloids accumulates in soils**, and that the soluble metal fraction is removed from the soil through leaching and plant uptake. The calculated accumulation of the respective trace metal in soils is then dependent 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 soils. A simple spreadsheet-based model based on a set of reasonable assumptions is applied for this purpose as outlined in detail in Section 16.1. The **calculated soil screening acceptable limit contents** are then qualitatively compared to metal/metalloid concentrations that are typically found across the diverse range of thermal oxidation materials & derivates derived from different eligible input materials. This assessment is applied **to spot possible issues** that might lead to human health and environmental protection issues due to the accumulation of metals and metalloids present in the CMC material. The assessment may not be interpreted as a risk-based assessment as it solely focuses on the accumulation of specific metals in soils, but does not consider the resulting impacts on relevant end-points such as soil and aquatic organisms and humans.

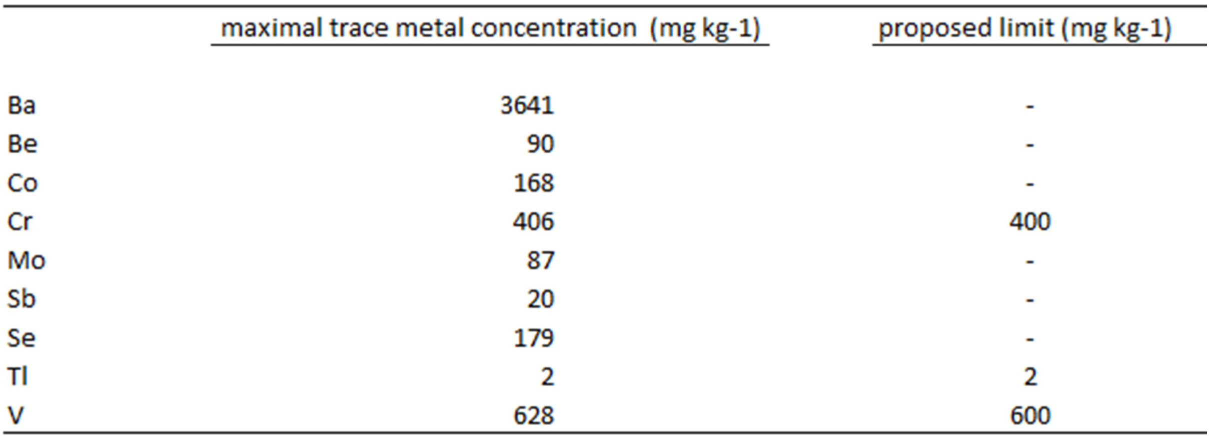
3) In a final step, **an interpretation and validation of the soil screening acceptable limit concentration is performed by reviewing metal-specific available information in a risk-based context.** This is especially relevant for thermal oxidation materials & derivates as some of the elements for scrutiny are not routinely present in fertilising materials. Based on the precautionary principle, technical requirements on maximal permissible limit values for metals/metalloids in thermal oxidation materials & derivates could be proposed in the event that there is qualitative techno-scientific evidence to suggest that the use of specific thermal oxidation materials & derivates could lead to unacceptable human health or environmental risks.

Hence, the assessment is **principally based on the soil screening values for metal and metalloid concentrations in the soil as established by the EU Member States**. A full risk assessment of ecological and human health risks from the presence of the metals and metalloids in fertilising materials falls beyond the scope of this study.

Acceptable soil screening limit concentrations

The outcome of this analysis indicates the soil screening acceptable limit concentrations of selected metals and metalloids in CE fertilising products derived from thermal oxidation materials & derivates as given in Table 4.

Table 4: Outcome of the soil screening acceptable limit concentrations for selected metals and metalloids in EU fertilising products derived from thermal oxidation materials & derivates (mg kg^{-1} dry matter in EU fertilising product)



	maximal trace metal concentration (mg kg-1)	proposed limit (mg kg-1)
Ba	3641	-
Be	90	-
Co	168	-
Cr	406	400
Mo	87	-
Sb	20	-
Se	179	-
Tl	2	2
V	628	600

NB:
- indicates that the calculated maximal trace metal concentrations are well above concentrations found for thermal oxidation materials & derivates (Section 15.2.2) so no maximum value is proposed.

Interpretation and validation of the soil screening acceptable limit concentrations

The numbers obtained were compared to typical values observed in thermal oxidation materials & derivates (see Section 15.2.2). It was observed that, for Be, Co, Mo and Se, typical values present in the materials are much lower than the derived soil screening acceptable limit concentrations. Sensitivity analyses indicated that variations in model parameters, such as for instance the variation of Kd values along the ranges observed for European soils (Janik et al., 2015b), did not change the outcome of this comparison analysis. **Therefore, it is proposed not to set legal requirements for Be, Co, Mo or Se for this CMC.**

For barium, it is indicated that most thermal oxidation materials & derivates show Ba contents that are below the derived soil screening acceptable limit concentrations (typical range, 100-1 500 mg kg^{-1}). Nonetheless, **wood fly ashes** could show values up to 4 000 mg kg^{-1} (Section 15.2.2). Also, ashes derived from **coal** could show values up to 5 000 mg kg^{-1} (WHO, 1990), but the presence of coal ashes in the final CE fertiliser product will be limited by the proposed criteria on input materials (see Section 5.4.2). It is also indicated that Ba contents in Ba-rich thermal oxidation materials & derivates closely correlate to other contaminants, such as Pb, Ni and Cd (Section 15.2.2) (Krüger and Adam, 2015). Wood fly ash, the lightest component that accumulates in the flue system, can contain high concentrations of cadmium, copper, chromium, lead and arsenic which is why this ash cannot be used as an EU fertiliser (Pitman, 2006). Therefore, thermal oxidation materials & derivates that exceed the proposed Ba limit value of 3 641 mg kg^{-1} will also exceed the metal limit values at PFC levels for any of the PFC classes that could be targeted by these materials (e.g. macro- and micronutrient fertilisers, liming materials). Moreover, it should be noted that only a limited number of EU Member States have soil screening established for Ba (Table 34). For all these reasons, it is proposed **not to set legal requirements for Ba for this CMC.**

For the remaining elements, Cr, Sb, Tl and V, a more in-depth assessment to evaluate the risks was performed:

Chromium

Chromium is considered to be **non-essential for plant growth** and although trivalent chromium is essential to normal carbohydrate, lipid and protein metabolism, the consumption of Cr-contaminated food can cause human health risks by inducing severe clinical conditions (Shahid et al., 2017). Chromium ore deposits are primarily used for **metallurgical applications** such as the production of stainless steel, but other uses in wood preservation, leather tanning, pigments, and refractories exist (Barnhart, 1997). Chromium has several oxidation states (−2 to +6), but hexavalent chromate [Cr(VI)] and trivalent chromite [Cr(III)] forms are the most common and stable in the natural environment, with the former being less abundant than the latter. Therefore, Cr(total) is a good proxy for Cr(III). At PFC level, limits for Cr(VI) have been included in the EU Fertilising Products Regulation ((EU) 2019/1009) but, given that mineral ores which are used in the iron and steel industry are a possible input material for thermal oxidation materials & derivates and the fact that many EU Member States have established soil screening values for Cr(total), there is a need for a more detailed assessment on Cr(total). Specific fertilising products, including industrial by-products, can contain high Cr(III) concentrations. Ashes from the incineration of municipal and industrial sludges may contain Cr(total) contents of up to 1 502 mg kg^{-1} (Krüger et al., 2014), and about 20% of the sewage sludge samples included in the ECN database have values that exceed 400 mg kg^{-1} prior to incineration (ECN, 2017). Slags from the steel industry and by-products from the tannery industry have Cr(III) concentrations that range from 250 mg kg^{-1} to 2-3% (Pillay et al., 2003; Cornelis et al., 2008; Wang et al., 2015; Reijonen, 2017). The concentrations of Cr(III) in steel slags can thus be up to four orders of magnitude higher than the limit value as established for Cr(VI) in the EU Fertilising Products Regulation ((EU) 2019/1009) (2 mg kg^{-1} for PFC 1 - fertilisers).

Environmental and human health risks due to the presence of Cr(III) in fertilising materials could occur through: (i) the conversion of Cr(III) to Cr(VI), and (ii) through the leaching of Cr(III):

i. The two **species of Cr differ greatly with respect to their sorption and bioavailability in soil, root absorption, translocation to aerial plant parts and plant toxicity.** In most plant species, Cr is poorly translocated towards aerial parts and is mainly retained in the root tissues and converted into Cr(III) (Kabata-Pendias and Mukherjee, 2007; Peralta-Videa et al., 2009; Jaison and Muthukumar, 2017), thereby reducing the possible risk of human uptake for most crops. Also, most soils' microorganisms are only marginally affected by the presence of Cr(III) (European Chemicals Bureau, 2005). Nonetheless, **experimental test work under laboratory conditions with a number of slags and sludges indicates that very gradual oxidation of trivalent to hexavalent chromium does occur when the slag is exposed to atmospheric oxygen, rendering a quantifiable but small portion of chromium in this much more mobile and toxic form**. Pillay et al. (2003) showed that steel slag with a 1-3% Cr(III) content, released 1 000-10 000 mg kg^{-1} Cr(VI) within 6-9 months of exposure to an ambient atmosphere. Apte et al. (2006) showed that $Cr(OH)_3$ present in Cr-containing sludge slowly converts to hexavalent chromium, with short-term (90 days) conversion rates of up to 0.05% in moist soils and in the presence of MnO_2. When CaO and Cr_2O_3 coexist in the slag, oxidation of Cr_2O_3 occurs, and Cr(III) can be transformed into Cr(VI) under the action of O_2 from the atmosphere (Li et al., 2017b). $CaCrO_4$ formed at the surface area of the particles, and this Cr(VI)-enriched phase is freely soluble and almost dissolved completely at pH 7 (Li et al., 2017b). The likeliness of the last process is, however, unlikely for real slag systems in steelmaking that also comprise Al_2O_3, MgO and MnO (Cheremisina and Schenk, 2017).

ii. **The leaching of Cr(III).** Because of the high solubility of many thermal oxidation material derivates, leaching of Cr(III) can reasonably be expected from derivates that have been manufactured from specific input materials, such as municipal and industrial sludges. Also, steel slags may be prone to Cr(III) leaching, at least in the short term. Li et al. (2017a) indicated, for instance, average and maximum Cr concentrations in leachates from steel slags (3 500 mg Cr kg^{-1}) from steel slags that were 15 µg L^{-1} and 42 µg L^{-1}, respectively. Proctor et al. (2000) indicated Cr leaching values from steel slags (132 mg Cr kg^{-1} to 3 046 mg Cr kg^{-1}) that range from 10 µg L^{-1} to 60 µg L^{-1}. Using a modelling approach, De Windt et al. (2011) even indicated higher potential leaching from basic oxygen steel slags (1 900 mg Cr kg^{-1}) of up to 325 µg L^{-1}. Although increased concentrations of Cr(total) in the soil layer were only found at the depth of the ploughing layer (Algermissen et al., 2016) or slightly below (Kuhn et al., 2006), significant effects on EDTA-extractable (sensu 'easily-mobilisable') and HNO_3-extractable (sensu 'potentially mobilisable'), but not $CaCl_2$-extractable Cr (sensu 'plant-available) soil concentrations have been observed following long-term steel slag application to soils (Hejcman et al., 2009). The leaching of Cr(III) will be reduced through its adsorption on the soil matrix, but the processes in the soil are still not sufficiently investigated (Algermissen et al., 2016). The adsorption of Cr(III) onto soil follows the pattern typical of cationic metals and increases with increasing pH (lowering the pH results in increased protonation of the adsorbent, leading to fewer adsorption sites for the cationic metal) and the organic matter content of the soil and decreases when other competing (metal) cations are present (Jing et al., 2006; Kabata-Pendias, 2011). Certain dissolved organic ligands may, however, reduce the adsorption of Cr(III) to the solid phase by forming complexes which enhance the solubility of Cr(III) in the aqueous phase (Richard and Bourg, 1991). Moreover, Cr present in fertilising materials can reach nearby water

> bodies in a concentrated form through erosion and soil redistribution processes. Thus, it was deliberated whether **there is insufficient evidence to demonstrate the absence of a risk of Cr(III) being transferred from the soil towards the water bodies and consequently of possible adverse impacts upon aquatic biotic communities.** The European Chemical Bureau (2005) predicted no-effect concentrations (PNEC) for water bodies of 3.4 µg L^{-1}, which is in the same order of magnitude as the value for Cr(VI). The PNEC is the concentration of a chemical below which no adverse effects of exposure of the substance is expected to occur. This value is two orders of magnitude lower than the Cr leaching potential from steel slags as documented by De Windt et al. (2011).

Considering the possible transformation of Cr(III) to the highly toxic Cr(VI), the proposal is to retain the assessment based on the maximal permissible soil concentration limit value of 100 mg Cr kg^{-1} soil determined based on the soil screening values of EU Member States. Moreover, limiting the maximal Cr concentration in thermal oxidation material will further limit ecotoxicity effects due to the leaching of Cr(III).

For thermal oxidation materials & derivates derived from certain eligible input materials (e.g. sewage sludge, tannery sludge, textile waste, basic oxygen furnace slags derived from mineral ores and recycled ores), the proposed chromium limit of 400 mg kg^{-1} could be a limitation. Possibly, the permitted post-combustion manufacturing processes could help to decrease the Cr to acceptable levels.

Antimony

Antimony has a wide range of uses, including in the **manufacture of semiconductors, diodes, flameproof retardants, lead hardeners, batteries, small arms, tracer bullets, automobile brake linings, and pigments** (Filella et al., 2002a). The use of antimony in many different applications can be expected to drop in the future (van Vlaardingen et al., 2005). Antimony is not an essential element in plants or animals (Fowler and Goering, 1991). Its bioavailability and toxicological effects depend on its chemical form and oxidation state. The two common inorganic forms of antimony present in natural waters are antimonate ($Sb(OH)_6^-$) and antimonite ($Sb(OH)_3$) (Filella et al., 2002b). Experimental and clinical trials with compounds containing antimony have shown that the trivalent compounds are generally more toxic than the pentavalent compounds (Winship, 1987; Filella et al., 2002a; WHO, 2006).

Antimony can be present in thermal oxidation materials & derivates that are produced from the eligible input materials. In particular, **ashes of crop residues and sewage sludges show the highest Sb concentrations**, with values up to **70 mg Sb kg^{-1}** (Section 15.2.2; Kruger et al., 2015; Izquierdo et al., 2008). The bioavailability as expressed by the **leaching potential of Sb present in thermal oxidation materials & derivates typically ranges from 0% to 10%** (Kim et al., 2003; Cornelis et al., 2008; Izquierdo et al., 2008), but values **up to 36%** have been observed (Miravet et al., 2006). Antimony **is mostly retained in soils** (McLaren et al., 1998; Flynn et al., 2003; Wilson et al., 2010). Obviously, the extent of retention influences the bioavailable and mobile fraction. Many factors impact retention, but Sb is generally retained on silicate clay minerals, iron oxides and hydroxides and organic matter. In many natural environments, adsorption on the Mn and Fe oxyhydroxides is responsible for retention of a high proportion of the soil bound Sb.

Information on the acute and chronic toxicity of dissolved antimony to a variety of aquatic, soil and sediment organisms is reviewed in the risk assessment report prepared by the EU (EURAR, 2008) and van Vlaardingen (2005). The EURAR risk assessment is for antimony trioxide and is only available in draft status. The data reviewed indicate that soluble forms of antimony generally only have a **low to moderate potential to cause harm to aquatic, soil and sediment organisms**. Relevant predicted no-effect concentrations (PNEC), as derived by dividing the lowest no-observed-effect concentration by an assessment factor of 10, in the EURAR report are as follows: $PNEC_{surface\ water}$: 113 µg Sb/L; $PNEC_{sediment}$: 11.2 mg Sb/kg(dry weight); $PNEC_{microorganisms}$: 2.55 mg Sb/L; $PNEC_{soil}$: 37 mg Sb/kg (dry weight). Van Vlaardingen (2005) determined a serious risk concentration value for the soil compartment of 54 mg kg^{-1} soil and chronic ecotoxicological serious risk concentration values of 11 mg Sb L^{-1}. **The values for the soil compartment are much higher than the value applied in this assessment, based on the 25th percentile of the soil screening value as determined by the EU Member States (3 mg Sb/kg; see above).** Moreover, experimental bioconcentration factors obtained for fish, aquatic invertebrates, plants and algae vary between 0.19 L/kg and 24 L/kg wet weight (Shigeru et al., 1997; Tschan et al., 2008), **much lower than the limit values for the bioaccumulation criterion** (> 2 000 L/kg) as established by Regulation (EC) 1907/2006.

Based on the information presented, **it is concluded that the risk of Sb entering the environment in a harmful quantity or concentration or under conditions that have or may have an immediate or long-term harmful effect on the environment or its biological diversity due to the application of thermal oxidation materials & derivates is low**. Therefore, the proposal is to not retain the assessment based on the maximal permissible soil concentration limit value of 3 mg Sb kg^{-1} soil determined based on the soil screening values of EU Member States, and no limits for Sb in thermal oxidation materials & derivates are proposed.

Thallium

Thallium (Tl) is ubiquitous in nature and is found especially in sulphide ores, usually at low concentrations (WHO, 1996). An estimated global industrial consumption of 10-15 tonnes/year was estimated for 1991. **Activity of mineral smelters, coal-burning power-generating plants, brickwork and cement plants** generate man-made emissions to air and in waste deposits of approximately 2 000-5 000 tonnes/year. A large fraction of thallium is released into the atmosphere, since thallium compounds are volatile at high temperatures. Further sources of thallium emission are iron and steel production, non-ferrous metal (e.g. Zn, Cd) smelting and gold production (WHO, 1996). Thallium occurs in **two oxidation states** in the environment: monovalent Tl(I) and trivalent Tl(III). The oxidation state directly influences the toxicity of thallium; trivalent Tl is approximately 50 000 times more toxic than monovalent Tl. Furthermore, Tl(I) may be oxidised to Tl(III) due to the activity of phytoplankton (Twining et al., 2003). As a result, the toxicity of both species is influenced by their stability, which is associated with the type of sample matrix and the corresponding environmental conditions. Specific ashes and melting materials, mainly those partially derived from coal and mineral ores can be rich in Tl, with concentrations ranging from **7 mg kg^{-1} to 11 mg kg^{-1} for steel slags** (Proctor et al., 2000) and from **1 mg kg^{-1} to 76 mg kg^{-1} for coal ashes** (Frattini, 2005; Lopez Anton et al., 2013; Karbowska, 2016; Świetlik et al., 2016; Vaněk et al., 2016). Although no data are available for sewage sludge ashes, the few observed Tl values of sewage sludge in Austria, Germany and the Netherlands (Scharf et al., 1997; Wiechmann et al., 2013b; ECN, 2017) of 0.2-0.9 mg kg^{-1} indicate that sewage sludge ashes could possibly have values that exceed 2 mg kg^{-1} (assuming an ash content of ~ 35-40% for sewage sludge).

Data regarding significant sources of thallium in the environment and risks for the environment and human health have been reviewed by Karbowska (2016). **Thallium is considered toxic for human and animal organisms, microorganisms and plants** (Makridis and Amberger, 1996; Nriagu, 1998; Peter and Viraraghavan, 2005). The toxicity of this element is higher compared to mercury, cadmium and lead (maximum admissible concentration at 0.1 mg mL^{-1}) (Repetto et al., 1998; Peter and Viraraghavan, 2005). The toxicity of thallium-based compounds is mainly caused by the similarity between thallium (I) ions and potassium ions (Grösslová et al., 2015), which results in the disorder of potassium-associated metabolic processes due to thallium interference (Wojtkowiak et al., 2016). **Human exposure** to thallium is mainly associated with the consumption of contaminated food or drinking water. Thallium rapidly enters the bloodstream and is transported around the whole organism, which leads to accumulation in bones, the kidneys and the nervous system. In consequence, the functioning of several relevant enzymes is disrupted. Stomach and intestinal ulcers, alopecia and polyneuropathy are considered classic symptons of thallium poisoning. Other symptoms include astral disorders, insomnia, paralysis, loss of body mass, internal bleeding, myocardial injury and, in consequence, death (Peter and Viraraghavan, 2005). Ingestion of more than 1.5 mg of thallium per kg of body mass may be fatal. Recent studies also indicate that high levels of Tl may be associated with an increased risk of low birth weight (Xia et al., 2016).

Świetlik et al. (2016) indicated that **13-30% of the Tl of coal ashes can be present in the water-soluble and weak-acid-soluble fractions**, considered mobile fractions or bioavailable fractions (Pettersen and Hertwich, 2008). Standard leaching tests in acidic and neutral conditions indicated that **thallium leaching from steel slags was negligible** (Proctor et al., 2000). **No information is, however, available on the long-term release patterns of Tl for these materials, and the Tl solubility could be further increased in the event that ashes or slags are further processed to more soluble derivates**. In the terrestrial environment, thallium is usually bound within the soil matrix, which considerably limits its transport, although dissolved thallium (soluble thallium salts) are susceptible to flushing and may be introduced into the aquatic environment. A high concentration of thallium in shallow soil also poses a notable threat due to possible uptake by plant roots and storage in plant biomass. Thallium concentrations in soils are closely correlated to **phytotoxicity** (Makridis and Amberger, 1996). Thallium has been used as rodenticide and insecticide and there are indications of inhibition of soil nitrification in the range of **1-10 mg kg^{-1} soil** (van Vlaardingen et al., 2005). Van Vlaardingen et al. (2005) also indicated low chronic ecotoxicological serious risk concentration values of 6.5 µg L^{-1}. Experimental bioconcentration factors obtained for aquatic organisms above **the limit value for the bioaccumulation criterion** (2 000 L/kg; as established by Regulation (EC) 1907/2006 have been reported (7 000 L/kg wet weight; Smith and Kwan, 1989). Hence, **thallium-based compounds exhibit a high tendency to accumulate in the environment, and the prolonged presence of thallium in terrestrial, aerial and aquatic systems may notably increase the exposure risks** (Karbowska, 2016).

In order to prevent thallium from entering the food chain and affecting the functioning of living organisms, and poisoning, **safe limits for Tl concentration in soils have been proposed by regulatory bodies** (1 mg Tl kg-1; Canadian Council of Ministers of the Environment, 2003) and **risk assessment studies** (1-2 mg Tl kg-1; van Vlaardingen et al., 2005; Xia et al., 2016). Considering the risks for human health and the environment due to the application of Tl-rich thermal oxidation materials, the proposal is to retain the assessment based on the maximal permissible soil concentration limit value of 2 mg Tl kg^{-1} soil.

Based on the information received from the STRUBIAS subgroup, **the proposed Tl limit value of 2 mg Tl kg^{-1} should not be a major limitation for any of the thermal oxidation materials & derivates that are targeted as ingredients for EU fertilising products**, including slags from the iron and steel industry.

Vanadium

Vanadium (Z = 23) is a hard, steel-grey metal listed as a transitional element (Imtiaz et al., 2015). Along with the transition elements Mo, W, Mn, Fe, Co, Ni, Cu and Zn, vanadium is an essential bioelement, but, in contrast to most of these elements, functional vanadium compounds have so far been detected only in the form of vanadium nitrogenases and vanadate-dependent haloperoxidases in a comparatively restricted number of organisms (Rehder, 2015). Vanadium can exist in a variety of oxidation states: − 1, 0, + 2, + 3, + 4, and + 5 (Larsson et al., 2013). In solution, under environmental conditions, mainly vanadium (IV) and vanadium(V) are present (Wanty and Goldhaber, 1992). Vanadium(IV) is an oxocation that occurs in moderately reducing environments. Under more aerobic conditions, the oxocation of vanadium (V), VO_2^+, prevails at a solution pH below 4, whereas the oxyanion vanadate (V), $H_2VO_4^-$, dominates above that pH (Baes and Mesmer, 1976). About 80% of the globally produced V is being used in the steel industry as an additive. As a result, **slags from the steel industry can show high V contents, with values documented that range from 54 mg kg^{-1} to 26 000 mg kg^{-1}** (Proctor et al., 2000; Cornelis et al., 2008; Reijkonen, 2017). According to the evaluation of CEN TC 260/WG 3 ‘Liming materials, 75% of the European slags used in agriculture however have a V content below 3 000 mg kg^{-1}. Also ashes from P-rich industrial sludges may show high contents, with values of up 1 206 mg kg^{-1} documented for mixed industrial and municipal sludges. Ashes exclusively derived from sewage sludges have rather low V contents of about maximum 100-200 mg V kg^{-1} (Krüger et al., 2014).

The bioavailability and toxicity of vanadium to soil microorganisms and plants has been reviewed by Larsson et al. (2013). The toxicity of vanadium to plants has mainly been studied in nutrient solution, and acute toxicity starts between 1 mg V L^{-1} and 5 mg V L^{-1} for the most sensitive species (Kaplan et al., 1990; Carlson et al., 1991; Imtiaz et al., 2015). Larsson et al. (2013) and Smith et al. (2013b) reported **soil vanadium toxicity thresholds (EC50) for higher plants**, ranging from 18 mg V kg^{-1} to 510 mg V kg^{-1} with a median of **91 mg V kg^{-1}** in five different soils. In her review, Smit (2012) proposed a long-term environmental risk limit for freshwater organisms of 1.2 µg V L^{-1}. Chronic ecotoxicological serious risk concentration values of 99 µg V L^{-1} have been documented by van Vlaardingen et al. (van Vlaardingen et al., 2005)

The V-release patterns are dependent on the chemical composition and stability of the CMC material. Similar to other metals, a high V release can be expected for thermal oxidation material derivates that typically show a high solubility under field conditions. Depending on the type of slag, the immediate V release from steel slags can also be high (Chaurand et al., 2006; Reijkonen, 2017). In the short term (15 days), 1.7% of the V present in blast oxygen furnace slag may be leached (De Windt et al., 2011). Reijonen indicated that up to 8-12% of the total V in blast oxygen furnace slag (14 000 mg V kg^{-1}) and high-vanadium slag (26 000 mg V kg^{-1}) was in the **water-soluble form**. Significant enrichment of the soil matrix to depths below the ploughing layer have been observed after the long-term application of V-rich steel slags in agricultural soil (Kuhn et al., 2006; Algermissen et al., 2016), indicating the V mobility within the soil matrix. Although V can effectively be immobilised in forest soils with a high organic matter content (Larsson et al., 2015b), the fate of released V depends on soil pH, redox potential and organic matter content (Gäbler et al., 2009; Larsson et al., 2015a; Reijonen et al., 2016). Moreover, in non-acidic soils, the soluble V exists predominantly as vanadium(V), considered more harmful to biota than vanadium(IV) (Larsson et al., 2015b). Overall, it is clear that a significant proportion of the V is potentially bioavailable or susceptible to be leached into recipient water systems (Larsson et al., 2015a; Reijkonen, 2017), and that transformation kinetics in soils may show spatial and temporal variations as a function of the soil properties and material ageing (Martin et al., 1998). Moreover, the **surface run-off** from slag-amended fields increases the risk of V release to aquatic organisms. This is attributable to the fact that desorption of V from the particle surfaces is favoured by increasing the solution to soil ratio (De Windt et al., 2011; Reijkonen, 2017). The risk limits associated with extreme V additions in soils are also defined in a report by the National Institute for Public Health and the Environment of the Netherlands (van Vlaardingen et al., 2005). In this work, the addition of 25 mg V kg^{-1} soil (total soil concentration of 67 mg kg^{-1}) was considered a serious risk for terrestrial ecosystems.

Considering the toxicity of vanadium to soil microorganisms and plants and the possible mobility of V in the soil matrix, the proposal is to retain the assessment based on the maximal permissible soil concentration limit value of 125 mg V kg^{-1} soil determined based on the soil screening values of EU Member States. Moreover, limiting the maximal V concentration in thermal oxidation materials will further limit ecotoxicity effects due to the leaching of vanadium.

For thermal oxidation materials & derivates derived from the eligible input materials, with the exception of basic oxygen furnace slag, there is no techno-scientific evidence that the proposed vanadium limit of 600 mg kg^{-1} will be a major limitation for the targeted STRUBIAS materials – particularly basic oxygen furnace slag – on the emergent market.

Conclusion and proposals

In summary, **it is indicated that the long-term application of thermal oxidation materials & derivates can involve Cr, Tl and V concentrations that could accumulate in soils and lead to the exceeding of soil quality standards established by EU Member States and possibly jeopardise environmental and human health protection.** The soil quality standards applied in this assessment for Cr, Tl and V are respectively 67%, 200% and 107% higher than the current median soil background concentrations for these elements in Europe. The assessment revealed that risks are only indicated for thermal oxidation materials & derivates that are derived from sewage sludges, other (industrial) waste, minerals, ores, ore concentrates and coal.

The toxicity threshold values (PNEC, EC50 values) for the different end-points in toxicity assessments are typically based on toxicity data from soils spiked in the laboratory with a soluble metal salt (Oorts et al., 2007; Li et al., 2011; Baken et al., 2012). Spiking soils with soluble metal salts in the laboratory causes a sudden stress to biotic communities and results in an unrepresentative exposure compared to the field, where metals are generally added gradually, can equilibrate and release for several years (Oorts et al., 2007; Baken et al., 2012). Moreover, some of the studies cited in the above assessment performed leaching tests under laboratory settings in short-term experiments, and not in undisturbed agricultural soils. Therefore, the actual toxicity of specific materials that are associated with slow-release and breakdown patterns under field conditions, particularly steel slags that are not further processed to derivates, may possibly be overestimated. For these materials, several processes obscure the relationship between soil total metal concentrations and toxicity, including the aging after metal amendment and long-term reactions taking place in the soil matrix (Smolders et al., 2009). It is recognised that the absence of (i) quantitative data on field-based toxicity thresholds for thermal oxidation materials & derivates, and (ii) suitable testing methods to assess the metal release patterns from the heterogeneous group of thermal oxidation materials & derivates hinders a risk-based approach for relevant end-points in toxicity assessments.

In view of limiting the accumulation of specific elements in soils to levels of concern for EU Member States, and considering the potential mobility of metals in thermal oxidation materials & derivates that disintegrate in order to fulfil their intended function as EU fertilising materials, the following proposal is made for the CMC requirements:

6. Thermal oxidation materials & derivates incorporated into the EU fertilising product shall:

a) contain no more than 3% Cl^- on a dry matter basis, only applicable when Cl^- is an unintentional constituent coming from the input material(s);
b) contain no more than 400 mg kg^{-1} dry matter of total Chromium (Cr), if derived from materials listed under point g), h) or j) of paragraph 1;
c) contain no more than 2 mg kg^{-1} dry matter of thallium (Tl), if derived from materials listed under point g), h), i) or j) of paragraph 1; and
d) contain no more than 600 mg kg^{-1} dry matter of vanadium (V), if derived from materials listed under point h) or j) of paragraph 1.

Note that the possibly high contents of Tl in auxiliary fuels (up to 76 mg Tl kg^{-1}) justify the testing requirements for this element. This stands in contrast to the elements Cr and V for which the lower concentrations in combination with the auxiliary nature of the use of the input materials classified under point h) will effectively ensure low Cr and V presence in the resulting ashes.

Possible limit values for Cr(total) have also been discussed in the Fertilisers Working Group. It was decided that where the EU fertilising product contains a total chromium (Cr) concentration above 200 mg/kg, information about the maximum concentration and exact source of total Cr should be included in the technical documentation to be established by the manufacturer as part of the conformity assessment procedures. Such documentation shall make it possible to assess the EU fertilising product's conformity with the relevant requirements, and shall include an adequate analysis and assessment of the risk(s). Hence, total chromium concentrations in EU fertilising products were considered an element of concern in view of human health safety, and additional provisions for waste and by-products may be justified. Here, an additional criterion on maximal Cr(total) contents in thermal oxidation materials & derivates is proposed based on the risk of Cr(total) accumulation in European soils after the long-term application of such materials.

Note that limit values for Cr(total) in fertilising materials or waste and residues used as fertilisers and soil improvers vary from 75 mg to 300 mg Cr kg^{-1} in Austria, Denmark, Finland, Flanders, the Netherlands and Sweden. Chromium (total) labelling requirements apply for fertilisers that exceed 300 mg Cr kg^{-1} in Germany. Vanadium limits form part of the legislative framework in Finland (400 mg kg^{-1}) and Austria (4 500 mg kg^{-1}).

Leaching of metals, metalloids, non-metals and halogens

Due to the combination of high bulk contents and solubility, the most prominently leached elements from ashes are Ca and SO_4^{2-}, followed by Cl, Na and K to a lesser extent. Nevertheless, the large number of trace elements that are leached at generally lower levels are of the highest concern due to their toxicity to **aquatic organisms** and the significant **human health hazard** they may entail for groundwater resources (Hjelmar, 1990; Izquierdo et al., 2008; Freire et al., 2015). The risks associated with the leaching of metals and metalloids were considered when deriving risk-based soil screening values that were used in the assessment above. **Moreover, leaching tests have shown that the environmental impact of most trace elements present in ashes upon their application or disposal is expected to be low** due to the relatively low water solubility of most trace metals and their strong tendency to adsorb to soil particles (Sheppard et al., 2009; Barbosa et al., 2011; Vassilev et al., 2013b). The leachability of chemicals is known to increase with the presence of organic matter (European Commission, 2006b; Secretariat of the Stockholm Convention on Persistent Organic Pollutants, 2008); the proposed 3% organic carbon limit for ashes and slags will thus also effectively reduce the leaching of metals. (STRUBIAS subgroup comments; Vamvuka et al., 2005; Skodras et al., 2006; Izquierdo et al., 2008; Vamvuka and Kakaras, 2011; Freire et al., 2015).

Barber et al. (2003) and Barbosa et al. (2011) indicated that the leachable cyanide concentrations in ash were low, and similar to those in biomass and soil. This is in line with observations that leachable cyanide in residual ash is < 1% of the mass of cyanide emitted to the atmosphere (Barber et al., 2003). Any dilution of ash with soil after ash applications gives rise to low, acceptable cyanide concentrations in leachate and run-off water from the field samples. Therefore, it is proposed that **cyanide content for this CMC should not be regulated**.

Therefore, it is concluded that the **leaching of above-mentioned minor and trace elements** from ashes is not of particular concern, and no **specific limits are proposed for the leachable fraction for the CMC thermal oxidation materials & derivates.**

5.4.5.2 PAHs, PCDD/F and dl-PCB

The combustion of plant and animal biomass, waste and other materials can cause the generation of persistent organic pollutants **(POPs), such as polycyclic aromatic hydrocarbons (PAHs), dioxins and furans (polychlorinated dibenzo-p-dioxins and dibenzofurans - PCDD/F)), and dioxin-like polychlorinated biphenyls (dl-PCBs).** The presence of POPs is thus a major issue of concern for thermal oxidation materials & derivates (Pitman, 2006; Insam and Knapp, 2011; Freire et al., 2015; Masto et al., 2015). Persistent organic pollutants are **toxic** chemicals that adversely affect human health and the environment around the world. They **persist** for long periods of time in the environment, and can accumulate and pass from one species to the next through the food chain. Because they can be **transported** by wind and water, most POPs generated in one country can and do affect people and wildlife far from where they are used and released. To address this global concern, the United States joined forces with 90 other countries and the European Union to sign a ground-breaking United Nations treaty in Stockholm, Sweden, in May 2001. Under the treaty, known as the Stockholm Convention, countries agreed to reduce or eliminate the production, use and/or release of particular POPs, and specified under the Convention a scientific review process that has led to the addition of other POP chemicals of global concern.

Most POPs present in the input materials destined for thermal oxidation materials are **destroyed** during thermal oxidation, but the **formation of new POPs may occur** because of incomplete combustion or formation in the flue-gas path at levels that depend both on the fuel composition, combustion conditions and flue-gas treatment (Lavric et al., 2004; Enell et al., 2008; Masto et al., 2015). Persistent organic pollutants are subsequently distributed between flue-gas and ash streams, but the distribution of POPs among fly ashes and bottom ashes is different between PAHs (higher in bottom ashes) and PCDD/F and BCBs (higher in fly ashes (Gulyurtlu et al., 2007; Lopes et al., 2009). During combustion, POPs are formed via organic precursors like phenols and lignin, via *de novo* reactions in the presence of particulate carbon and chloride or by pyrosynthesis (high-temperature gas phase formation) (Lavric et al., 2004; Gulyurtlu et al., 2007; Shibamoto et al., 2007; Van Caneghem et al., 2010).

The **technology used in modern incineration and biomass plants**, including grinding the feedstock into very fine particles, a short residence time of the particles in the boiler and optimum fuel to air ratio, **ensures nearly complete combustion to low organic C levels in the ashes and slags while preventing the creation of such pollutants and their accumulation in the ash at hazardous concentrations**. Indeed, the findings of tests that were performed on various ashes in Europe confirm this assessment (European Commission, 2006b).

Polyaromatic hydrocarbons (PAHs)

Polyaromatic hydrocarbons are typical products of incomplete combustion and should be lower in concentration in all residue streams the more efficient the combustion control and the burnout are. **A review of the data indicates that the concentrations of PAHs in the boiler and filter residues are, apart from some exceptions, lower than those in bottom ashes (Vehlow et al., 2006)**. This behaviour is to be expected since the burnout of the particulate matter in the flue-gas is typically higher than that of the bottom ashes as is documented by the lower total organic carbon in those residues. In principle, for residues from modern waste thermal oxidation plants, no major problems concerning PAHs should be expected (European Commission, 2006b; Vehlow et al., 2006). However, **the database for this class of compounds is too weak to support that statement, and higher PAH values have also been observed in practice** (European Commission, 2006b; Vehlow et al., 2006; Van Caneghem and Vandecasteele, 2014). Concentrations of specific and highly toxic compounds, for example of benzo(a)pyrene, were not found. Rey-Salgueiro et al. (2016) also indicated that the concentrations of **benzene, toluene, ethylbenzene, the ortho-, para- and meta-xylenes and styrene** (BTEX + S) in all samples analysed in their study **were low for bottom and fly ashes** with maximum concentrations of 0.3 mg kg^{-1}.

Polychlorinated dibenzo-p-dioxins and dibenzofurans (PCDD/F)

PCDD/Fs are emitted from thermal processes involving **organic matter and chlorine as a result of incomplete combustion or chemical reactions**. It is well known that transient combustion conditions and especially the start-up and shutdown procedure are characterised by elevated PCDD/F levels in the raw gas (Hunsinger et al., 2002); PCDD/Fs **form in the temperature window of 200 °C to about 450 °C** (Lundin and Marklund, 2005). **Feedstock characteristics** also play a major role in PCDD/F formation during biomass combustion. PCDD/Fs are always formed during wood combustion via precursors such as phenols and lignin, or via *de novo* reactions in the presence of particulate carbon and chlorine. High emission levels can also be expected from burning treated wood and wood waste (Lavric et al., 2004). Temperatures in the range of 1 100-1 200 °C are considered to be most efficient for destroying halogenated hazardous compounds, including PCDD/PCDF (Secretariat of the Stockholm Convention on Persistent Organic Pollutants, 2008). Nonetheless, depending on the composition of the fly ash, lower temperatures have also been shown to enable the destruction of PCDD/F (Lundin and Marklund, 2005). **The concentration of PCDD/Fs in the fly ash is typically higher than in the bottom ash** (Lavric et al., 2004; Vehlow et al., 2006). The bottom ash PCDD/F inventory correlates well with the organic carbon content, and PCDD/F levels are usually below 20 ng WHO toxicity equivalents kg^{-1} when the organic carbon content in the ashes is below 1% (Vehlow et al., 2006). Nonetheless, **the adsorption of PCDD/Fs on fly ashes can be relatively high,** and depends on the presence of elementary carbon or soot particles in the fly ashes, since inorganic surfaces have a poor adsorption potential for PCDD/F (Vehlow et al., 2006). PCDD/F concentrations in the range of 100 ng to 10 000 ng WHO toxic equivalents kg^{-1} have actually been found in fly ashes from modern waste incineration plants for municipal solid waste, sewage sludge and poultry litter (Vehlow et al., 2006; Rigby et al., 2015; Egle et al., 2016). Moreover, the database on PCDD/F for sewage sludge mono-incinerators and biomass combustion plants is limited.

Dioxin-like polychlorinated biphenyls (dl-PCBs)

Dioxin-like polychlorinated biphenyls are organic chlorine compounds with the formula C12H10−xClx. Dioxin-like PCBs are formed **through a similar mechanistic pathway as PCDD/Fs**, and maximum PCB formation occurs at temperatures around 350 °C (Lemieux et al., 2001). Dioxin-like PCBs are more efficiently destroyed if higher combustion temperatures are used (e.g. above 1 200 °C); however, lower temperatures (e.g. 950 °C) together with appropriate conditions of turbulence and residence time have also been found to be effective for dl-PCB removal. Van Caneghem and Vandecasteele (2014) indicated low dl-PCB ranges in ashes, with average concentrations for ashes derived from refuse-derived fuels and sewage sludge samples of 8.57 µg and 4.90 µg kg^{-1} dry matter, respectively. Dioxin-like PCB data for poultry litter ashes (Rigby et al., 2015) and sewage sludge ashes and their derivates (Egle et al., 2016) also indicated very low dl-PCB concentrations. Hence, **ashes of a low organic C content and PAHs as result of a complete combustion process show low PCDD/F concentrations**. Moreover, a close relationship between PCDD/F and dl-PCB concentrations has been observed (Lemieux et al., 2001; Pandelova et al., 2006; Li et al., 2018).

EU and national regulations in EU Member States on POPs in fertilising materials

Commission Regulation (EU) No 756/2010, amending Regulation (EC) No 850/2004 of the European Parliament and of the Council **on persistent organic pollutants as regards Annexes IV and V, lays down limit values for wastes for thermal processes**. The limit values in that Regulation are 15 µg kg^{-1} for PCDD/F and 50 mg kg^{-1} for dl-PCBs.

The POP content in ash-based fertilising products is regulated as follows in different EU Member States (Haglund and Expertsgroup, 2008):

- In **Denmark** (BEK1636 of 22 December 2006), the PAH content only has to be analysed if the loss on ignition (LOI) is > 5%. The limit value for PAHs is 3 mg/kg dry ash (12 mg/kg dry ash in the proposed update).
- **Portugal's** legislation: DL 103/2015 for non-harmonised fertilisers imposes, only for fertilisers containing industrial sludge or their mixtures with sewage sludge, limits for PCDD/F (100 ng TEQ/kg), PAHs (6 mg/kg) and dl-PCB (0.8 mg/kg). Ashes (EWC 100101, 100102 and 100103) may be incorporated in several fertilising types without limits for these organic pollutants.
- In **Germany**, the sum of dioxins and dl-PCBs (WHO-TEQ 2005) is limited to 30 ng kg^{-1} dry matter with a reduced limit of 8 ng kg^{-1} dry matter for fields producing fodder and if the soil is not ploughed after the application of the fertiliser, except where maize is grown.
- The application of ashes in **Austria** is regulated through 'Rückführung von Pflanzenaschen auf Böden' [Recycling of plant ashes to the soil]. In the Salzburg area there is 'Amt der Salzburger Landesregierung Abt4/Abt16 Richtlinien 2006 – Richtlinien für die Aufbringung von Asche aus Holzfeuerungsanlagen auf landwirtschaftlich genutzte Böden'. These regulations indicate that if the total of unburnt C is above 5%, PAH_6 should be < 6 mg kg^{-1} and PCDD/F < 20 ng WHO TE kg^{-1}.
- For the **UK**, a **Quality Protocol for Poultry Litter Ash** (End-of-Waste Criteria for the Production and Use of Treated Ash from the Incineration of Poultry Litter, Feathers and Straw - Waste and Resources Action Programme and Environment Agency) is available with a limit value for PCDD/F of 10 ng TEQ/kg.

Conclusion and proposals

Modern thermal oxidation plants with good combustion control produce bottom wood ashes with inventories of POPs that are not much higher than those encountered in European soils (Lavric et al., 2004; Pitman, 2006; Vehlow et al., 2006; Rohr et al., 2015). Hence, even without post-combustion treatment (e.g. 3R process, acid extraction followed by secondary thermal treatment) for the abatement of organic compounds, acceptable levels of POPs can be achieved for the proposed input materials if **stable combustion conditions are established and no unburnt residues remain**. **Bottom ashes** are typically more enriched in **PAHs**, whereas **PCDD/Fs and dl-PCBs are abundantly adsorbed to the fly ash fraction**.

Nonetheless, some thermal oxidation materials & derivates show **high PAH and PCDD/F values.** Dioxin-like **PCB levels are typically low in the ash fraction, with maximum levels typically below 0.2 mg kg^{-1}, and common levels for well-operated plants of below**

0.01 mg kg^{-1} (Vehlow et al., 2006) because appropriate conditions of temperature, turbulence and residence time have been found to be effective for PCB incineration **(European Commission, 2017b)**. This conclusion is **confirmed by the (mostly confidential) data that were received from the STRUBIAS subgroup**. Moreover, low quantities of dl-PCBs are found in most municipal waste streams and also in some industrial wastes (European Commission, 2017b). Wastes with large proportions of dl-PCBs, however, generally only arise from specific PCB collection and destruction programmes (European Commission, 2017b), and will therefore be classified as hazardous waste that is not eligible as input material for this CMC. Finally, levels of dl-PCBs and PCDD/F are closely correlated due to their similar nature of formation in thermal oxidation processes (Vehlow et al., 2006). National legislation also typically focuses on PAHs and PCDD/Fs, and - with the exception of Germany that has a limit for combined PCDD/F and dl-PCB content in fertilisers - none of the national legislative frameworks impose additional limits for dl-PCBs in the ashes and slags.

Based on the data collected, the following technical requirement for this CMC is proposed:

4. The thermal oxidation materials shall have:
   - no more than 6 mg/kg dry matter of PAH_{16}[25], and
   - no more than 20 ng WHO toxicity equivalents/kg dry matter of PCDD/F[26].

5.4.5.3 Other organic chemical pollutants

Other pollutants of concern may be present in sewage sludge (e.g. **pharmaceuticals, personal care products, nanomaterials**) and animal by-products (e.g. **hormones, veterinary medicines, metabolites**). However, the concentration of these compounds is generally less than 1% halogenated organic substances, expressed as chloride. Thermal destruction is generally considered to be an effective method for the removal of these pollutants that occur in diluted form in non-hazardous waste streams to levels below environmental or human health concern (UNEP, 2004). The ability of temperatures exceeding 850 °C to destroy or inactivate organic contaminants with high destruction efficiencies have been measured for aldrin, dieldrin, HCB, DDT, BSE, pharmaceutical compounds and other organic pollutants (UNEP, 2004; INTECUS GmbH, 2013).

5.4.5.4 Biological pathogens

Heat acts to kill or inactivate by denaturation of essential proteins (enzymes, viral capsids) and nucleic acids. From the biological pathogens that could be present in the eligible input

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[25] Sum of naphthalene, acenaphthylene, acenaphthene, fluorene, phenanthrene, anthracene, fluoranthene, pyrene, benzo[a]anthracene, chrysene, benzo[b]fluoranthene, benzo[k]fluoranthene, benzo[a]pyrene, indeno[1,2,3-cd]pyrene, dibenzo[a,h]anthracene and benzo[ghi]perylene.

[26] van den Berg M., L.S. Birnbaum, M. Denison, M. De Vito, W. Farland, et al. (2006) The 2005 World Health Organization Re-evaluation of Human and Mammalian Toxic Equivalency Factors for Dioxins and Dioxin-like Compounds. Toxicological sciences: an official journal of the Society of Toxicology 93:223-241. doi:10.1093/toxsci/kfl055.

materials, spore-forming bacteria such as *Bacillus* and *Clostridium* are the most resistant to heat inactivation. Of the non-spore-forming waterborne and foodborne enteric pathogens, enteric viruses are the most heat-resistant, followed by bacteria and protozoa. Parvoviruses are among the most heat-resistant heat viruses. The thermal destruction has been studied in great detail by the food industry because of the importance of this process in killing pathogenic bacteria and preventing foodborne spoilage. The findings from this indicated that much lower temperatures are required for the dry heat deactivation of biological pathogens than those required for the production of thermal oxidation materials. **Above temperatures of 120 °C, minimal thermal death times are required to inactivate biological pathogens**, even under dry conditions. The thermal oxidation process efficiently causes the **thermal death of all biological microorganisms** present in the selected input materials (Gerba, 2015). Therefore, no specific measurements on biological pathogens are proposed as criteria.

5.4.5.5 Radioactivity

Concerns over the potential radioactivity of ashes stem from the expectation that natural or manufactured radioactivity present in the input material can become concentrated in ash upon combustion. This is majorly a concern for **wood ashes, as trees may accumulate radioactive nuclides over prolonged periods of time**. Overall, the concern has been less for natural radiation (which is generally considered to be negligible), and more for anthropogenic radionuclides that may be present at higher levels in plants and soils in areas that have **experienced nuclear fallout** (Pitman, 2006).

The principal radionuclide of concern is **cesium-137** (^{137}Cs), with a half-life (time taken for radioactivity to decay to 50% of the original levels) of 30.2 years. The half-life of this isotope results in contamination remaining for many decades after the original event; significant quantities were released into some regions of Europe from the 1986 Chernobyl accident (Steinhauser et al., 2014). Based on the data available in the biodat database (ECN, 2017), the activity concentration of ^{137}Cs in wood ashes varies between 81 Bq/kg and 4 460 Bq/kg (limited dataset of 15 samples of unknown geographic origin), with more than 50% of the samples having activity values above 1 000 Bq/kg. In order to protect human health safety aspects of workers, the risk assessment of the International Atomic Energy Agency (IAEA, 2003) recommended a unified ^{137}Cs limit value of 1 000 Bq kg^{-1} for timber and wood products that is applicable to all the considered conditions, i.e. local (contaminated areas), regional, national and international (IAEA, 2003). Hence, there is a possible risk associated with ^{137}Cs radioactivity in wood ashes.

The main legal instrument for radiation sources and protection from these is **Council Directive 2013/59/Euratom[27] laying down basic safety *Standards* for protection against the dangers arising from exposure to ionising radiation**. The Directive provides a legal

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[27] Council Directive 2013/59/Euratom of 5 December 2013 laying down basic safety standards for protection against the dangers arising from exposure to ionising radiation, and repealing Directives 89/618/Euratom, 90/641/Euratom, 96/29/Euratom, 97/43/Euratom and 2003/122/Euratom, Official Journal of the European Union (OJ L13, 17.01.2014, p. 1-73).

framework for the regulatory control of practices involving radiation sources and provisions for the protection of workers and the public exposed to these radiation sources that show activities above specific threshold values, i.e. 100 Bq/kg for ^{137}Cs. Member States are responsible for establishing legal requirements and an appropriate regime of regulatory control for radioactive exposure based on a risk assessment.

Sweden is the only EU Member State that has set restrictions in place to limit radioactive exposure through ashes (Regulation SSI FS 2005.1). The limit has been set at 10 000 Bq/kg for ashes applied on forest land and **500 Bq/kg for agriculture** and for reindeer grazing land.

As potential risks associated with radioactivity in wood ashes are regulated through Directive 2013/59/Euratom, **no specific provisions or activity concentration limit values are proposed for the CMC thermal oxidation materials & derivates**.

5.4.5.6 Emissions

The mechanisms that lead to emissions from the handling and application of fertilising during material are outlined in Section 5.3.5.4.

One of the mechanisms may generate **airborne dusts and particulate matter emissions**. Epidemiological and toxicological studies have shown particulate mass < 2.5 µm, < 10 µm and < 100 µm (PM2.5, PM10 and PM100) comprises fractions with varying types and degrees **of health effects for workers** that are involved in the handling of ashes. This suggests a role for both the chemical composition (such as transition metals and combustion-derived primary and secondary organic particles) and physical properties (size, particle number and surface area). Exposure to particles from biomass may be associated not only with respiratory, but also with cardiovascular health issues (United Nations Economic Commission for Europe (UN ECE), 2009). A particularly relevant aspect of thermal oxidation materials & derivates is that they contain **silica** among the ash-forming material in significant quantities (Section 15.2.1). While Si may be required to sustain plant growth, respirable free crystalline silica (i.e. quartz) is associated with silicosis (a nodular pulmonary fibrosis), lung cancer, pulmonary tuberculosis, and other airway disorders (NIOSH—Publications Dissemination, 2002). In view of the potential risk associated with airborne dusts and particulate matter emissions, the following aspects are relevant to consider:

a) The dustiness of a powder product, defined as the propensity of a material to generate airborne dust during its handling (Lidén, 2006), not only depends on the **intrinsic physical properties of the material but also on the handling scenario.**
b) Exposure to ash results in exposure to respirable free silica, but no well-designed epidemiological study has established an association between silica exposure from this source and adverse health effects (Meij et al., 2000; Hicks and Yager, 2006). Some research has demonstrated that the lack of health effects may be because the free quartz in combusted material is vitrified and unable to interact with biological targets (Van Eijk et al., 2011). The tendency of silica in biomass ash to fuse has also

> been observed (Van Loo and Koppejan, 2008). This feature, in conjunction with the understanding that in general biomass has a lower silica content than conventional solid fuel, indicates that the silica in ash is unlikely to pose an occupational health concern (Meij et al., 2000; Rohr et al., 2015).

In line with the discussion provided in Section 5.3.5.4, it is concluded that **correct classification and labelling as foreseen in EU legislation allows downstream users to assess the risk with associated airborne dust emissions and other emissions to air during the handling and application of certain products, and to take the necessary measures to prevent any potential adverse impacts in the event that a risk is identified. It is indicated that the provisions in the EU Fertilising Products Regulation ((EU) 2019/1009) on labelling and European regulations are sufficiently effective to control for any adverse impacts associated with emissions during the handling and application of thermal oxidation materials & derivates.**

5.4.5.7 Occupational health

Council Directive 89/391/EEC on the introduction of measures to encourage improvements in the safety and health of workers at work seeks to adequately protect workers and encourages improvements in occupational health and safety in all sectors of activity, both public and private. The Directive also promotes workers’ rights to make proposals relating to health and safety, to appeal to the competent authority and to stop work in the event of serious danger. No further legal requirements are therefore proposed.

5.4.6 Physico-chemical properties

5.4.6.1 Dry matter content

Biological pathogens are destroyed during the combustion process, so there is no risk of biological re-contamination of the thermal oxidation materials after combustion. Therefore, no further criteria on moisture content are proposed.

5.4.6.2 pH

Reactive ash with high pH and high salt dissolution rates may cause burns to the vegetation, for instance to *Sphagnum* mosses which have been found to be especially sensitive. Reactive ashes with a very high or low pH are not suitable for land application as they will induce a pH shock effect on soil fauna and flora. Nonetheless, it is noted that ashes with a high pH may fulfil a function as liming material, and that other liming materials (e.g. aglime) can also be applied as EU fertilising materials. Moreover, CMC materials can be physically mixed with other CMC with the resulting changes in the pH of the PFC material. Therefore, no requirements on the pH value have been proposed for thermal oxidation materials & derivates.

5.4.6.3 Granulometry

Agglomeration is used as a means of improving product characteristics and enhancing processing conditions. In addition to these benefits, agglomeration also solves a number of problems associated with ash fines:

- significant dust reduction/elimination and mitigation of product loss;
- improved handling and transportation;
- improved application and use;
- increased water infiltration in the soil as there is no risk of the blocking of soil pores by small water-insoluble particles.

At the same time, it should be noted that thermal oxidation materials & derivates are CMCs, meaning that they can be mixed with other CMCs prior to becoming a PFC (e.g. compost).

Moreover, it is noted that the particle form (granule, pellet, powder, or prill) of the product shall be indicated on the **label** of solid inorganic macronutrient fertilisers (see labelling requirements in the EU Fertilising Products Regulation ((EU) 2019/1009)). Therefore, no **criteria on granulometry or particle size distribution are proposed at CMC level.**

5.4.7 Handling and storage

It is proposed that physical contact between input and output materials must be avoided after the thermal oxidation process, including during storage.

5.5 Pyrolysis & gasification materials

5.5.1 Scope delimitation and possible uses

This CMC comprises organic materials that have **undergone a thermochemical conversion in an oxygen-limiting environment**, resulting in the production of carbonaceous materials like char, charcoal, soot and graphite (see Section 5.1.3). Such black carbon materials, or combinations thereof, are often referred to as 'biochar' when applied as a soil amendment. The scope of this CMC aims to cover carbonaceous materials that are not fully oxidised, and thus fall in between thermal oxidation materials (Section 5.4, with a maximum organic C content of 3%) and non-carbonised biomass along the biomass transformation spectrum. These materials are the result of different production processes that take place in an oxygen-limiting environment, including gasification, dry pyrolysis and wet pyrolysis (also referred to as hydrothermal carbonisation). In line with the other STRUBIAS CMCs, a reference will be made to the production process and the name **pyrolysis & gasification materials will be used to cover the spectrum of production processes that take place under oxygen-limiting conditions**. The materials produced are of a very heterogeneous nature, and their properties also depend on the time-temperature profiles applied and the feedstock used.

The variability in biomass feedstock and production process conditions means that **pyrolysis & gasification materials cover a very heterogeneous product property spectrum that may fulfil a variety of fertilising functions** when applied on the soil (Neves et al., 2011).

- Pyrolysis & gasification materials may be used as a **nutrient source for plants**. Pyrolysis & gasification materials may contain inorganic plant nutrients. Macronutrients such as P, K, Mg and Ca are largely conserved in the end material (60% to 100%, Gaskin et al., 2008), and their bio-available nutrient content is generally correlated to total concentration (Ippolito et al., 2015). Phosphorus availability is, however, not controlled by the total P content, but is likely determined by the coordinated cations present (Al, Fe, Ca, Mg) in the feedstock (Wang et al., 2012b). The loss of N is highly variable during pyrolysis (0-80%, depending on the process conditions applied), but the pyrolysis/gasification process may transform a large share of N to complexes that are unavailable to plants (Biederman and Harpole, 2013). These observations illustrate the overall importance of the feedstock source for the potential of pyrolysis & gasification materials to supply nutrients to plants. The pyrolysis of feedstock from animal production systems (bone material, manure) and human waste treatment (sewage sludge) creates nutrient-rich end materials, while most plant-based pyrolysis & gasification materials have lower quantities of macronutrients (Section 15.3.1).
- Pyrolysis & gasification materials may in some cases, independent of the feedstock they are produced from, act as a **soil improver** (Chia et al., 2015). The addition of pyrolysis & gasification materials to soils may lead to unique interactions that influence the **soil's physical properties** such as porosity, particle size distribution, density and packing. Plant yield can then be impacted through, for example, the availability of water and air in the vicinity

of the plant root, or the **stimulation of soil microbial activities in the plant rhizosphere** (Jeffery et al., 2015). It should be noted that the effect of adding nutrient-poor pyrolysis & gasification materials without complementary fertilisation to soils of temperate climates does not, on average, increase plant yield (Biederman and Harpole, 2013). This may be attributed to the fact that most European soils have good physical properties and the addition of C-rich pyrolysis & gasification materials to soils might stimulate microbial nutrient scavenging, ultimately decreasing nutrient availability for plants in the short term (Biederman and Harpole, 2013).

- There are claims that some pyrolysis & gasification materials can increase the **efficiency of mineral fertiliser** due to their ability to retain nutrients within the soil matrix as a result of the increase in surface area and cation and anion exchange capacity (Ippolito et al., 2015; Aller, 2016). Also, the addition of pyrolysis & gasification materials to soil may improve root traits, particularly root mass density and root length density (Brennan et al., 2014).
- Studies in soil-less systems indicate that some pyrolysis & gasification materials can provide nutrients (Ruamrungsri et al., 2011; Locke et al., 2013), reduce nutrient leaching (Beck et al., 2011; Altland and Locke, 2012) and improve both the biological (Graber et al., 2010a) and physical properties of growing media as a whole (Dumroese et al., 2011). The use of pyrolysed materials might, therefore, represent a promising development for **soil-less growing media** components (Barrett et al., 2016; Kern et al., 2017). The potential use of pyrolysis & gasification materials for soil-less growing media was also supported by various participants at the STRUBIAS Kick-off Meeting and supported by the feedback received via the ensuing questionnaires.
- Pyrolysis at high temperatures removes acidic functional groups and increases the ash content, ultimately causing increased **basicity** of pyrolysis & gasification materials (Novak et al., 2009; Cantrell et al., 2012). Because of their basic pH, pyrolysis & gasification materials have been used to ameliorate acidic soil conditions, thus they could serve as a **liming agent** (Hass et al., 2012; Kloss et al., 2012). While an increase of soil pH might have beneficial effects for the plant, it should be noted that the liming equivalent of pyrolysis & gasification materials is typically much lower than that of commonly applied liming products (Ippolito et al., 2015; Jeffery et al., 2015). As a matter of fact, it is unlikely that pyrolysis & gasification materials will meet the liming requirements at PFC level in the EU Fertilising Products Regulation ((EU) 2019/1009) (Feedback on questionnaries received from the STRUBIAS subgroup; Ippolito et al., 2015). It may thus not be economically feasible for farmers to use pyrolysis & gasification materials in crop production solely for pH adjustment due to the high cost (Collins, 2008; Galinato et al., 2011). Similarly to thermal oxidation materials, it is proposed to **label the**

neutralising value if pyrolysis & gasification materials are used as a CMC in quantities > 50% in the PFC fertiliser (PFC 1), soil improver (PFC 3), growing medium (PFC 4) and plant biostimulants (PFC 6).

- Finally, pyrolysis & gasification materials are used as a compost additive and as admixtures in NPK fertiliser **blends** (Steiner et al., 2015). The utilisation of the absorptive binding capacity of pyrolysis & gasification materials to alter the nutrient-release patterns of other fertilising products is often referred to as the 'charging' of pyrolysis & gasification materials. It should, however, be noted that, even without the admixing of other CMCs onto pyrolysis & gasification materials, the end material of a pyrolysis/gasification process should have a demonstrated agricultural value (see Section 4.2.1).

It is concluded that the inclusion of pyrolysis & gasification materials as a CMC in the EU Fertilising Products Regulation ((EU) 2019/1009) enables potential applications **for PFC 1 (fertiliser), PFC 3 (soil improver), PFC 4 (growing medium) and PFC 6 (non-microbial plant biostimulant).** The EU Fertilising Products Regulation ((EU) 2019/1009), however, does not place any restriction on the use of CMCs for any of the PFCs. Hence, the information presented here is only informative to introduce pyrolysis & gasification materials in view of their possible intended uses and associated application rates.

The efforts on the standardisation of the technical specifications of pyrolysis & gasification materials have resulted in voluntary industry-driven product standards and harmonisation actions. Especially relevant are the quality standards that have been developed by the International Biochar Initiative (IBI) (International Biochar Initiative, 2016b) and the European Biochar Certificate (EBC, 2012). **These voluntary standards form the basis for many legislative initiatives in the European Union and the European Free Trade Association (see Bachmann et al., 2016 and Meyer et al., 2017 for an excellent overview).**

5.5.2 Input materials

Pyrolysis is the thermochemical decomposition of **organic material**, which is why the input material list is confined to materials that originate from or contain matter from plants, animals, wastewater treatment sludges, and certain organic industrial by-products that are generated during the processing of organic materials (e.g. paper sludge and distillers grain).

Pyrolysis & gasification materials derived from **plant-based materials, biowaste and certain animal by-products** (e.g. inedible animal by-products such as bone material) form the basis of the currently used input materials for pyrolysis/gasification processes in voluntary standardisation schemes (EBC, 2012; International Biochar Initiative, 2016a) and national legislation (Meyer et al., 2017). The presence of organic contaminants in these input materials is limited, as is the concentration of inorganic metals and metalloids such as Cd, Cr, Pb, Hg and Ni (Gaskin et al., 2008; Uchimiya et al., 2012; Beesley et al., 2015; Someus, 2015; ECN, 2017). Also, specific residues from the bio-energy industry, vegetable waste from the food processing industry, and fibrous vegetable waste from pulp production and from production of paper from pulp could be considered due to the low contaminant levels, other than those included in the proposed testing scheme, in these feedstocks. The manufacturing of pyrolysis & gasification materials may also be an attractive alternative for **manure or processed manure**, especially in those situations where no local disposal is available and the feedstock is applied on land in a non-sustainable manner that negatively impacts upon the environment. Also, other animal by-products, such as meat and bone meal or animal bones, can be pyrolysed (e.g. 3R agrocarbon process).

At present, very few research results are available on the **behaviour during the pyrolysis/gasification process of the many organic contaminants** (e.g. phthalates, surfactants present in cleaners and detergents (e.g. linear alkylbenzene sulphonates (LAS), di-2-ethylhexyl phthalate (DEHP), and personal care products, pharmaceuticals and endocrine-disrupting compounds) **that are possibly present in sewage sludge and other highly contaminated materials** (Lehmann and Joseph, 2015; Aller, 2016). Whereas organic compounds can be degraded under oxidative conditions at high temperatures, the necessary techno-scientific evidence is lacking that demonstrates their removal under oxygen-limiting conditions. It is known that stringent time-temperature pyrolysis profiles (> 550 °C, > 20 minutes) induce a weight loss in pyrolysis & gasification materials due to burning out of organic compounds (Deydier et al., 2005a; Koutcheiko et al., 2007; Ro et al., 2010; Marculescu and Stan, 2012), but the knowledge base of studies that assessed the proportional removal of specific organic pollutants is limited and restricted to a limited amount of model pollutants (Sütterlin et al., 2007; Weiner et al., 2013; Hoffman et al., 2016; Ross et al., 2016; vom Eyser et al., 2016; Liang et al., 2017). Although a significant reduction in contaminant levels is typically achieved, especially at high temperatures (Lizarraga et al., 2007), limitations in the potential of dry and wet pyrolysis processes to remove organic pollutants can be seen for, for instance, nonylphenol, chlorinated aromatic fractions and specific veterinary antibiotics (Weiner et al., 2013; vom Eyser et al., 2016). For other contaminants, it is suggested that compounds that are not fully removed may be immobilised to the solid fraction (Sütterlin et al., 2007). It should be noted that the cited literature includes publications that cover both wet pyrolysis (‘hydrothermal carbonisation’) and dry pyrolysis, that the extent of degradation may vary across techniques, and that dry pyrolysis with severe time-temperature profiles and high-temperature gasification processes may show greater contaminant removal rates (Hoffman et al., 2016; Ross et al., 2016) than the low-temperature wet pyrolysis processes (Weiner et al., 2013; vom Eyser et al., 2016). Moreover, the formation of transformation products that may exhibit higher toxicity or persistency than the parent compound can occur. Such processes might be particularly relevant because the presence of elementary carbon with a high specific surface area and soot particles result in a high adsorption potential for organic contaminants. Hence, the mechanisms, nature and soil residence times of any decay products of organic contaminants that could be formed remain unclear, with only a few works that have addressed this topic (Sütterlin et al., 2007; Weiner et al., 2013; Ross et al., 2016; vom Eyser et al., 2016). The absence of those organic contaminants in pyrolysis & gasification materials derived from contaminated feedstocks is, therefore, not sufficiently demonstrated through robust and extensive techno-scientific research. **Based on the precautionary principle and in view of the broad list of emerging contaminants in human-derived waste streams (Petrie et al., 2015), it is justified to exclude highly contaminated feedstocks (e.g. sewage sludge, municipal solid waste, hazardous waste) from the positive input material list to ensure human health and environmental safety.** Contaminated input materials such as sewage sludge and mixed municipal solid waste are also absent on the list of allowed input materials according to voluntary standardisation schemes for pyrolysis & gasification materials (EBC, 2012) and national legal frameworks (Meyer et al., 2017). The lack of ensured emerging contaminant removal is especially problematic for specific input material streams that may contain a broad range of emerging contaminants and that are currently not regulated in the framework of the EU Fertilising Products Regulation ((EU) 2019/1009); enabling a market entry for such materials may lead to a reduced level of environmental and human health safety relative to the counterfactual scenario of waste incineration.

The JRC recommends that the scientific knowledge base be further developed in order to demonstrate that the use of EU fertilising products derived from (specific) pyrolysis & gasification materials does not present an unacceptable risk to human, animal or plant health, to safety or to the environment. Article 42 of the EU Fertilising Products Regulation ((EU) 2019/1009) foresees that Annexes I to IV to the Regulation can be amended for the purposes of adapting them to technical progress in the light of new scientific evidence. Therefore, the current proposal to exclude sewage sludge from the eligible input material list for CMC pyrolysis & gasification materials could possibly be revised once robust and extensive techno-scientific evidence underpins the safe use of (specific) pyrolysis & gasification materials derived from sewage sludge.

Moreover, there is a substantial **risk of the accumulation of non-volatile pollutants such as inorganic metals and metalloids in the pyrolysis & gasification materials** as these mostly remain in the solid phase and become concentrated during the production process. In contrast to thermal oxidation materials & derivates, no post-combustion manufacturing processes have been described that are able to remove inorganic metals and metalloids from the final material. Pyrolysis & gasification materials obtained from **contaminated organic input materials such as sewage sludge** (He et al., 2010; Hossain et al., 2010; Gascó et al., 2012; Méndez et al., 2012; Van Wesenbeeck et al., 2014; Lu et al., 2016), **mixed municipal solid waste** (Henrich et al., 1999; Vassilev and Braekman-Danheux, 1999; Vassilev et al., 1999) and **chemically treated wood** (Helsen et al., 1997; Lievens et al., 2009; ECN, 2017) often appear unable to comply with the limits suggested for these elements at PFC level. Inorganic metals and metalloids like Cd, Pb and Ni encountered in such pyrolysis & gasification materials typically exceed the limit levels set for fertilisers and soil improvers at PFC level in the EU Fertilising Products Regulation ((EU) 2019/1009) (Shackley et al., 2013). Therefore, the potential of pyrolysis & gasification materials derived from sewage sludge, municipal waste and industrial by-products for the internal fertilising market appears limited.

The choice of a positive input material list for pyrolysis & gasification materials may help to avoid pollution risks which cannot be easily addressed by limits for organic pollutants due to their inherent heterogeneous chemical nature. A **positive input material list is thus proposed** to ensure the production of pyrolysis & gasification materials associated with acceptable risks for adverse environmental or human health impacts that can be enforced through straightforward and cost-effective compliance schemes:

1. An EU fertilising product may contain materials exclusively obtained through the thermochemical conversion under oxygen-limiting conditions of one or more of the following input materials:

h) derived products referred to in Article 32 of Regulation (EC) No 1069/2009 for which an end-point in the manufacturing chain has been determined in accordance with the third subparagraph of Article 5(2) of that Regulation;

i) animal by-products, the products derived from which are referred to in Article 32 of Regulation (EC) No 1069/2009 and have an end-point in the manufacturing chain determined in accordance with the third subparagraph of Article 5(2) of that Regulation;

j) living or dead organisms or parts thereof, which are unprocessed or processed only by manual, mechanical or gravitational means, by dissolution in water, by flotation, by extraction with water, by steam distillation or by heating solely to remove water, or which are extracted from air by any means, except:
  - materials originating from mixed municipal waste,
  - sewage sludge, industrial sludge or dredging sludge, and
  - animal by-products or derived products falling within the scope of Regulation (EC) No 1069/2009;

k) vegetable waste from the food processing industry and fibrous vegetable waste from virgin pulp production and from production of paper from virgin pulp;

l) bio-waste within the meaning of Directive 2008/98/EC resulting from separate bio-waste collection at source, other than those included above;

m) residues from the production of bioethanol and biodiesel as referred to in Directive 2009/28/EC of the European Parliament and of the Council of 23 April 2009 on the promotion of the use of energy from renewable sources and amending and subsequently repealing Directives 2001/77/EC and 2003/30/EC[28]; or

n) pyrolysis/gasification additives which are necessary to improve the process performance or the environmental performance of the pyrolysis/gasification process, provided that the additives classify as intermediates within the meaning of Regulation (EC) No 1907/2006 and with the exception of:
  - those listed under points a) to f),
  - waste within the meaning of Directive 2008/98/EC,
  - substances or mixtures which have ceased to be waste in one or more Member States by virtue of the national measures transposing Article 6 of Directive 2008/98/EC,

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[28] OJ L 140 5.6.2009, p. 16.

- substances formed from precursors which have ceased to be waste in one or more Member States by virtue of the national measures transposing Article 6 of Directive 2008/98/EC, or mixtures containing such substances,
- non-biodegradable polymers, and
- animal by-products or derived products within the scope of Regulation (EC) No 1069/2009.

The total concentration of all additives must not exceed 25% of the total input material fresh weight.

[Note: The exclusion of a material from a lettered item does not prevent it from being an eligible component material by virtue of another lettered item.]

Note that the input material list is similar to the input material list proposed in the first draft of this document (STRUBIAS Interim Report of May 2017). It is noted that this proposed input material list is also generally **in line with the positive input material list proposed by the European Biochar Certificate** (EBC, 2012).

5.5.3 Production process conditions

Pyrolysis processes generate three main materials: gases (syngas), condensable vapours (oil) and solid (char-rich) materials. This implies that pyrolysis can be used for two specific aims: (1) the recovery of energy embedded in the feedstock through the combustion of syngas or oil fractions, and (2) the production of solid pyrolysis & gasification materials that can possibly be applied on agricultural land. As there is some degree of complementarity between the different phases from the pyrolysis of biomass, it is proposed that the end material can be obtained from pyrolysis facilities that are specifically designed for the **purpose** of producing pyrolysis & gasification materials for further **fertiliser use** as well as from a process aimed at serving **energy recovery purposes** as long as product quality conditions are fulfilled.

The pyrolysis/gasification process is also used in the **chemical industry** to produce non-food products, for example to produce activated carbon, charcoal, methanol, and other chemicals from wood, to convert ethylene dichloride into vinyl chloride to make PVC, to produce coke from coal, to turn waste plastics into usable oil, and for transforming medium-weight hydrocarbons from oil into lighter ones like gasoline. Pyrolysis is also used in the creation of nanoparticles, zirconia and oxides utilising an ultrasonic nozzle in a process called ultrasonic spray pyrolysis. These specialised uses of pyrolysis may be called various names, such as dry distillation, destructive distillation, or cracking. As the solid end materials of these processes do not have agricultural value, pyrolysis & gasification materials from the chemical industry are not considered for the purposes of this document (see Section 5.5.2 – eligible input materials).

The proposal is to include pyrolysis plants that operate as **stand-alone** installations as well as integrated systems in line with the principle of technological neutrality. After all, the integration of pyrolysis units into an integrated system will not necessarily affect end material quality.

5.5.3.1 Pre-processing

For dry pyrolysis, input materials with a high **moisture content** are typically subjected to mechanical processes such as solid-liquid separation, thickening, dewatering, or drying treatments. The **energy and nutrient density of the feedstock can be increased** by applying techniques (e.g. hydrothermal carbonisation, fast pyrolysis, anaerobic digestion, composting) to produce intermediate nutrient carriers. Increasing the dry matter content of the feedstock decreases the volumes that have to be processed in the pyrolysis reactor, as well as the supplementary energy requirements. Pyrolysis co-products (syngas, pyrolysis oil) could also pass through an energy recovery system at the pyrolysis plant, whereby the energy can be (partly) recovered in the form of heat or electricity. The heat can be used for heating the pyrolysis reactor or for **pre-drying** the input material prior to pyrolysis. Such practices increase the possibility for long-distance transportation from several regionally distributed conversion plants to a few central large-scale pyrolysis plants. Wet pyrolysis or hydrothermal carbonisation involves the thermochemical dehydration of biomass in a closed vessel under autogenous pressure of water. Under these conditions and a residence time of several hours, the biomass is chemically dehydrated and its carbon content concentrated. Hence, the chemical composition becomes less polar, enabling a significant reduction in moisture content (to ~ 50%) by mechanical techniques, such as filter pressing.

While sometimes the output materials obtained through wet pyrolysis or fast pyrolysis are marketed as end materials that can be applied on agricultural land, some of the output materials might lack material properties that are in line with their intended use as a soil improver (Kambo and Dutta, 2015; Aller, 2016) or may contain high amounts of organic micropollutants that cause negative effects on plant growth and productivity (Becker et al., 2013; Wang et al., 2016). **Note that, according to the draft proposals, none of the pyrolysis spectrum variants are excluded as core processes in the present study, as long as the end material meets the product quality standards.**

Also, the physical material properties of pyrolysis & gasification materials can be modified by applying a '**physico-chemical activation process**' (Kambo and Dutta, 2015), a process similar to the commercial production of activated carbon. Activation of pyrolysis & gasification materials can significantly increase the surface area thanks to the development of internal porous structures within a biomaterial (Gratuito et al., 2008). Physical and chemical activation methods are the two common techniques used for the activation of chars (Chia et al., 2015). In both techniques, char is exposed in a pyrolysis reactor to elevated temperatures in the presence of activation agents such as CO_2 or steam, which develops and improves the porous structure through the removal of C atoms or volatiles (Rodríguez-Reinoso and Molina-Sabio, 1992; Alaya et al., 2000). Activation through chemical reagents such as zinc salts, metal hydroxides (KOH, NaOH) or phosphoric acid can also induce very high pore densities (Lillo-Ródenas et al., 2007; Lin et al., 2012). In such a case, it is proposed to consider the physico-chemical activation process as part of the core pyrolysis/gasification process (and not as a post-production process) and any materials that are added to the reactor as **additives** (see Section 5.5.3.3). The STRUBIAS subgroup indicated, however, that such production techniques are unlikely to be deployed for materials that will be used as soil amendments due to the high production costs.

No major limitations on any pre-processing steps are proposed as long as the positive input material list is respected. This implies that the input materials, and a combination thereof, may be physically mixed, screened, sized and chemically reacted. The following provision is therefore foreseen in paragraph 1 of the proposals for the legal requirements on the input:

In addition, EU fertilising products may contain pyrolysis & gasification materials obtained through thermochemical conversion under oxygen-limiting conditions of any material listed in points (a) to (g), or combination thereof, processed by manual, mechanical or gravitational means, by solid-liquid fractionation using biodegradable polymers, by dissolution in water, by flotation, by extraction with water, by steam distillation or by heating solely to remove water, by composting, or by anaerobic digestion.

5.5.3.2 Core process

The pyrolysis technology spectrum covers a broad range of production process conditions, with **slow pyrolysis** processes (300-700 °C, long residence time in the reactor) being the most common for the production of pyrolysis & gasification materials that can be applied on agricultural land. Nevertheless, other processes such as **fast-pyrolysis** (300-700 °C, short residence time in the reactor), **gasification** (low-oxygen environment, temperatures > 500 °C), **wet pyrolysis** (sometimes referred to as hydrothermal carbonisation - HTC, in subcritical water conditions, 175-300 °C) and **torrefaction** (200-320 °C) also fall under the umbrella of the pyrolysis technology spectrum. Hence, it is proposed to permit their application as long as the output material meets the product quality criteria. **With product quality being of primordial importance, it is proposed not to impose any constraints on the pyrolysis/gasification process, as long as the output material meets the product quality criteria**.

It has been indicated that it is challenging to predict the molecular structure and agronomic value of pyrolysis & gasification materials based on the specific temperature profile applied because of the complex and **little-understood interactions** of heating temperature, heat exposure time, feedstock properties, mineral admixtures, reaction media, etc. (Kleber et al., 2015). Therefore, it does not appear suitable **to set strict criteria for production conditions provided that the pyrolysis material has a demonstrated agronomic value and does not pose a risk for human health and the environment**. Instead, it is proposed to impose minimum requirements so as to ensure that the pyrolysis & gasification materials incorporated into EU fertilising materials have undergone a thermochemical conversion process that is in line with the scope of this CMC, as follows:

The thermochemical conversion process of the input materials shall take place under oxygen-limiting conditions:

- in such a way that a temperature of at least 180 °C for at least 2 seconds is reached in the reactor, and
- in a pyrolysis/gasification reactor which only processes input materials referred to in paragraph 1 above, excluding materials that are knowingly contaminated with material streams not listed in paragraph 1 unless such contamination is unintentional, only results in trace levels of exogenous compounds, and constitutes a one-off incident.

5.5.3.3 Additives

Similar to ashes, non-biomass materials are sometimes added as **a catalyst or additive** to the pyrolysis/gasification process with the aim of changing the relative proportions or quality of the altering solid, liquid and gaseous compounds produced during the pyrolysis/gasification process (Jensen et al., 1998; Wang et al., 2010; Li et al., 2014), at addition rates up to 22%. The supply of additives shall serve to improve and facilitate the pyrolysis/gasification process, and should, rationally, not be used to improve the nutrient content of the pyrolysis & gasification materials obtained. It is, therefore, proposed to enable **a maximum of 25% of additives defined as intermediates within the meaning of Regulation (EC) No 1907/2006, with the exception of waste, materials which have ceased to be waste, polymers and animal by-products.** An intermediate is defined as ‘a substance that is manufactured for and consumed in or used for chemical processing in order to be transformed into another substance’. Note that minerals are also classified as substances. The use of this terminology will prevent inert materials from being added for the manufacturing of CE fertilising products with the sole intention of reducing the contaminant levels of the final CE product.

Therefore, it is proposed to add following text to the proposals for the legal requirements on the input materials for pyrolysis & gasification materials:

g) pyrolysis/gasification additives which are necessary to improve the process performance or the environmental performance of the pyrolysis/gasification process, provided that the additives classify as intermediates within the meaning of Regulation (EC) No 1907/2006 and with the exception of:
    - those listed under points a) to f),
    - waste within the meaning of Directive 2008/98/EC,
    - substances or mixtures which have ceased to be waste in one or more Member States by virtue of the national measures transposing Article 6 of Directive 2008/98/EC,
    - substances formed from precursors which have ceased to be waste in one or more Member States by virtue of the national measures transposing Article 6 of Directive 2008/98/EC, or mixtures containing such substances,
    - non-biodegradable polymers, and
    - animal by-products or derived products within the scope of Regulation (EC) No 1069/2009.

5.5.3.4 Post-processing

Pyrolysis & gasification materials leaving the pyrolysis reactor may undergo further post-processing steps with the intention being the following:

a. **Agglomerate** the material as pellets or granules through adding binder solutions (Bowden-Green and Briens, 2016) or pelletising with additives as with wood flour, polylactic acid and starch (Dumroese et al., 2011).
b. Increase the chemical and physical stability by **washing and rewetting** with water (Schulze et al., 2016).
c. Alter product granulometry through **mechanical treatments** such as screening and sizing.

It is proposed to allow these post-processing steps. No supplementary requirements related to these post-processing techniques have to be included at CMC level.

5.5.4 Agronomic value

Pyrolysis & gasification materials can be applied with two different objectives in agricultural ecosystems: (1) to increase the primary production of agroecosystems as a **fertilising product**, and (2) to **impact upon the global C balance,** greenhouse gas emissions and climate change (Lehmann and Joseph, 2015). It should be clear that the primary focus of this work is on its use as a fertilising product, as defined in Article 2 of the EU Fertilising Products Regulation ((EU) 2019/1009).

5.5.4.1 Carbon stability

Considering the intended uses of pyrolysis & gasification materials as a soil amendment, **pyrolysis & gasification materials** should have:

- product properties and compound release dynamics that have a positive influence on plant growth and that by no means cause plant toxicity;
- physico-chemical properties (e.g. surface area, porosity, ion exchange capacity) that have the potential to positively influence air, water and microbial nutrient dynamics in the soil.

Nutrient-rich pyrolysis & gasification materials that are applied as fertilisers should also have sufficient quantities of one or more of the following macronutrients (P, N, K, Mg and Ca) available for plants in the short term.

The **stability of the carbon** present in the pyrolysis material is a determining factor for the potential of pyrolysis & gasification materials to be applied on soils because of its close relationship with the following:

a. **Toxicity and air quality:** Volatile organic compounds[29] with a boiling point lower than the pyrolysis temperature might, depending on the extent and nature of interaction between pyrolysis gases and solids, end up in the pyrolysis material (Spokas et al., 2011; Buss et al., 2015). Moreover, recondensation and trapping of volatile organic compounds that are normally associated with the pyrolysis liquid fraction in the pores of pyrolysis & gasification materials is possible (Spokas et al., 2011). During the subsequent handling and use of pyrolysis & gasification materials, these compounds can be emitted to the atmosphere, to the soil matrix and to the water bodies as water-soluble leachates (Spokas et al., 2011; Smith et al., 2013a; Buss and Mašek, 2016). **Plant toxicity and adverse effects on soil and aquatic organisms and plants have been documented due to the VOC release from pyrolysis & gasification materials** (Titirici et al., 2008; Spokas et al., 2011; Becker et al., 2013; Buss and Mašek, 2014; Buss and Mašek, 2016; Smith et al., 2016). Dutta et al. (2016) reported 76 different types of VOCs that exist in the pyrolysis & gasification materials, some of which are highly toxic (e.g. benzene, toluene). The volatile organic compounds impact upon various plant and microbial responses by mimicking plant hormones and impacting seed germination, herbivore resistance, and nutrient uptake (Almeida et al., 2009; Insam and Seewald, 2010; Dutta et al., 2016). Volatile short-carbon-chain alkanes with less than 11 carbon atoms are the most phytotoxic while non-volatile long-carbon-chain hydrocarbons (nC12 to nC20) in soil have no apparent toxic effect on germination of perennial ryegrass and are generally attacked most readily by microorganisms (Siddiqui and Adams, 2002; Wang et al., 2016). The possible presence of potentially harmful compounds, such as benzene, toluene, ethylbenzene, xylenes, phenols, volatile fatty acids, and polycyclic aromatic hydrocarbons in pyrolysis & gasification materials with a H/C ratio > 0.7, highlights the importance of directing the biochar production process towards stable compounds (Ghidotti et al., 2017a). **A negative correlation has been observed between the abundance of volatile organic compounds and molar H/C ratios** (Budai et al., 2014; Aller, 2016; Conti et al., 2016; Ghidotti et al., 2017b). The pyrolysis & gasification materials assessed by Ghidotti et al. with a molar H/C ratio < 0.70 did not release VOCs at ambient temperatures, and showed no presence of specific toxic volatile organic compounds such as benzene and toluene (Ghidotti et al., 2017a). Also Smith et al. (2016) indicated that pyrolysis & gasification materials should be produced with more severe time-temperature profiles in order to increase the carbon stability and to limit its effects through the leaching of dissolved organic matter into the environment. Hence, it is concluded that **targeting process conditions towards the formation of stable pyrolysis &**

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[29] For pyrolysis & gasification, the term ‘volatile matter’ refers to the proportion of carbon that is easily removed (labile), but not necessarily as a gas. This class of compounds includes, for instance, pyrazines, pyridines, pyrroles and furans.

gasification materials significantly reduces the risks associated with the release of VOCs towards the atmosphere, the soil matrix and the percolation water.

Nonetheless, the **design and configuration of the pyrolysis reactor is also critical**. More specifically, it should be ensured that pyrolysis oils and vapours are promptly evacuated from the pyrolysis reactor and even temperature profiles should be ensured to prevent pyrolysis oil vapours from recondensing onto the solid material in ‘cold zones’ of the reactor (Buss and Mašek, 2016; Smith et al., 2016). However, in the case of contamination by (recondensed) pyrolysis vapours, not only would VOCs be trapped in the pyrolysis & gasification materials, but also PAHs (Buss et al., 2015) and other persistent organic pollutants (PCDD/F, dl-PCB), compounds for which strict limits have been proposed in the compliance scheme for this CMC (see Section 5.5.5.2). Hence, it is indicated that, especially in pyrolysis/gasification reactors with a low technological readiness level often used for scientific research, significant VOC contents could be trapped in the solid material, but that one could effectively exclude those materials from entering the EU fertilising market by including POPs as part of the compliance scheme for pyrolysis & gasification materials.

Therefore, **it is concluded that VOC concentrations in pyrolysis & gasification materials with a high carbon stability (molar $H:C_{org} < 0.7$) and with minimal pyrolysis liquid contamination are generally below limits that could cause phytotoxic effects or adverse impacts on aquatic organisms through leaching under realistic field application conditions** (Buss et al., 2015; Smith et al., 2016; Ghidotti et al., 2017a).

b. **Physical properties:** Structure, porosity, pore size distribution, total amount of pores, surface area, and adsorption capacity are the physical properties of pyrolysis & gasification materials most frequently described in the literature. Rutherford et al. (2004) found evidence that aliphatic C in feedstocks **must first be converted into fused-ring, aromatic C before porosity can develop**. Fused-ring structures of aromatic C provide a matrix in which micropores can be created. Moreover, most of the surface area and thus cation exchange capacity derives from pores created during the pyrolysis/gasification process (Schimmelpfennig and Glaser, 2012). Interplanar distances of aromatic C forms decrease with increased ordering and, thus, the surface area per total volume increases alongside the aromaticity. However, upon heating to temperatures in the range of 800-1 000 °C, the C crystallites reorient themselves into parallel sheets of C atoms, causing the destruction of the porosity of the material (Brown et al., 2015). The high porosity and surface area of pyrolysis & gasification materials may also provide a habitat for microbial communities in the soil.

c. **Nutrient properties:** Soil microorganisms are largely homeostatic, implying that they need to assimilate energy and nutrient sources in relatively fixed

> proportional quantities (Griffiths et al., 2012). Therefore, the addition of pyrolysis & gasification materials that contain large amounts of non-stabilised, labile C to agricultural soils but low amounts of available nutrients may actually cause microorganisms to **immobilise soil nutrients, especially nitrogen**, in order to enable microbial homeostasis. Such an effect is particularly of concern for pyrolysis & gasification materials that contain at least some nitrogen in a largely plant-unavailable form (see Section 5.5.4.2). Hence, the microbial immobilisation of plant nutrients is sometimes observed when pyrolysis & gasification materials with a high labile C content are added to the soil, hence nutrient availability to plants is decreased (Bruun et al., 2012; Nelissen et al., 2012; Schimmelpfennig et al., 2014; Reibe et al., 2015). While such an effect is mostly temporary and can potentially be overcome by applying the pyrolysis material some months prior to planting, it should be considered that it may be challenging to convince farmers to use pyrolysis & gasification materials and pay for the product under competitive market conditions where products are available that have a guaranteed economical return within a much shorter time frame. Hence, in order to safeguard short-term returns of increased plant yield after the addition of pyrolysis & gasification materials to the soil, the pyrolysis & gasification materials should be characterised by C atoms that are present in a stabilised form. Moreover, higher emissions of greenhouse gases after the application of pyrolysis & gasification materials with a low C stability in the soil have been observed (Maestrini et al., 2015), which are most likely the result of increased microbial activity due to the easy degradability of C. In contrast, slow-pyrolysis & gasification materials were found to be more stable in the soil and showed a reduced effect on GHG emissions (Kambo and Dutta, 2015).

Hence, the extent to which the C in pyrolysis & gasification materials has been transformed into energetically stable aromatic ring structures contributes decisively to the agronomic value of the pyrolysis & gasification materials (Schimmelpfennig and Glaser, 2012**).** The carbonisation of the input materials is a complex process in which many reactions such as dehydrogenation, hydrogen transfer and isomerisation take place concurrently. Consequently, there is great interest in methods that are able to characterise in a simple and effective manner the proportion of C in condensed ring structures relative to total C. By far the most common, economical and straightforward approach used is to assess elemental ratios of H, C and O. In general, molar H:C ratios decrease with increasing heat treatment temperature from ~1.5 to a level significantly below 0.5 for pure compounds such as lignin and cellulose as well as for more complex biomass. Similarly, O/C ratios decline with the duration and intensity of heat treatments (Kleber et al., 2015). Hence, **the elemental ratios are excellent and robust tools to show an estimate of the general structural characteristics of pyrolysis & gasification materials** (Kleber et al., 2015). Elemental ratios of O:C, O:H and C:H have been found to provide a reliable measure of both the extent of pyrolysis and level of oxidative adjustment of pyrolysis & gasification materials (Crombie et al., 2013; Aller, 2016; Xiao et al., 2016). The H:C ratio value especially is a reflection of the basic and elemental structure of the pyrolysis

& gasification materials, which further constitutes the different pore size and surface area of pyrolysis & gasification materials. Xiao et al. (2016) established, for instance, a quantitative relationship between H:C atomic ratio and pyrolytic temperature, and aromatic structure.

The following criterion is proposed:
The pyrolysis & gasification materials shall have a molar ratio of H/organic C of less than 0.7, with testing to be performed in the dry and ash-free fraction for materials that have an organic C content of < 50%.

The reference to the testing being performed in the dry and ash-free fraction for materials that have an organic C content of < 50% is explained by the fact that ashes contain a low organic C content, but possibly increased H contents due to the presence of, for instance, hydroxides and water in those materials.

Based on the review of Aller (2016), it can be observed that most pyrolysis & gasification materials of different feedstocks (lignin-rich, manure/waste, black carbon (other), nuts/shells and cellulose-rich) meet the H/C conditions proposed, with the exception of wet pyrolysis & gasification materials ('hydrochars') with a high ash content and lignin-rich materials with a low ash content .

5.5.4.2 C-rich and nutrient-rich pyrolysis & gasification materials

Pyrolysis & gasification materials derived from plant-based input materials such as crop and wood residues are characteristically C-rich materials that are mostly applied as soil improvers, liming materials, growing media or plant biostimulants, or possibly as a carrier for NPK fertilisers. The intention for their application is mostly to improve soil characteristics such as organic matter, soil pH, physical properties such as water-holding capacity, or a combination of different soil properties. These pyrolysis & gasification materials typically have a carbon content of > 50% (see Section 15.3.1).

Specific pyrolysis & gasification materials derived from mineral-rich eligible input materials (manure, animal bone materials, food and kitchen waste, etc.) could also serve to supply plant-available nutrients such as P, K and Ca. Pyrolysis & gasification materials from these input materials show a nutrient content with P_2O_5 + CaO + K_2O + MgO + N that ranges from 15% to more than 75% (Section 15.3.1). The plant availability of nutrients in pyrolysis & gasification materials varies widely for the different elements and is also dependent on production process conditions (Camps-Arbestain et al., 2015; Ippolito et al., 2015):

- Phosphorus: The availability of P present in pyrolysis & gasification materials depends primarily on P-solubility. At temperatures < 760 °C, P availability is likely controlled by pH and the coordinated cations present (Al, Fe, Ca, Mg) (Wang et al., 2012b; Ippolito et al., 2015); it is therefore largely dependent on the input material applied in the pyrolysis production processes. Ca-P and Mg-P complexes, dominant in pyrolysis & gasification materials from the eligible mineral-rich input materials manure and bone, are mostly plant-available, although some contrasting results of high temperatures (> 760 °C) on P-availability have been observed (Kercher and Nagle, 2003; Ippolito et al., 2015).

- Potassium: Due to the high solubility of K-containing salts, K in pyrolysis & gasification materials has been shown to be readily available (Yao et al., 2010; Gunes et al., 2015).
- Calcium and magnesium: It is indicated that the availability depends on the presence of other elements and compounds such as P and silicates, with the elements being relatively less available under basic conditions for Si-rich pyrolysis & gasification materials, such as those derived from plant materials (Angst and Sohi, 2013). Calcium and magnesium in pyrolysis & gasification materials obtained from nutrient-rich input materials are, however, largely available, especially in plant rhizospheres of a somewhat lower pH than bulk soils (Martins Abdao dos Passos et al., 2015).
- Sulphur: The availability of S depends on whether it is available as C-bonded S, ester-S or sulphate-S. Sulphur in mineral-rich pyrolysis & gasification materials produced at a temperature of 550 °C was found to be non-crystalline, and is therefore readily available to plants as it easily dissolves (Yao et al., 2010; Churka Blum et al., 2013).
- Nitrogen: Low extractable mineral N concentrations in pyrolysis & gasification materials have been observed. As a result of charring, aromatic and heterocyclic N-ring structures are formed that are considered mostly unavailable to plants (Almendros et al., 1990; Almendros et al., 2003). Nonetheless, some recent publications also indicate the presence of hydrolysable N fractions in pyrolysis & gasification materials. The N fertilisation value of pyrolysis & gasification materials under realistic application scenarios is considered low to moderate at best (Camps-Arbestain et al., 2015).

Pyrolysis & gasification materials from eligible input materials are mostly expected to show a good plant nutrient availability as mineral-rich eligible input materials (animal by-products of categories 2 and 3 as well as specific bio-wastes) are depleted in Al, Fe and Si. Therefore, nutrients will be likely bound into labile complexes such as K-containing salts and Ca and Mg phosphate salts.

It is **proposed to classify the pyrolysis & gasification materials in a non-exclusive manner depending on their carbon content** to evaluate the accumulation of metals/metalloids in soils. Such distinction is useful as the STRUBIAS subgroup indicated that the application rates are typically larger for C-rich pyrolysis & gasification materials (up to 20 tonnes material ha^{-1} yr^{-1}) than for nutrient-rich pyrolysis & gasification materials (up to 5 tonnes material ha^{-1} yr^{-1}). As a result, the intended use and elemental composition has an influence on the load of material that will be applied, and will consequently impact upon the permissible limit values for relevant contaminants (see Section 5.5.5).

The minimum C-content set by the European Biochar Certificate (EBC, 2012) to differentiate between C-rich and nutrient-rich pyrolysis & gasification materials in the assessment that evaluates the accumulation of metals/metalloids in soils is as follows:

C-rich pyrolysis & gasification materials: total C ≥ 50% by mass of dry matter

and

Nutrient-rich pyrolysis & gasification materials: total C < 50% by mass of dry matter

5.5.4.3 Salinity

Salinity is a generic term used to describe elevated concentrations of **soluble salts** in soils and water. Comprised primarily of the most easily dissolved ions - sodium (Na) and chloride (Cl), and to a lesser extent calcium, magnesium, potassium, and sulphate - salinity in the environment adversely impacts water quality, soil structure, and plant growth (Pichtel, 2016). Although minimal accumulations (some in trace amounts) are required for normal biological function, excess salinisation is becoming one of the leading constraints on crop productivity and could reduce the diversity of salt-intolerant plant and epiphyte species in natural ecosystems. Reactive ash with high dissolution rates of salts may cause burns to the vegetation and excess sodicity can cause clays to deflocculate, thereby lowering the permeability of soil to air and water.

a. **Chloride.** Feedstocks such as grasses, straws and food waste (which contains sodium chloride, i.e. salt) can be a source of chloride. Other potential sources of chloride in feedstocks include biomass that has been exposed to salt (such as crops or trees grown near seashores). The Cl- contents for pyrolysis & gasification materials are limited, but documented values in the ECN database vary from 0.2% to 3.6% (ECN, 2017). Therefore, a significant risk is present for crops when pyrolysis & gasification materials are applied during prolonged periods of time. The Finnish legislation on the use of ashes in forest ecosystems contains a limit value of 2% for chloride (Haglund and Expertsgroup, 2008). In the EU Fertilising Products Regulation ((EU) 2019/1009) (Annex III to the proposal – Labelling requirements), it is stated that the phrase 'poor in chloride' or similar may only be used if the chloride (Cl-) content is below 3%. Nonetheless, a labelling requirement cannot prevent a product high in chloride from having adverse impacts on the environment. Therefore, **it is proposed to set a 3% limit value for chloride for pyrolysis & gasification materials.**

b. **Sodium** plays a role as a 'functional nutrient', with a demonstrated ability to replace potassium in a number of ways for vital plant functions, including cell enlargement and long-distance transport, and is even a requirement for maximal biomass growth for many plants (Subbarao et al., 2003). Considering the relative low Na contents in pyrolysis & gasification materials, no limits are proposed for the Na content of pyrolysis & gasification materials, but the total Na content should be declared on the label.

c. At present, there are no reliable methods other than leaching tests to characterise pyrolysis & gasification materials with regard to the speed of salt dissolution in the field. One way of estimating the salinity of pyrolysis & gasification materials is to measure the conductivity in water extracts. This gives a total measurement of the dissolution of salts from the pyrolysis material and indicates the risk of acute damage to vegetation. Given the labelling provisions for the closely related parameter Cl, it is, however, not proposed to add any further criteria or labelling requirements for **electrical conductivity**.

5.5.4.4 Boron toxicity

Boron (B) is a very common element that may be present **in some pyrolysis & gasification materials**, and is **readily water-soluble from pyrolysis & gasification materials** (Gunes et al., 2015). Although boron is an essential nutrient in plants at low concentrations, it becomes toxic in many plants at concentrations only slightly higher than the optimal range (Ayers and Westcot, 1985; Sartaj and Fernandes, 2005). Boron toxicity depends, however, not only on the concentration, but also on the form, since the element can occur in an undissociated form as boric acid ($B(OH)_3$), which the plant does not absorb. To the best of our knowledge, no research has been published on the forms of boron that are leached from pyrolysis & gasification materials, and their potential toxic effects for plants.

The B levels in pyrolysis & gasification materials are typically $\leq$ 100 mg B kg^{-1} (ECN, 2017), and are about one order of magnitude lower than most mineral and organo-mineral fertilisers (Kratz et al., 2016).
Recent evidence indicates that human B intake from food and water in the EU are below the tolerable upper intake level (EFSA, 2004), and that increased human B uptake is even promoted to enhance health due to the beneficial effects at low B concentrations (Nielsen, 2014; Pizzorno, 2015). Moreover, the most extensive and most recent dataset for rivers/catchments or regions in the EU contains consistently low B values (Heijerick and Van Sprang, 2004). This observation is in line with a publication by Neal et al. (2010) examining changes in boron concentrations for the Thames catchment (UK) over a decade. The observed decreases correspond with the timing and extent of an EU-wide trend for B reduction in industry and domestic usage, such as the reductions in the direct application of sewage sludges (Schoderboeck et al., 2011). In the EU, the harmonised classification as toxic to reproduction category 1B (H360: may damage fertility or the unborn child) is regulated pursuant to Regulation (EC) No. 1272/2008 - classification, labelling and packaging of substances and mixtures. According to this Regulation, consumer products that have concentrations of > 1% of some boron compounds should be labelled accordingly as toxic.

For all these reasons, **it is proposed not to set a limit for the B content of pyrolysis & gasification materials at CMC level.**

5.5.5 Environmental and human health safety aspects

Based on the feedback received from the STRUBIAS subgroup, it has become clear that modern pyrolysis plants show a **high technological readiness level** and that both pyrolysis material properties and the environmental footprint of their production **are highly dependent on the technological readiness level of pyrolysis plants and the type of feedstocks**.

Similar to thermal oxidation materials & derivates, contaminants present in pyrolysis & gasification materials may originate from **the feedstock source** used (e.g. inorganic metals and metalloids, veterinary medicines) or **can be formed** by the thermochemical processes used to make pyrolysis & gasification materials (e.g. persistent organic pollutants such as PAHs, PCDD/Fs, dl-PCBs).

5.5.5.1 Metals and metalloids

This section considers concerns associated with the exposure **to alkali, alkaline earth metals, transition metals and other metals.** While some of them are plant micronutrients, the potential **dissolution and accumulation to toxic levels of these inorganic metals and metalloids present in pyrolysis & gasification materials requires a more in-depth risk assessment.** Metal or metalloid species may be considered 'contaminants' if their presence is unwanted or occurs in a form or concentration that causes detrimental human or environmental effects.

In broad terms, the addition of pyrolysis material has been indicated to reduce the metal and metalloid concentrations in plant tissues, possibly because of metal adsorption on the reactive surface of the pyrolysis material (Peng et al., 2018). The effects of pyrolysis & gasification materials on residual metals and metalloids in the soil have, however, not been considered in this assessment as the long-term fate of the adsorbed contaminants remains unknown. Instead, this analysis is focused on the supplementary addition of metals and metalloids to soils through the application of pyrolysis & gasification materials.

Aluminium, iron and manganese

The assessment for aluminium, iron and manganese in pyrolysis & gasification materials is largely similar to that performed for thermal oxidation materials & derivates (see Section 5.4.5.1). Although the leaching of Al from pyrolysis & gasification materials is somewhat higher than for thermal oxidation materials & derivates (Hernandez et al., 2011), the leaching of Al is still about one to two orders of magnitude lower than for Ca. Given the critical importance of the Ca/Al ratio of the leachate to influence plant phytotoxicity (Godbold et al., 1988), no major risks are indicated for Al. Iron leaching from pyrolysis & gasification materials is indicated to be very low, similar to thermal oxidation materials (Hernandez et al., 2011). The Mn contents in pyrolysis & gasification materials are typically lower than for CMC thermal oxidation materials due to the presence of organic carbon for this CMC, but some gasification processes could produce ashes that have properties, including organic C content, that are similar to CMC thermal oxidation materials & derivates. In conclusion, **no specific criterion is proposed for Al and Fe in pyrolysis & gasification materials, but it is proposed that EU fertilising products derived from CMC pyrolysis & gasification materials that have a Mn content > 3.5% shall be labelled.**

Assessment of the potential accumulation of trace metals/metalloids in soil

Metals and metalloids present in feedstock will mostly likely **concentrate in pyrolysis & gasification materials**, although methods such as the selective removal of metal-concentrated ashes and high-temperature pyrolysis might possibly reduce the contaminant levels in pyrolysis & gasification materials (Shackley et al., 2013). Relative to thermal oxidation materials & derivates, little information on the content of metals and metalloids in pyrolysis & gasification materials is available (Section 15.3.2).

Possible environmental and human health risks due to the presence of inorganic metals and metalloids (As, Be, Ba, Cd, Cr, Cu, Hg, Mo, Ni, Pb, Sb, Se, V and Zn) in pyrolysis material should therefore be evaluated. An overview of the inorganic metals and metalloids present in pyrolysis & gasification materials has been compiled in Section 15.3.2, based on the information found in the scientific literature and the completed questionnaires from the STRUBIAS subgroup.

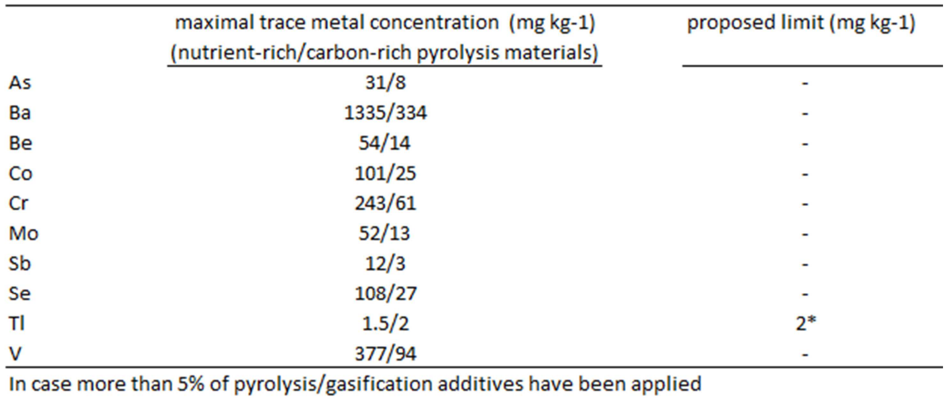
Some inorganic metals and metalloids are already regulated for different PFCs in the EU Fertilising Products Regulation ((EU) 2019/1009). Specifically, limit values for Cd, Cr(VI), Hg, Ni and Pb have already been put forward in the EU Fertilising Products Regulation ((EU) 2019/1009) for the different PFCs where pyrolysis & gasification materials might be used as ingredients. Also, regulating Zn and Cu at PFC level is being discussed, which is why these elements are not included in this assessment at CMC level. **Based on the information presented, the present assessment is restricted to As, Ba, Be, Co, Mo, Sb, Se and V.**

Considering the large overlap in input materials for thermal oxidation processes and pyrolysis/gasification processes, a similar approach for inorganic metals and metalloids will be considered, focusing primarily on the risk of **accumulation of inorganic metals and metalloids in soils**.

In a **first step**, soil screening values were collected for the different EU Member States as given in Section 16.1. **Soil screening values** are generic quality standards that are used to regulate land contamination and are adopted in many Member States in Europe in order to protect the environment and human health (Carlon, 2007).

In a **second step**, a maximal permissible concentration of the element in the CMC derived fertilising material is calculated based on the principle that predicted metal/metalloid accumulation as a result of the long-term application of the fertilising material and the atmospheric deposition in the soil shall not exceed the so-called soil screening value *('soil* *screening acceptable limit concentration'*) (see section 16.1 for methodological details). A mass balance approach is applied assuming that the **non-soluble fraction of metals and metalloids accumulates in soils**, and that the soluble metal fraction is removed from the soil through leaching and plant uptake. The calculated accumulation of the respective trace metal in soils is then dependent 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 soils. A simple spreadsheet-based model using a set of reasonable assumptions is applied for this purpose as outlined in detail in section 16.1. The **calculated soil screening acceptable limit contents** are then qualitatively compared to metal/metalloid concentrations that are typically found across the diverse range of pyrolysis & gasification materials derived from different eligible input materials. This assessment is applied **to spot possible issues** that might lead to human health and environmental protection due to the accumulation of metals and metalloids present in the CMC material. The outcome of this analysis indicated soil screening acceptable limit concentrations of selected metals and metalloids as given in Table 5.

Table 5: Outcome of the soil screening acceptable limit concentrations of selected metals and metalloids in CE fertilising products derived from pyrolysis & gasification materials



	maximal trace metal concentration (mg kg-1) (nutrient-rich/carbon-rich pyrolysis materials)	proposed limit (mg kg-1)
As	31/8	-
Ba	1335/334	-
Be	54/14	-
Co	101/25	-
Cr	243/61	-
Mo	52/13	-
Sb	12/3	-
Se	108/27	-
Tl	1.5/2	2*
V	377/94	-

In case more than 5% of pyrolysis/gasification additives have been applied

In a final step, **a validation of the soil screening acceptable limit concentration is performed by comparing them to the concentrations of the metals and metalloids observed in pyrolysis & gasification materials** (see Section 15.3.2**)**. It is indicated that by effectively restricting the eligible input materials for pyrolysis & gasification materials to bio-waste, category 2 and 3 animal by-products, living and dead organisms and a limited share of additives (see Section 5.5.2), **the concentrations of metals and metalloids that could possibly accumulate in pyrolysis & gasification materials is reduced (Section 16.3.2; Beesley et al., 2015; confidential information received from the STRUBIAS subgroup), and mostly well below the soil screening acceptable limit concentration given in Table 5**, with the exception of Sb. However, as outlined in Section 5.4.5.1 for thermal oxidation materials & derivates, the low soil screening value of 3 mg kg^{-1} used in this assessment might not be justified due to the low to moderate potential to cause harm to aquatic, soil and sediment organisms and the lack of Sb bioaccumulation in the soil. As a matter of fact, the value is about a factor 20 lower than the value of 54 mg kg^{-1} soil determined for a serious risk by van Vlaardingen et al. (2005). It is indicated that the possibility of the concentration of these metals and metalloids to levels of unacceptable risk is limited because (1) the pyrolysis/gasification process only **concentrates elements to a limited extent** relative to the raw feedstock applied (typically factor 2-5; Boateng et al., 2015), and (2) **feedstocks that could be enriched in metals and metalloids, such as sewage sludge, fossil fuels and ore and ore concentrates, are not listed as eligible input materials** (see Section 5.5.2). The voluntary standardisation scheme of the European Biochar Certificate has established limit values for Pb, Cd, Cr, Cu, Ni, Hg and Zn (Section 15.3.2). With the exception of Cr, all these metals are regulated at PFC level in the EU Fertilising Products Regulation ((EU) 2019/1009). The values for Cr for pyrolysis & gasification materials derived from the eligible input materials range are typically much lower than the soil screening acceptable limit concentration, for both carbon-rich and nutrient-rich pyrolysis & gasification materials (Section 15.3.2).

When pyrolysis & gasification materials have exclusively been derived from plant- and/or animal-based biomass, the derived limit values for Tl will not be exceeded. However, some pyrolysis/gasification materials, including specific minerals (e.g. feldspar, biotite, muscovite and many others), can contain high thallium contents (Jovic, 1993). Therefore, it is proposed that pyrolysis & gasification materials **shall contain no more than 2 mg kg^{-1} dry matter of thallium (Tl), in the event that more than 5% of pyrolysis/gasification additives relative to the total input material fresh weight have been applied**.

Leaching of metals, metalloids, non-metals and halogens

Similar to thermal oxidation materials & derivates, no environmental risks are expected **due to the leaching of inorganic metals and metalloids when their concentration in the pyrolysis & gasification materials does not exceed the proposed limits.** As a matter of fact, the percolation of these is **highly reduced** due to the physico-chemical properties of the pyrolysis & gasification materials, and their accretion in a stable, aromatic matrix.

5.5.5.2 PAHs, PCDD/F and dl-PCB

Three particular classes of contaminants that are not strictly feedstock-dependent can be **formed by the thermochemical processes** used to produce pyrolysis & gasification materials. These *de novo* formed compounds are polycyclic aromatic hydrocarbons (PAHs), and dioxins and furans (PCDD/F), and dioxin-like polychlorinated biphenyls (dl-PCB). The formation of other organic pollutants, such as low-molecular-weight mineral-oil-like substances and other volatile organic carbon compounds, is limited in the production process of pyrolysis & gasification materials that meet the proposed criteria on H:C ratio (~carbon stability) and PAH content (Smith et al., 2016).

Low-temperature formation (< 600 °C) of PAHs takes place as a result of condensation, carbonisation and aromatisation of the solid material as it transforms (Bucheli et al., 2015). During biomass pyrolysis, PAHs are also formed by pyrosynthesis, i.e. where different gaseous hydrocarbon radicals are generated by cracking of organic material under high-temperature conditions. These radicals then undergo a series of bimolecular reactions to form larger polyaromatic ring structures (Bucheli et al., 2015). A wide range of PAHs has been detected in pyrolysis & gasification materials (Bucheli et al., 2015; for a good overview and summary tables), which is why it **is proposed to limit PAH content** and to include this parameter as part of the Conformity Assessment Procedure for pyrolysis & gasification materials.

Little information on PCDD/F and dl-PCB contents in pyrolysis & gasification materials is available. In principle, the **formation of these contaminants requires both the presence of significant amounts of chloride in the feedstock (e.g. specific herbaceous biomass types) and a high pyrolysis temperature** (~ 750 °C) (Libra et al., 2011; Aller, 2016). Nonetheless, PCDD/F and dl-PCBs can be formed at temperatures starting at 300 °C (Lundin and Marklund, 2005), and their formation might thus occur at the typical operation temperatures applied in pyrolysis plants. Moreover, the **adsorption of PCDD/Fs and dl-PCBs** can be favoured by the presence of elementary carbon or soot particles in the pyrolysis & gasification materials (Vehlow et al., 2006). At present, there is only limited information available on the presence of PCDD/Fs and dl-PCBs in pyrolysis & gasification materials (Section 16.3.2; Bucheli et al., 2015) (confidential information from the STRUBIAS subgroup), so the precautionary principle should apply. For that reason, the European Biochar Certificate also not only includes PAHs, but also PCDD/F and dl-PCB as part of the compliance scheme.

As indicated by the STRUBIAS subgroup and scientific literature (Buss et al., 2016), **the current technology readiness level enables the production of pyrolysis & gasification materials with low levels of persistent organic pollutants**. Even without post-combustion treatment for the abatement of organic compounds, acceptable levels of POPs can be reached for many pyrolysis & gasification materials (Bucheli et al., 2015; Someus and Pugliese, 2018).

Therefore, it is proposed to adhere to the PAH limit proposed for some other CMCs and to the strictest levels for PCDD/Fs and PCBs set by existing national legislation and quality standards, specifically those of the European Biochar Certificate (EBC, 2012):

- PAHs (16 US EPA congeners, mg kg^{-1} dry matter): < 6;
- PCDD/F (ng WHO toxicity equivalents kg^{-1}dry matter): < 20;
- Dl-PCB (sum of 6 congeners PCB 28, 52, 101, 138, 153, 180, mg kg^{-1}): < 0.2.

The STRUBIAS subgroup indicated that, for pyrolysis & gasification materials, test methods that rely on extraction with toluene (e.g. DIN EN 15527, edition 2008-09 with toluene extraction; DIN ISO 13877:1995-06 – Principle B with GC-MS; see Section 5.7.3) may be suitable because PAHs can be strongly adsorbed to the organic matrix so PAH analysis methods adapted for soils may not accurately detect PAHs present in the pyrolysis &

gasification materials. It is noted, however, that the testing and possible development of new *Harmonised Standards* falls beyond the scope of the JRC mandate.

Note that these values are substantially lower than the limit values established in EU Regulation (EU) No 756/2010 amending Regulation (EC) No 850/2004 of the European Parliament and of the Council on persistent organic pollutants as regards their Annexes IV and V (limit values of 15 μg kg^{-1} for PCDD/F and 50 mg kg^{-1} for dl-PCBs).

5.5.5.3 Other organic chemical pollutants

At present, **relatively little research has been conducted on the subject of organic pollutants,** other than PAHs, dl-PCBs and PCDD/F, present in pyrolysis & gasification materials. As indicated in Section 5.5.3.2**, the weight loss from pyrolysis & gasification materials at temperatures of > 500 °C is due to the removal of organic compounds** (Deydier et al., 2005a; Koutcheiko et al., 2007; Ro et al., 2010; Marculescu and Stan, 2012). The high-temperature process **reduces to a variable extent the concentrations of organic micropollutants,** including those originating from veterinary medicines, hormones, and herbicides that may be present in the input materials (Ross et al., 2016). Therefore, the main important organic chemical pollutants, such as veterinary antibiotics, have been largely removed from pyrolysis & gasification materials that show an increased C stability and low $H:C_{org}$ ratio < 0.7 (STRUBIAS subgroup information received from testing by Hitz; Ross et al., 2016; Liang et al., 2017). Nonetheless, little is known about concentration, bioavailability, and possible decay products that can be formed and adsorbed during the production process. This has been one of the main reasons for proposing **a positive input material list** for pyrolysis & gasification materials that includes only materials for which the pyrolysis/gasification process might lead to acceptable risks for the environment and human health (see Sections 5.2.9 and 5.5.2). Considering that (1) the eligible input materials list only comprises bio-waste, animal by-products and living or dead organisms and (2) purging of veterinary drugs takes place during the pyrolysis production process, **it is proposed not to request any additional chemical analyses for organic contaminants other than those already mentioned for PAHs, PCDD/Fs and dl-PCBs.**

5.5.5.4 Biological pathogens

The pyrolysis process has been shown to thermally decompose biological pathogens and to effectively reduce microbial communities (Liu et al., 2014; Uchimiya, 2014). The thermal destruction has been studied in great detail by the food industry because of the importance of this process in killing pathogenic bacteria and preventing foodborne spoilage. It was observed that much lower temperatures than those observed in pyrolysis/gasification are required for the dry heat deactivation of biological pathogens. **Above temperatures of 120 °C, minimal thermal death times are required to inactivate biological pathogens**, even under dry conditions (Gerba, 2015). Microorganisms as well as viruses and enzymes are thus denatured at the temperatures applied during pyrolysis, with survival rates decreasing exponentially as a function of temperature and reaction time (Gerba, 2015).

Slow or mild pyrolysis takes place at 300-650 °C with relatively long residence times. Based on the proposed requirement of a maximal H/C_{org} ratio of 0.7 (see Section 5.5.4.1), the minimal residence times will be at least be a few minutes (Kambo and Dutta, 2015) (Table 6). Higher temperatures (500-900 °C) but lower residence times (10-20 seconds) are typical for **gasification processes** (Table 6). Also, **hydrothermal carbonisation**, a wet heating process in which the input materials are placed in a closed reactor (i.e. autoclave) and treated at about 180-250 °C in a confined system under pressure (2-6 MPa) for 5-240 minutes (Table 6), will result in the effect inactivation of all biological pathogens. The heat resistance of microbial cells even decreases with increasing humidity and moisture (Gerba, 2015).

Table 6: Classification of different pyrolysis/gasification processes in terms of operating conditions (adopted from Kambo and Dutta, 2015)

Pyrolysis process	Operating temperature	Residence time	Heating rate
Slow pyrolysis	300-650 °C	5 min – 12h	10-30 °C/min
Mild pyrolysis	200-300 °C	30 min – 4h	10-15 °C/min
Gasification	600-900 °C	10-20s	50-100 °C/min
Hydrothermal carbonisation	180-260°C	5 min-12h	5-10 °C/min

Specific criteria to control for biological pathogens have been proposed in the EU Fertilising Products Regulation ((EU) 2019/1009) at PFC level (for organic and organo-mineral fertilisers included in PFC 1, organic soil improvers (PFC 3), growing media (PFC 4), and non-microbial biostimulants (PFC 6). Therefore, **no specific criteria for biological pathogens are proposed**.

Many of the standard processing techniques for animal by-products (see Regulation (EU) 142/2011, e.g. pressure sterilisation, Brookes gasification) could potentially be included as part of the production process of pyrolysis & gasification materials. Therefore, **it is proposed that the end point in the manufacturing chain as defined in the Animal By-Products Regulation can be reached at a stage *prior to* or *during* the pyrolysis material production process**. As such, the compliance with the Animal By-Products Regulation will be met for all pyrolysis & gasification materials derived from animal by-products.

According to Article 32 of Regulation (EC) 1069/2009, animal by-products derived from category 2 or category 3 material are allowed for the production of organic fertilisers and soil improvers provided that they have **been produced in accordance with the conditions for pressure sterilisation or have been digested or composted**. Moreover, the Brookes' gasification process as described in point E, Section 2, Chapter IV, of Annex IV to Regulation (EU) No 142/2011, is considered a processing method for animal by-products of categories 2 and 3.

As outlined in Section 5.2.6, **the placing on the market of processed manure, products derived from processed manure and guano from bats is subject to the requirements laid down in Annex XI (Chapter I, Section 2) to Regulation (EU) 142/2011.** The standard processing method that such material must undergo includes a heat treatment process of at least 70 °C for at least 60 minutes and it shall have been subjected to reduction in spore-forming bacteria and toxin formation, where they are identified as a relevant hazard. Nonetheless, the competent authority may authorise the use of other standardised process parameters besides those referred to above, **provided that such parameters ensure the minimisation of biological risks**. This involves, amongst others, the identification and analysis of possible hazards, a validation of the intended process by measuring the reduction of viability/infectivity of endogenous indicator organisms, including, for instance, *Enterococcus faecalis*, thermoresistant viruses such as parvovirus, parasites such as eggs of *Ascaris* sp., *Escherichia coli*, *Enterococcaceae*, and *Salmonella* spp. It can be reasonably assumed that the combination of the proposed production techniques (Table 6) and the maximum $H:C_{org}$ ratio of 0.7 (~ high degree of carbonisation) will ensure process conditions that are more stringent than the default heat treatment process (at least 70 °C for at least 60 minutes) for the placement of **processed manure** on the market.

Therefore, the following proposals are made (see also Section 5.2.6):

Pyrolysis & gasification materials that have been derived from manure, non-mineralised guano, and digestive tract content pursuant to Regulation (EC) No 1069/2009 provided that they:

- are compliant with the proposed criteria for CMC ZZ.

Pyrolysis & gasification materials that have been derived from animal by-products and derived materials from category 2 or category 3 material as defined by Regulation (EC) No 1069/2009 provided that they:

- are compliant with the proposed criteria for CMC YY,
- have undergone one of the following treatments at a stage prior to or during the pyrolysis & gasification material production process:
  - pressure sterilisation or other conditions to prevent risks arising to public and animal health, in accordance with the requirements laid down pursuant to Article 15 of Regulation (EC) No 1069/2009,
  - transformation into biogas or compost as set out in Annex V to (EU) No 142/2011, or
  - Brookes' gasification process as described in point E, Section 2, Chapter IV, of Annex IV to Regulation (EU) No 142/2011.

5.5.5.5 Emissions

The mechanisms that lead to emissions from the handling and application of fertilising during material are outlined in Section 5.3.5.4.

One of the mechanisms may generate **airborne dusts and particulate matter emissions**. There are concerns that pyrolysis & gasification materials can be lost from the soil during and after the application through the **physical erosion and the abrasion of pyrolysis material particles**, thus offsetting any delayed decomposition on account of chemical recalcitrance (Ravi et al., 2016). Additionally, particulate matter emissions from soils amended with pyrolysis & gasification materials may impact on air quality. In this respect, the following aspects are relevant to consider:

a) The dustiness of a powder product, defined as the propensity of a material to generate airborne dust during its handling (Lidén, 2006), not only depends on the **intrinsic physical properties of the material but also on the handling scenario.**
b) Only significant losses relative to control soils have been observed upon the application of unsieved pyrolysis & gasification materials (produced at a mild temperature of 300 °C) at application rates of 10-20% of the soil (v/v) (Ravi et al., 2016). Assuming a ploughing depth of 20 cm and a bulk density of 1.4 g cm^{-3}, this would correspond to unrealistic application rates of 630-1 260 tonnes $ha-^{1}$. At lower application rates (e.g. 5% of the soil) and following sieving (> 2 mm), no significant losses were observed of the pyrolysis material. Moreover, the often applied rewetting practices to levels > 15% provide an effective solution to overcome particulate matter emissions during the land use phase of the product (Silva et al., 2015).

In line with the discussion provided in Section 5.3.5.4, it is concluded that **correct classification and labelling as foreseen in EU legislation allows downstream users to assess the risk associated airborne dust emissions and other emissions to air during the handling and application of certain products, and to take the necessary measures to prevent any potential adverse impacts in the event that a risk is identified. It is indicated that the provisions in the EU Fertilising Products Regulation ((EU) 2019/1009) on labelling and European Regulations are sufficiently effective to control for any adverse impacts associated with emissions during the handling and application of thermal oxidation materials & derivates.**

5.5.5.6 Flammability

The handling, storage and application of pyrolysis & gasification materials can represent a **fire hazard** (Dzonzi-Unidm et al., 2012). Dust particles from pyrolysis & gasification materials can form explosive mixtures with air in confined spaces, and there is a danger of spontaneous heating and ignition when biochar is tightly packed. This occurs because fresh pyrolysis material quickly absorbs oxygen and moisture, and these sorption processes are exothermic, thus potentially leading to a high temperature and ignition of the material. The volatile compounds present in pyrolysis & gasification materials may also represent a fire hazard, which is reduced if the proposed criteria on carbon stability are met.

Water can also reduce flammability, but its effectiveness is dependent on the degree of water saturation of the pyrolysis material. Addition of water to pyrolysis & gasification materials, however, increases the weight of the material and thus shipping costs. The best way to prevent fire is to store and transport **biochar in an atmosphere which excludes oxygen** (Blackwell et al., 2009). Pelletising and admixing of pyrolysis & gasification materials with composts, or the production of biochar-mineral complexes will also yield materials which are much less flammable.

Correct classification and labelling of the material properties allows downstream users to assess the flammability risk during the handling and application of certain products, and to take the necessary measures to prevent any potential adverse impacts in the event that a risk is identified. Hence, it is indicated that the provisions in the EU Fertilising Products Regulation ((EU) 2019/1009) on labelling and European Regulations ((EC) No. 1272/2008, Regulation (EC) No 1907/2006) are sufficiently effective to control for flammability risks during the handling and application of pyrolysis & gasification materials. **Therefore, no further requirements are proposed to control for material flammability.**

5.5.5.7 Occupational health

Council Directive 89/391/EEC on the introduction of measures to encourage improvements in the safety and health of workers at work seeks to adequately protect workers and encourages improvements in occupational health and safety in all sectors of activity, both public and private. The Directive also promotes workers' rights to make proposals relating to health and safety, to appeal to the competent authority and to stop work in the event of serious danger. No further legal requirements are therefore proposed.

5.5.6 Physico-chemical properties

5.5.6.1 Physical impurities

Given that the eligible input materials only include bio-waste, living or dead organisms and animal by-products (but not municipal solid waste, sewage sludge), impurities in the form of glass, plastics and metals could be present in low to moderate amounts in the input materials. Materials that are more likely to contain impurities, such as municipal solid waste and sewage sludge, have been excluded as eligible input materials for pyrolysis & gasification materials. The general framework of the EU Fertilising Products Regulation ((EU) 2019/1009) explicitly mentions that 'impurities in EU fertilising products derived from bio-waste, in particular polymers but also metal and glass, should be either prevented or limited to the extent technically feasible by detection of such impurities in separately collected bio-waste before processing'. Moreover, the pyrolysis/gasification process may effectively decompose certain impurities, like plastics. **Therefore, no further criterion is proposed to limit visually detectable physical impurities > 2 mm.**

5.5.6.2 Dry matter content

It is proposed **not to set a criterion on moisture content**, but to enable the material producer to adjust the dry matter content along with other material properties to manage issues related to flammability, material handling, storage, transport and application.

5.5.6.3 pH

Reactive pyrolysis & gasification materials with a very high or low pH are not suitable for land application as they will induce a pH shock effect both on soil and flora. Nonetheless, it is noted that pyrolysis & gasification materials with a high pH may fulfil a function as liming material, and that other liming materials (e.g. aglime) can also be applied as EU fertilising materials. Moreover, CMC materials can be physically mixed with other CMC materials with the resulting changes in the pH of the PFC material. **Therefore, no requirements on the pH value have been proposed for pyrolysis & gasification materials**.

5.5.6.4 Granulometry

It is noted that the particle form, i.e. granule, pellet, powder, or prill, of the product shall be indicated on the **label** of solid inorganic macronutrient fertilisers (see labelling requirements in the EU Fertilising Products Regulation ((EU) 2019/1009)). The particle size distribution of the pyrolysis & gasification materials is related to the loss, transport and interaction of pyrolysis & gasification materials in the environment. It has an influence on health and safety protocols relating to handling, storage, transport, and human exposure in regard to pyrolysis material dust particles (IUPAC, 1990; Ravi et al., 2016). **Therefore, it is important that the end users of pyrolysis & gasification materials are informed of the material properties. No further requirements for granulometry are proposed.**

5.5.7 Handling and storage

As indicated above, the pyrolysis/gasification process causes an effective reduction or complete elimination of microbial communities. Therefore, (re-)contamination of the material with biological pathogens is unlikely if good management practices during storage are applied. It is proposed that **physical contact between input and output materials after the pyrolysis/gasification process must be avoided, including during storage**. Similar provisions have been formulated for compost (CMC 3) and digestates (CMC 4 and 5).

5.5.8 Bioassays

Pyrolysis & gasification materials have been showing promise for increasing crop productivity (Jeffery et al., 2015). Nonetheless, in contrast to many traditional fertilising products, pyrolysis & gasification materials vary widely in their product properties, hence their behaviour in the soil is often difficult to predict. Indeed, despite intensive research on the interactions between pyrolysis & gasification materials and soils, there is **still not sufficient mechanistic understanding of such interactions to produce a reliable decision support tool that would be universal across the different soil-pyrolysis material combinations** (Camps-Arbestain et al., 2015; Jeffery et al., 2015), and that would define **the short-and long-term risks for the environment of some endogenous contaminants present in pyrolysis material** (Clements et al., 2015; Dutta et al., 2016; Kuppusamy et al., 2016; Hilber et al., 2017).

In the STRUBIAS Interim Report, a bioassay based on earthworm avoidance was proposed to check for unknown pollutants or contaminants for which no limit value could be established at that time. For the final report, we have, however, removed that proposal from the compliance scheme for the following reasons:

- **Supplementary assessments and additional techno-scientific information have been collected and included in the assessment** (e.g. on VOCs, Mn, B; see respective sections in the report). This information confirmed that pyrolysis & gasification materials derived from eligible input materials that are compliant with the proposed limit values for contaminants do not pose unacceptable risks for the environment and human health.
- The STRUBIAS subgroup indicated that bioassays could **provide little added value in terms of an increased level of environmental and human health protection for fertilising materials**. The reason is that both the nutrients and the contaminants contained in the fertilising material could affect the behaviour of test species such as soil fauna (Abbiramy et al., 2014) or algae (Parker et al., 1997). Therefore, even inorganic fertilisers with low contaminant levels may fail to pass bioassay tests (Abbiramy et al., 2014; Abbiramy and Ross, 2016). The compliance costs for bioassays, varying from EUR 2 500 (e.g. algal growth inhibition test) to EUR 4 500-6 500 (earthworm avoidance test) per sample, are thus not proportionate to the possibly protective benefit due to the inability to provide a rapid, sensitive, reproducible and reliable screening of environmental and human health risks. This observation especially holds true for biochars that contain a significant share of plant-available nutrients (e.g. pyrolysis materials derived from animal by-products).
- At relevant application rates, pyrolysis & gasification materials with low nutrient amounts (e.g. pyrolysis materials derived from vegetable matter) **as produced by industrial facilities and compliant with the proposed compliance scheme for this CMC generally pass bioassay tests** (Busch et al., 2012; Domene et al., 2015). Pyrolysis & gasification materials not passing bioassay tests are often not compliant with the proposed criteria (e.g. Busch et al., 2012) and/or result from production processes under (deficient) conditions leading to materials that (likely) do not meet other contaminant limit values such as PAHs, PCDD/F or dl-PCBs (Smith et al., 2016; Visioli et al., 2016). These observations indicate that the proposed technical requirements for pyrolysis & gasification materials will result in the selection of materials that show a high potential to serve as added-value products in EU agriculture.

For all these reasons, **it is proposed to omit the inclusion of bioassays for pyrolysis & gasification materials.** This proposal is in line with voluntary standardisation schemes (e.g. European Biochar Certificate) and existing national legislation in the EU and the EFTA state Switzerland (Meyer et al., 2017).

5.6 Tolerance rules

Part 3 of Annex III (on labelling requirements) to the EU Fertilising Products Regulation ((EU) 2019/1009) outlines the tolerance rules for different PFCs. This section indicates, for instance, that:

- the declared P_2O_5 content of a mineral fertiliser (PFC 1(C)) shall have a permissible tolerance of ± 50% of the declared content up to a maximum of 1 percentage point in absolute terms;
- the permissible tolerance for the declared neutralising value of a liming material (PFC 2) is ± 3; or
- the declared organic carbon (C) content of a soil improver (PFC 3) shall have a permissible tolerance of ± 10% relative deviation of the declared value up to a maximum of 1.0 percentage points in absolute terms.

It is noted that the STRUBIAS materials are often derived from an eligible input material composition that shows a larger degree of possible material property variations relative to other CMCs, for instance due to temporal variations or different sites of collection. Especially for some of the materials covered under the CMCs ‘thermal oxidation materials & derivates’ and ‘pyrolysis & gasification materials’ variations in the feedstock properties could impact upon the variation of the output material quality parameters such as nutrient content and physico-chemical properties. Nonetheless, similar to other CMCs, **transparency and consistency in the CMC material quality is central for downstream users, including** retailers, fertilising blending companies, and farmers, and may **stimulate increased customer confidence** for the innovative STRUBIAS materials. Moreover, possible variations in output material quality can be addressed through batch mixing processes.

The STRUBIAS subgroup highlighted, however, the need to **test, validate, and possibly develop new *Harmonised Standards*** for each of the testing requirements for the STRUBIAS material groups.

Therefore, it is **proposed that the tolerance rules as outlined in Part 3 of Annex III on labelling requirements shall, in principle, also apply to STRUBIAS materials.** It is proposed to re-evaluate the assessment of the tolerance rules in the event that such work indicates:

(i) a significantly higher degree in sampling variability, or

(ii) a significantly lower degree of analytical precision when analysing the nutrient content or physico-chemical characteristics,

for STRUBIAS materials relative to other CMCs included in the EU Fertilising Products Regulation ((EU) 2019/1009).

5.7 Quality management

5.7.1 Selection of conformity assessment procedures

In the EU Fertilising Products Regulation ((EU) 2019/1009), the essential requirements relating to products laid down in the Union legislation as *Harmonised Standards* have not been adopted for all product requirements, or do not cover with sufficient detail all elements of the quality system to safeguard the safety requirements for specific manufacturing processes and fertilising products. Therefore, it is necessary to provide a presumption of conformity for EU fertilising products which are in conformity with *Harmonised Standards* that are adopted in the comprehensive regulatory framework of Regulation (EC) No 765/2008 and Decision No 768/2008/EC[30]. **The essential objective of a conformity assessment procedure is to demonstrate that products placed on the market conform to the requirements expressed in the provisions of the relevant legislation**. They include procedures for sampling, testing and inspection; evaluation, verification and assurance of conformity; and registration, accreditation and approval.

On the basis of ISO/IEC documentation, Regulation (EC) No 768/2008 outlines consolidated conformity assessment procedures and the rules for their selection and use in legislation. As a general rule, products are subject to both **design and production modules** before being placed on the market. There are **eight modules** (named with the letters from A to H). They lay down the responsibilities of the manufacturer (and their authorised representative) and the degree of involvement of the accredited in-house body or notified conformity assessment body. They are the components of the conformity assessment procedures laid down under Decision No 768/2008/EC, the ‘horizontal menu’. Several modules have their variants (e.g. A1, D1). The reason for providing variants within modules is **to enable the necessary level of assured protection for products presenting a higher level of risk while avoiding the imposition of a heavier module**. The idea is to minimise the burden on manufacturers to the extent that is possible.

The legislator selects from the menu of conformity assessment modules/procedures (laid down under Decision No 768/2008/EC) the most appropriate one(s) in order to address the specific needs of the sector concerned. **The complexity of the modules selected should be proportional to the risk and the level of safety required** (impact on public interest, health, safety, environment) of the product, its design complexity, the nature of its production (large series vs small series, custom-made, simple vs complex production mechanism, etc.). It is necessary to offer a choice of clear, transparent and coherent conformity assessment procedures, restricting the possible variants.

In the EU Fertilising Products Regulation ((EU) 2019/1009), modules A, A1, B + C, and D1 are listed, but the selection of the modules is dependent on the PFC and CMC under which the fertiliser material will be placed on the market.

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[30] 2016/C 272/01 Commission Notice — The ‘Blue Guide’ on the implementation of EU products rules 2016.

Table 7: Consolidated conformity assessment modules of Regulation (EC) No 768/2008 used in the EU Fertilising Products Regulation ((EU) 2019/1009)

Module	Description
A - Internal production control	Covers both design and production. The manufacturer himself ensures the conformity of the products to the legislative requirements (no EU-type examination).
A1 - Internal production control plus supervised product testing	Covers both design and production. A + tests on specific aspects of the product carried out by an accredited in-house body or under the responsibility of a notified body chosen by the manufacturer.
B - EU-type examination	Covers design. It is always followed by other modules by which the conformity of the products to the approved EU-type is demonstrated. A notified body examines the technical design and or the specimen of a type and verifies and attests that it meets the requirements of the legislative instrument that apply to it by issuing an EU-type examination certificate. There are three ways to carry out EU-type examination: 1) production type, 2) combination of production type and design type and 3) design type.
C - Conformity to EU-type based on internal production control	Covers production and follows module B. The manufacturer must internally control his production in order to ensure product conformity against the EU-type approved under module B.
D1 - Quality assurance of the production process	Covers both design and production. The manufacturer operates a production quality assurance system (manufacturing part and inspection of final product) in order to ensure conformity to legislative requirements (no EU-type). The notified body assesses the production quality system.

Products which are regarded **as presenting a high risk to the public interest require conformity assessment by a third party, i.e. a notified body**. Notified bodies are conformity assessment bodies which have been officially designated by their national authority to carry out the procedures for conformity assessment within the meaning of applicable Union harmonisation legislation when a third party is required. If the demonstration of conformity of products cannot be left to the manufacturer but requires that products are supervised by a notified body during the production process, then the legislator may require the manufacturer to operate an **approved quality system** (for example module D). In the case of products of simple design but complicated production/manufacturing, the legislator may consider selecting module variants and using thus the advantages of the main module, without the necessity of resorting to a more formal specimen examination (for example, as provided for under module B which precedes module D where the notified body assesses the production quality system).

It is proposed that module D1 shall be used for any EU fertilising product derived from STRUBIAS materials. This proposal to request the manufacturer to operate a production quality assurance system is mainly based on the risks associated with the handling of the eligible input materials such as animal by-products and wastes pursuant to Regulation 2008/98/EC that are present on the eligible input material list for all STRUBIAS CMCs. Implementing and applying a module based on the quality of the design and production phase is therefore appropriate to ensure full compliance with the complex and extensive legislative requirements that apply to the handling, transport, and operating conditions of the STRUBIAS production processes. Also, the provisions laid down in the parts of EU legislation that regulate industrial emissions and large combustion plants (Directive 2010/75/EU, including large combustion plants > 50 MWth), Directive (EU) 2015/2193 (> 1 MWth, but < 50 MWth), and Regulation (EU) 2015/1189 (smaller appliances (heaters and boilers < 1 MWth) require that treatment plants and production processes are supervised by a notified body. In the EU Fertilising Products Regulation ((EU) 2019/1009), **module D1 is also the only module available for CMCs that have animal by-products listed as eligible input materials** (i.e. CMC 3 (compost) and CMC 5 (digestate other than energy crop digestate)). Moreover, **module D1 can be applied for any EU fertilising product**, with the exception of a straight or compound solid inorganic macronutrient ammonium nitrate fertiliser with a high nitrogen content, or a fertilising product blend containing such a product.

For the purpose of complying with the applicable legislation, the manufacturer must ensure that the **quality system is implemented and applied in such a way that it ensures full compliance of the products with the legislative requirements in question**. The D1 module based on quality assurance techniques describes the elements a manufacturer must implement in their organisation in order to demonstrate that the product fulfils the essential requirements of the applicable legislation. This means that a manufacturer is given the possibility of using an approved quality system for the purpose of demonstrating compliance with regulatory requirements. The quality system is assessed by the notified body. A quality system implemented on the basis of the EN ISO 9001 *Standard* gives a presumption of conformity with the respective modules with regard to the provisions in the modules that these *Standards* cover, provided that the quality system takes into consideration the specificities of the products concerned. However, the manufacturer is free to apply other quality system models than those based on EN ISO 9001 for the purpose of complying with these modules. In any case, the manufacturer must specifically address all regulatory provisions while applying their quality system, in particular:

- the quality objectives, quality planning and quality manual must fully take on board the objective of delivering products that conform to the essential requirements;
- the manufacturer must identify and document the essential requirements that are relevant for the product and the *Harmonised Standards* or other technical solutions that will ensure fulfilment of these requirements;

- the identified *Harmonised Standards* or other technical solutions must be used as design input, and as verification that design output ensures that the essential requirements will be met;
- the measures taken to control manufacturing must ensure that the products conform to the identified essential requirements;
- quality records, such as inspection reports and test data, calibration data, qualification reports of the personnel concerned, must be suitable to ensure the fulfilment of the applicable essential requirements.

5.7.2 Additional elements of the quality management system and auditing

The elements of module D1 for application in the EU Fertilising Products Regulation ((EU) 2019/1009) have been laid down in the final legislative text. The provisions are largely based on the ‘default’ module D1 as laid down in Decision (EC) 768/2008, but include sections of text that have been **modified**. The adapted module D1 in the EU Fertilising Products Regulation ((EU) 2019/1009) **includes additional provisions that account for the specificity of techniques applied in manufacturing processes of fertilising products and the treatment of animal by-products** pursuant to Regulation (EC) No 1069/2009.

It is **proposed to further develop the conformity assessment procedure D1, and to incorporate the following:**

(1) **Provisions that relate to the testing of the technical requirements for production process conditions laid down in the STRUBIAS recovery rules.** The STRUBIAS recovery rules have, nonetheless, been developed with a focus on the testing of the CMC material that will be incorporated in the EU fertilising product. Therefore, the technical requirements for production process conditions are minimal. As a matter of fact, the single element that requires further description in the conformity assessment procedure is the mass balance approach to demonstrate the removal of contaminants to levels below the limit values established in this Regulation during post-combustion manufacturing processes of hazardous ashes.

(2) **Additional elements in the quality assurance procedure that are specific to the handling and treatment of specific input materials.** The production processes for the STRUBIAS CMCs may involve the transport and handling of waste as input materials, and possibly multi-operator installations/sites. Since the manufacturer that brings the EU fertilising material onto the market bears the quality control responsibility, they should be given the possibility to demonstrate full compliance of the technical specifications. It is noted that notified bodies have the possibility to verify compliance at different production sites, even if those are located in different EU Member States.

(3) **A different testing frequency regime based on material quantities of EU fertilising products that are placed on the internal market.** The proposed requirements that are currently laid down for module D1 had been based on input material for composting and anaerobic digestion plants, where input and output material tonnages are in the some order of magnitude. For STRUBIAS materials, the input material amounts processed may be one or more orders of magnitude different from the produced output material amounts, so it is proposed to base the testing frequency on the tonnage of the EU fertilising material produced and to increase the frequency correspondingly. Additional testing elements to have been proposed include possible modifications to the default frequency of testing by considering average values and the variability of historical samples, and the need to store retainer samples for possible corrective management actions in case of a non-compliance event.

(4) Minor aspects related to certification, training and auditing in the adapted D1 module will be reviewed and updated with new proposals, if deemed appropriate.

To present the adapted D1 module included in the EU Fertilising Products Regulation ((EU) 2019/1009) and the proposals for additions resulting from the addition of the STRUBIAS CMC, colour codes will be used. The adapted D1 module will be presented in green, whereas the proposals for further additions will be highlighted in purple, as follows:

MODULE D1: QUALITY ASSURANCE OF THE PRODUCTION PROCESS

1. Description of the module

1. Quality assurance of the production process is the conformity assessment procedure whereby the manufacturer of the EU fertilising product fulfils the obligations laid down under Headings 2, 4, and 7, and ensures and declares on his or her sole responsibility that the EU fertilising products concerned satisfy the requirements of this Regulation that apply to them.

2. Technical documentation

2. The manufacturer of the EU fertilising product shall establish the technical documentation. The documentation shall make it possible to assess the product's conformity with the relevant requirements, and shall include an adequate analysis and assessment of the risk(s). The technical documentation shall specify the applicable requirements and cover, as far as relevant for the assessment, the design, manufacture and use of the product. The technical documentation shall, wherever applicable, contain at least the following elements:

(a) a general description of the product,

(b) conceptual design and manufacturing drawings and schemes, including a written description and a diagram of the production process, where each treatment, storage vessel and area is clearly identified,

(c) descriptions and explanations necessary for the understanding of those drawings and schemes and of the use of the EU fertilising product,

(d) a list of the *Harmonised Standards* applied in full or in part the references of which have been published in the *Official Journal of the European Union* and, where those *Harmonised Standards* have not been applied, descriptions of the solutions adopted to meet the essential requirements of this Regulation, including a list of common specifications or other relevant technical specifications applied. In the event of partly applied *Harmonised Standards*, the technical documentation shall specify the parts which have been applied,

(e) results of design calculations made, examinations carried out, *etc.*,

(e -bis) hazardous waste calculations

The testing referred to in paragraph 5 under CMC ‘Thermal oxidation materials & derivates’ in Annex II to demonstrate the removal or transformation of the contaminants to levels below the limit values as defined in Annex III to Directive 2008/98/EC for slags and ashes that display one or more hazardous properties, shall be carried out at least every year, or sooner than scheduled if triggered by any significant change that may affect the quality of the EU fertilising product (e.g. processing of input material batches of different composition, modification of process conditions). For a representative input material batch that is processed at the plant, the hazardous property identified (cfr. heading 5.1.3.1 (b – bis)) and the total mass shall be measured on the different input materials (1, …, n) and on the output material that will be incorporated in the CE market fertilising product. The incorporation rate of the hazardous property into the output material shall then be calculated as:

$$incorporation\ rate\ (\%) = \frac{HPC_{output\ material}\ \times\ M_{output\ material}}{\sum_{i=1}^{n}(HPC_{input\ material,\ i}\ \times\ M_{input\ material,\ i})}$$

where: HPC is the concentration of the hazardous property (mg kg^{-1}), M the total mass (kg), and i (1-n) the different input materials used in the production process.

The removal of the hazardous property during the production process shall be such that the incorporation rate multiplied by the concentration of the hazardous property of each individual input material below the limit values as defined in Annex III to Directive 2008/98/EC for that hazardous property.

(f) test reports, and

(g) where the product contains or consists of animal by-products within the meaning of Regulation (EC) No 1069/2009, the commercial documents or health certificates required pursuant to that Regulation, and evidence that the animal by-products have reached the end point in the manufacturing chain within the meaning of that Regulation.

3. Availability of technical documentation

3. The manufacturer shall keep the technical documentation at the disposal of the relevant national authorities for 10 years after the EU fertilising product has been placed on the market.

4. Manufacturing

4. The manufacturer shall operate an approved quality system for production, final product inspection and testing of the products concerned as specified in point 5, and shall be subject to surveillance as specified in point 6.

5. Quality system

5.1. The manufacturer shall implement a quality system which shall ensure compliance of the EU fertilising product with the requirements of this Regulation that apply to them.

5.1.1. The quality system shall include quality objectives and an organisational structure with responsibilities and powers of the management with regard to product quality.

5.1.1.1. For compost belonging to component material category ('CMC') 3 ~~and~~ , digestate belonging to CMC 5, precipitated phosphate salts & derivates belonging to CMC XX, thermal oxidation materials & derivates belonging to CMC YY and pyrolysis & gasification materials belonging to CMC ZZ, as defined in Annex II, senior management of the manufacturer's organisation shall:

(a) Ensure that sufficient resources (people, infrastructure, equipment) are available to create and implement the quality system;

(b) Appoint a member of the organisation’s management who shall be responsible for:

- Ensuring that quality management processes are established, approved, implemented and maintained;
- Reporting to senior management of the manufacturer on the performance of the quality management and any need for improvement;
- Ensuring the promotion of awareness of customer needs and legal requirements throughout the manufacturer's organisation, and for making the personnel aware of the relevance and importance of the quality management requirements to meet the legal requirements of this Regulation;
- Ensuring that each person whose duties affect the product quality is sufficiently trained and instructed; and
- Ensuring the classification of the quality management documents mentioned under paragraph 5.1.4. below;

(c) Conduct an internal audit every year, or sooner than scheduled if triggered by any significant change that may affect the quality of the EU fertilising product; and

(d) Ensure that appropriate communication processes are established within and outside the organisation and that communication take place regarding the effectiveness of the quality management.

5.1.2. The quality system shall be implemented through manufacturing, quality control and quality assurance techniques, processes and systematic actions.

5.1.2.1. For compost belonging to component material category ('CMC') 3 ~~and~~ , digestate belonging to CMC 5, precipitated phosphate salts & derivates belonging to CMC XX, thermal oxidation materials & derivates belonging to CMC YY and pyrolysis & gasification materials belonging to CMC ZZ, as defined in Annex II, the system shall ensure compliance with the ~~composting and digestion~~ process criteria specified in that Annex.

5.1.2.2. Where any of the requirements from Annex II for production processes for CMC XX (precipitated phosphate salts & derivates), CMC YY (thermal oxidation materials & derivates) or CMC ZZ (pyrolysis & gasification materials) have been fulfilled by prior holders of the input material, the manufacturer shall ensure that all such prior holders implement a quality system that complies with the requirements of this module.

5.1.3. The quality system shall comprise examinations and tests to be carried out before, during and after manufacture with a specified frequency.

5.1.3.1. For compost belonging to component material category ('CMC') 3 ~~and~~ , digestate belonging to CMC 5, precipitated phosphate salts & derivates belonging to CMC XX, thermal oxidation materials & derivates belonging to CMC YY and pyrolysis & gasification materials belonging to CMC ZZ, as defined in Annex II, the examinations and tests shall comprise the following elements:

(a) The following information shall be recorded for each lot of input materials:

(1) Date delivered;
(2) Amount by weight (or estimation based on the volume and density);
(3) Identity of the input material supplier;
(4) Input material type;
(5) Identification of each lot and delivery location on site. A unique identification code shall be assigned throughout the production process for quality management purposes; and
(6) In case of refusal, the reasons for the rejection of the lot and where it was sent.

(b) Qualified staff shall carry out a visual inspection of each consignment of input materials and verify compatibility with the specifications of input materials in CMC 3, CMC 5, CMC XX, CMC YY and CMC ZZ in Annex II. Where any of the requirements from Annex II for production processes for CMC XX (precipitated phosphate salts & derivates), CMC YY (thermal oxidation materials & derivates) or CMC ZZ (pyrolysis & gasification materials) have been fulfilled by prior holders of the input material, the manufacturer shall collect the necessary documentation from these prior holders, in order to be able to demonstrate full compliance with all requirements in Annex II.

(c) The manufacturer shall refuse any consignment of any given input material where ~~visual~~ inspection raises any suspicion of

- the presence of hazardous or damageable substances for the composting, digestion, precipitation, thermal oxidation or pyrolysis/gasification process or for the quality of the final EU fertilising product, or of
- incompatibility with the specifications of CMC 3~~ and~~, CMC 5, CMC XX, CMC YY and CMC ZZ in Annex II, in particular by presence of plastics leading to exceedance of the limit value for macroscopic impurities.

(d) The staff shall be trained on

- potential hazardous properties that may be associated with input materials, and
- features that allow hazardous properties and the presence of plastics to be recognised.

(e) Samples shall be taken on output materials, to verify that they comply with the component material specifications ~~for compost and digestate laid down in CMC 3 and CMC 5 in Annex II~~ for compost belonging to component material category ('CMC') 3 ~~and~~ , digestate belonging to CMC 5, precipitated phosphate salts & derivates belonging to CMC XX, thermal oxidation materials & derivates belonging to CMC YY and pyrolysis & gasification materials belonging to CMC ZZ, as defined in Annex II, and that the properties of the output material do not jeopardise the EU fertilising product's compliance with the relevant requirements in Annex I.

(f) The output material samples shall be taken with at least the following frequency, or sooner than scheduled if triggered by any significant change that may affect the quality of the EU fertilising product (e.g. processing of input materials listed under dissimilar letter items on the eligible input material list, modification of process conditions):

Annual ~~input~~ output (tonnes)	**Samples / year**
≤ 3000	4
3001 – 10000	8
10001 – 20000	12
20001 – 40000	16

40001 – 60000	20
60001 – 80000	24
80001 – 100000	28
100001 – 120000	32
120001 – 140000	36
140001 – 160000	40
160001 – 180000	44
> 180000	48

Manufacturers may reduce the default frequency of testing as indicated above by considering the statistical distribution of historical samples. Operators that meet the limit values for a specific parameter specified in the relevant sections of Annexes I and II to this Regulation after a minimum monitoring period of 1 year and a minimum number of 10 samples may reduce the default sampling frequency for that parameter by a factor 2 in case the upper boundary of the 95% confidence interval of the last 10 samples is smaller than the limit value for that parameter divided by a factor 2.

[Note that this proposal is based on a review of available quality management systems as proposed by the STRUBIAS subgroup, but that this proposal to base the testing frequency on the variability of historical samples has not been discussed with the STRUBIAS subgroup.]

(g) Each batch or portion of production shall be assigned a unique code for quality management purposes. At least one sample per 3 000 tonnes of EU fertilising material produced or one sample per 2 months, whichever is the soonest, shall be stored in good condition for a period of at least 2 years.

[Note that this proposal is based on a suggestion raised by a STRUBIAS subgroup expert during the final STRUBIAS meeting and following the review of available quality management systems proposed by the STRUBIAS subgroup, but that the number of samples to be stored has not been discussed with the STRUBIAS subgroup.]

(h) If any tested output material sample fails one or more of the applicable limits specified in the relevant sections of Annexes I and II to this Regulation, the person responsible for quality management referred to above in point 5.1.1.1(b) shall:

(1) Clearly identify the non-conforming products and their storage place,

(2) Analyse the reasons of the non-conformity and take any necessary action to avoid its repetition,

(3) Record in the quality records referred to in paragraph 5.1.4 if reprocessing takes place, or if the product is eliminated.

(h) Measure retainer samples and take the necessary corrective actions to prevent possible further transport and use of the material.

5.1.4. The manufacturer shall maintain the quality records, such as inspection reports and test data, calibration data, qualification reports on the personnel concerned, etc.,

5.1.4.1. For compost belonging to component material category ('CMC') 3 ~~and~~ , digestate belonging to CMC 5, precipitated phosphate salts & derivates belonging to CMC XX, thermal oxidation materials & derivates belonging to CMC YY and pyrolysis & gasification materials belonging to CMC ZZ, as defined in Annex II, the quality records shall demonstrate effective control of input materials, production, storage and compliance of input- and output materials with the relevant requirements of this Regulation. Each document shall be legible and available at its relevant place(s) of use, and any obsolete version shall be promptly removed from all places where it is used, or at least identified as obsolete. The quality management documentation shall at least contain the following information:

(a) A title,

(b) A version number,

(c) A date of issue,

(d) The name of the person who issued it,

(e) Records about the effective control of input materials,

(f) Records about the effective control of the production process,

(g) Records about the effective control of the output materials,

(h) Records of non-conformities,

(i) Reports on all accidents and incidents that occur to the site, their known or suspected causes and actions taken,

(j) Records of the complaints expressed by third parties and how they have been addressed,

(k) A record of the date, type and topic of training followed by the persons responsible for the quality of the product,

(l) Results of internal audit and actions taken, and

(m) Results of external audit review and actions taken.

5.1.5 The achievement of the required product quality and the effective operation of the quality system shall be monitored.

5.1.5.1. For compost belonging to component material category ('CMC') 3 ~~and~~ , digestate belonging to CMC 5, precipitated phosphate salts & derivates belonging to CMC XX, thermal oxidation materials & derivates belonging to CMC YY and pyrolysis & gasification materials belonging to CMC ZZ, as defined in Annex II, the manufacturer shall establish an annual internal audit program in order to verify the compliance to the quality system, with the following components:

(1) A procedure that defines the responsibilities and requirements for planning and conducting internal audits, establishing records and reporting results shall be established and documented. A report identifying the non-conformities to the quality scheme shall be prepared and all corrective actions shall be reported. The records of the internal audit shall be annexed to the quality management documentation.

(2) Priority shall be given to non-conformities identified by external audits.

(3) Each auditor shall not audit his or her own work.

(4) The management responsible for the area audited shall ensure that the necessary corrective actions are taken without undue delay.

(5) Internal audit realised in the frame of another quality management system can be taken into account provided that it is completed by an audit of the requirements to this quality system.

5.2. The manufacturer shall lodge an application for assessment of his or her quality system with the accredited notified body of his or her choice, for the products concerned. The application shall include:

- the name and address of the manufacturer and, if the application is lodged by the authorised representative, his or her name and address as well,
- a written declaration that the same application has not been lodged with any other notified body,
- all relevant information for the product category envisaged,
- the documentation concerning the quality system,
- technical documentation of all the quality system elements set out in paragraphs 5.1 and subparagraphs.

5.3. All the elements, requirements and provisions adopted by the manufacturer shall be documented in a systematic and orderly manner in the form of written policies, procedures and instructions. The quality system documentation shall permit a consistent interpretation of the quality programmes, plans, manuals and records. It shall, in particular, contain an adequate description of all the quality management elements mentioned above in paragraph 5.1 and subparagraphs.

5.4.1. The notified body shall assess the quality system to determine whether it satisfies the requirements referred to in paragraph 5.1 and subparagraphs.

5.4.2. It shall presume conformity with those requirements in respect of the elements of the quality system that comply with the corresponding specifications of the relevant *Harmonised Standard.*

5.4.3. In addition to experience in quality management systems, the auditing team shall have at least one member with experience of evaluation in the relevant product field and product technology concerned, and knowledge of the applicable requirements of this Regulation. The audit shall include an assessment visit to the manufacturer's premises. The auditing team shall review the technical documentation referred to in point 2 in order to verify the manufacturer's ability to identify the relevant requirements of this Regulation and to carry out the necessary examinations with a view to ensuring compliance of the EU fertilising product with those requirements.

5.4.4. The decision shall be notified to the manufacturer. The notification shall contain the conclusions of the audit and the reasoned assessment decision.

5.5. The manufacturer shall undertake to fulfil the obligations arising out of the quality system as approved and to maintain it so that it remains adequate and efficient

5.6.1. The manufacturer shall keep the notified body that has approved the quality system informed of any intended change to the quality system.
5.6.2. The notified body shall evaluate any proposed changes and decide whether the modified quality system will continue to satisfy the requirements referred to in point 5.2 or whether reassessment is necessary.
5.6.3. It shall notify the manufacturer of its decision. The notification shall contain the conclusions of the examination and the reasoned assessment decision.

6. Surveillance under the responsibility of the notified body

6.1 The purpose of surveillance is to make sure that the manufacturer duly fulfils the obligations arising out of the approved quality system.
6.2. The manufacturer shall, for assessment purposes, allow the notified body access to the manufacture, inspection, testing and storage sites and shall provide it with all necessary information, in particular:

- the quality system documentation,
- the technical documentation referred to in paragraph 2,
- the quality records, such as inspection reports and test data, calibration data, qualification reports on the personnel concerned.

6.3.1 The notified body shall carry out periodic audits to make sure that the manufacturer maintains and applies the quality system and shall provide the manufacturer with an audit report.
6.3.2 For compost belonging to component material category ('CMC') 3 ~~and~~, digestate belonging to CMC 5, precipitated phosphate salts & derivates belonging to CMC XX, thermal oxidation materials & derivates belonging to CMC YY and pyrolysis & gasification materials belonging to CMC ZZ, as defined in Annex II, the notified body shall take and analyse output material samples during each audit, and the audits shall be carried out with the following frequency:
(a) During the notified body's first year of surveillance of the plant in question: The same frequency as the sampling frequency indicated in the table included in paragraph 5.1.3.1(f); and
(b) During the following years of surveillance: Half the sampling frequency indicated in the table included in paragraph 5.1.3.1(f).
6.4 In addition, the notified body may pay unexpected visits to the manufacturer. During such visits the notified body may, if necessary, carry out product tests, or have them carried out, in order to verify that the quality system is functioning correctly. The notified body shall provide the manufacturer with a visit report and, if tests have been carried out, with a test report.

7. Conformity marking and EU declaration of conformity

7.1. The manufacturer shall affix the CE marking and, under the responsibility of the notified body referred to in paragraph 5.2, the latter's identification number to each individual product that satisfies the applicable requirements of this Regulation.
7.2.1 The manufacturer shall draw up a written EU declaration of conformity for each EU fertilising product lot and keep it at the disposal of the national authorities for 10 years after the EU fertilising product has been placed on the market.
The EU declaration of conformity shall identify the product lot for which it has been drawn up.
7.2.2. A copy of the EU declaration of conformity shall be made available to the relevant authorities upon request.

8. Availability of quality system documentation

8. The manufacturer shall, for a period ending at least 10 years after the product has been placed on the market, keep at the disposal of the national authorities:

- the documentation referred to in paragraph 5.3,
- the change referred to in paragraph 5.6 and subparagraphs, as approved,
- the decisions and reports of the notified body referred to in paragraph 5.6.1-5.6.3, paragraph 6.3 and paragraph 6.4.

9. Notified bodies' information obligation

9.1. Each notified body shall inform its notifying authorities of quality system approvals issued or withdrawn, and shall, periodically or upon request, make available to its notifying authorities the list of quality system approvals refused, suspended or otherwise restricted.
9.2. Each notified body shall inform the other notified bodies of quality system approvals which it has refused, suspended or withdrawn, and, upon request, of quality system approvals which it has issued.

10. Authorised representative

The manufacturer's obligations set out in paragraph 3, paragraph 5.2, paragraphs 5.6.1-5.6.3, Heading 7 and Heading 8 may be fulfilled by his or her authorised representative, on his or her behalf and under his or her responsibility, provided that they are specified in the mandate.

5.7.3 Standards

The presumption of conformity to a legal provision conferred by conformity to a *Harmonised Standard* should enhance recourse to compliance with the technical requirements.

Regulation (EU) No 1025/2012 provides definitions for the terms *'Standard'*, *'National Standard'*, *'European Standard'*, *'Harmonised Standard'*, and *'International Standard'*.

- '*Standards*' are defined as technical specifications, adopted by a recognised standardisation body, for repeated or continuous application, with which compliance is not compulsory.
- '*European Standards'* are *'Standards'* adopted by the European standardisation organisations (ESOs) listed in Annex I to Regulation (EU) No 1025/2012 (155). CENELEC is an EUropean regional standards organisation that together with its sister organisations CEN, the European Committee for Standardization, and ETSI, the European Telecommunications Standards Institute, compose the so-called and known European Standards Organizations (ESOs) that are officially recognised by the European Commission and act as a EUropean platform through which *European Standards* are developed. In the European Union, only *Standards* developed by CEN, CENELEC and ETSI are recognised as '*European Standards*'. CEN and CENELEC are the regional mirror bodies to their international counterparts, e.g. ISO (the International Organization for Standardization).
- Taking into account the first two definitions mentioned above, '*Harmonised Standards*' are '*European Standards*' adopted, upon a request made by the Commission, for the application of Union harmonisation legislation. *Harmonised Standards* maintain their status of voluntary application. The definition for a '*Harmonised Standard*', within the context of Regulation (EU) No 1025/2012, is not restricted to *Harmonised Standards* supporting harmonised product legislation as the Regulation prioritises the use of *Harmonised Standards* in harmonisation legislation for services in similar way as in Union harmonisation legislation for products.

The STRUBIAS subgroup highlighted the need to test and validate existing *Harmonised Standards*, and possibly develop new ones, for each of the STRUBIAS material groups. Therefore, the present list of possible *European Standards* is not exhaustive. Note that the list is restricted to *European Standards* and that (*national) Standards* are not included.

At present, the following *Standards* and measurement methods are available for the parameters that are included in the proposals for the technical requirements (note: the list is non-exhaustive):

- *Sampling*
  EN 1482-1 and 2: Fertilizers and liming materials – sampling and sample preparation – part 1: sampling and part 2: sample preparation
  EN ISO 10249: Fluid fertilizers – preliminary visual inspection and preparation of samples for physical testing
  EN 1482-3:2016: Fertilizers and liming materials – sampling and sample preparation – Part 3: sampling of static heaps
  CEN/TR 17040: Fertilizers and liming materials - Sampling of static heaps - Technical report on experimental sampling trials performed under mandate M/454

- *Total phosphorus*
  EN 15956: Extraction of phosphorus soluble in mineral acids
  EN 15959: Fertilizers - Determination of extracted phosphorus
  EN 16173: Sludge, treated bio-waste and soil - Digestion of nitric acid soluble fractions of elements
  EN 16174: Sludge, treated bio-waste and soil - Digestion of aqua regia soluble fractions of elements
  CEN/TS 16170: Sludge, treated bio-waste and soil – Determination of elements using inductively coupled plasma optical emission spectroscopy (ICP-OES)
  CEN/TS 16171: Sludge, treated bio-waste and soil – Determination of elements using inductively coupled plasma mass spectroscopy (ICP-MS)

- *Macroscopic impurities*
  CEN/TS 16202: Sludge, treated bio-waste and soil - Determination of impurities and stones

- *Biological pathogens*
  CEN/TR 16193: Sludge, treated bio-waste and soil – detection and enumeration of E. Coli
  CEN/TR 15214-1: Characterization of sludges - Detection and enumeration of Escherichia coli in sludges, soils, soil improvers, growing media and bio-wastes - Part 1: Membrane filtration method for quantification
  CEN/TR 15214-2: Characterization of sludges - Detection and enumeration of Escherichia coli in sludges, soils, soil improvers, growing media and bio-wastes - Part 2: Miniaturised method (Most Probable Number) by inoculation in liquid medium
  CEN/TR 15214-3: Characterization of sludges - Detection and enumeration of Escherichia coli in sludges, soils, soil improvers, growing media and bio-wastes - Part 3: Macromethod (Most Probable Number) in liquid medium
  EN 15788:2009: Animal feeding stuffs - Isolation and enumeration of Enterococcus (E. faecium) spp.
  prEN 15788 rev: Animal feeding stuffs: Methods of sampling and analysis - Isolation and enumeration of Enterococcus (E. faecium) spp.

EN ISO 7899-2:2000: Water quality - Detection and enumeration of intestinal enterococci - Part 2: Membrane filtration method (ISO 7899-2:2000)

CEN/TR 15215-1: Characterization of sludges - Detection and enumeration of *Salmonella* spp. in sludges, soils, soil improvers, growing media and bio-wastes - Part 1: Membrane filtration method for quantitative resuscitation of sub-lethally stressed bacteria (to confirm efficacy of log drop treatment procedures)

CEN/TR 15215-2: Characterization of sludges - Detection and enumeration of *Salmonella* spp. in sludges, soils, soil improvers, growing media and bio-wastes - Part 2: Liquid enrichment method in selenite-cystine medium followed by Rapport-Vassiliadis for semi-quantitative Most Probable Number (MPN) determination.

CEN/TC 308 CEN/TR 15215-3: Characterization of sludges - Detection and enumeration of *Salmonella* spp. in sludges, soils, soil improvers, growing media and bio-wastes - Part 3: Presence/absence method by liquid enrichment in peptone-novobiocin medium followed by Rapport-Vassiliadis

EN ISO 14189:2016: Water quality - Enumeration of Clostridium perfringens - Method using membrane filtration (ISO 14189:2013)

EN ISO 7937:2004: Microbiology of food and animal feeding stuffs - Horizontal method for the enumeration of *Clostridium perfringens* - Colony-count technique (ISO 7937:2004)

USEPA. Test Method for Detecting, Enumerating, and Determining the Viability of *Ascaris* Ova in Sludge. 1999. pp. 616–622. (*Note: not a EUropean Standard*)

- *Total organic carbon and hydrogen*
  EN 15936: Sludge, treated bio-waste, soil and waste - Determination of total organic carbon (TOC) by dry combustion
  ISO 29541: Determination of total carbon, hydrogen and nitrogen in coal and coke by instrumental methods
  ISO 925: Determination of carbonate carbon content -- Gravimetric method (*Note: determination of inorganic carbon; total organic carbon can then be calculated as the difference between total carbon minus total inorganic carbon*)

- *Moisture content and dry matter content*
  EN 12048:1996: Solid fertilizers and liming materials - Determination of moisture content - Gravimetric method by drying at (105 +/- 2)°C (ISO 8190:1992 modified)
  ISO/DIS 19745: Fertilizers and soil conditioners -- Determination of crude (free) water content of ammoniated phosphate products -- DAP, MAP -- by gravimetric vacuum oven at 50 °C
  EN 13466-1: Fertilizers - Determination of water content (Karl Fischer methods) - Part 1: Methanol as extracting medium
  EN 14787: Fertilizers and liming materials - Determination of water content - Guidelines and recommendations
  EN 13040: Soil improvers and growing media - Sample preparation for chemical and physical tests, determination of dry matter content, moisture content and laboratory compacted bulk density
  EN 15934: Sludge, treated bio-waste, soil and waste - calculation of dry matter fraction after determination of dry residue or water content

- *Persistent organic pollutants*
  CEN/TS 16181: Sludge, treated bio-waste and soil - Determination of polycyclic aromatic hydrocarbons (PAH) by gas chromatography (GC) and high performance liquid
  EN 15527: Characterization of waste - Determination of polycyclic aromatic hydrocarbons (PAH) in waste using gas chromatography mass spectrometry (GC/MS)
  ISO 13877: Soil quality -- Determination of polynuclear aromatic hydrocarbons -- Method using high-performance liquid chromatography
  EN 16181: Soil, treated bio-waste and sludge - Determination of polycyclic aromatic hydrocarbons (PAH) by gas chromatography (GC) and high performance liquid chromatography (HPLC)
  CEN/TS 16190: Sludge, treated bio-waste and soil - Determination of dioxins and furans and dioxin-like polychlorinated biphenyls by gas chromatography with high resolution mass selective detection (HR GC-MS)
  prEN 16190: Soil, treated bio-waste and sludge - Determination of dioxins and furans and dioxin-like polychlorinated biphenyls by gas chromatography with high resolution mass selective detection (HR GC-MS)

- *Metals and metalloids*
  EN 16173: Sludge, treated bio-waste and soil - Digestion of nitric acid soluble fractions of elements
  EN 16170: Sludge, treated bio-waste and soil - Determination of elements using inductively coupled plasma optical emission spectrometry (ICP-OES)
  EN 16171: Sludge, treated bio-waste and soil - Determination of elements using inductively coupled plasma mass spectrometry (ICP-MS)
  EN ISO 16968: Solid biofuels - Determination of minor elements
  EN 16319: Fertilizers – Determination of trace elements – determination of cadmium, chromium, lead and nickel by inductively coupled plasma-atomic emission spectrometry (ICP-AES) after aqua regia dissolution

- *Chloride and chlorine*
  EN 16195: Fertilizers - Determination of chlorides in the absence of organic material
  ISO 587: Solid mineral fuels – Determination of chlorine, Eschka method

- *Neutralising value*
  EN 12945: Determination of neutralizing value

- *Determination of the pH value*
  EN 13037: Soil improvers and growing media – determination of pH
  EN 15933: Sludge, treated bio-waste and soil – determination of pH

5.8 Other EU legislation of interest

A list of relevant EU legislation in relation to fertilising products is available in Annex V to the impact assessment accompanying the EU Fertilising Products Regulation ((EU) 2019/1009) of 17 March 2016. STRUBIAS materials that are in line with the recovery rules may become CMCs in the EU Fertilising Products Regulation ((EU) 2019/1009) and thus be used as ingredients for fertilising products. Additionally, the producers of the STRUBIAS materials may have to comply, amongst others, with EU legislation related to **waste management and shipment** (e.g. Waste Framework Directive - 2008/98/EC; Waste Shipment Regulation - 96/61/EC), **animal by-products and derived materials** (e.g. Regulation (EC) No 1069/2009, Regulation (EU) 142/2011, see also Section 5.2.6), **containment of emissions to the environment** (e.g. Industrial Emissions Directive - 2010/75/EU, Surface Water Directive - 75/440/EEC, Air Quality Directive - 2008/50/EC, Nitrates Directives - 91/676/EEC), **EU plant health legislation** (2000/29/EC), replaced by 2016/2031/EC from 15 December 2019 onwards), **control of hazards** (e.g. Council Directive 96/82/EC on the control of major-accident hazards involving dangerous substances), **safety of workers** during production processes (e.g. Council Directive 2013/59/Euratom of 5 December 2013 laying down basic safety standards for protection against the dangers arising from exposure to ionising radiation) and **transport** (e.g. Directive 2006/94/EC of the European Parliament and of the Council of 12 December 2006 on the establishment of common rules for certain types of carriage of goods by road).

STRUBIAS materials will likely **become products** when used as substances on their own or in mixtures with other CMCs when compliant with all requirements laid down for the corresponding PFC, and their placing on the market, application and use shall then have to comply with the legal framework of the **CLP Regulation** (‘Classification, Labelling and Packaging’, Regulation (EC) No 1272/2008) and **REACH Regulation (EC) No 1907/2006**.

Finally, any STRUBIAS materials applied on land will have to comply with all legislation related to **nutrient use and management in crop and livestock production** (e.g. CAP - Common Agricultural Policy; Nitrates Directive - 91/676/EEC), **biodiversity** (e.g. Habitats Directive - 92/43/EEC), and **containment of water pollution** (e.g. Water Framework Directive - 2000/60/EC).

PART B: MARKET STUDY

6 Agronomic efficiency

6.1 Introduction

Knowledge of the agronomic efficiency of STRUBIAS materials is critical to evaluate the added value of STRUBIAS materials in the EU Fertilising Products Regulation ((EU) 2019/1009). Moreover, it is also the key starting point in any assessment of the following impacts:

- Environment and human health impacts: the application rates for STRUBIAS fertilisers needed to achieve the same agronomic yields relative to typical present-day fertilising products will depend on the fertiliser efficiency. The contaminant load associated with the use of STRUBIAS materials thus not only depends on the concentration of the contaminants in the material, but also on the agronomic efficiency.
- Market impact: the price setting and development of the market share of any STRUBIAS materials will depend on what agronomic value they provide to the user, compared to alternative fertilising materials on the market.

In this section, the agronomic value of fertilising products containing recovered materials is evaluated for **different soils and plant types prevalent in the European context**. For this purpose, meta-analyses were performed that assessed the **fertiliser efficiency** of fertilisers derived from STRUBIAS materials. The term meta-analysis refers to **a statistical analysis of combined data from a series of well-conducted primary studies, in order to obtain a more precise estimate that reduces the size of the confidence interval of the underlying 'true effect' in comparison to any individual study** (Pogue and Yusuf, 1998; Garg et al., 2008). Meta-analysis techniques enable it to be established whether the scientific findings are **consistent and generalisable** across settings and facilitate an understanding of the reasons why some studies differ in their results. For these reasons, a meta-analysis of similar, well-conducted, randomised, controlled trials has been considered one of the highest levels of evidence (Garg et al., 2008).

6.2 P-fertilisers containing STRUBIAS materials

6.2.1 Meta-analysis approach

In the EU Fertilising Products Regulation ((EU) 2019/1009), **mined and synthetic inorganic fertilisers are considered in PFC 1**. The overall share of these materials is included in the category 'Straight solid inorganic macronutrient fertiliser' and 'Compound solid inorganic macronutrient fertiliser'. A P_2O_5 lower limit value of 12% is considered for the first category, whereas the second category requires a minimum P_2O_5 content of 3% P_2O_5 plus the presence of one of the other considered plant macronutrients (K_2O, MgO, N, CaO, SO_3, or Na_2O). Solid organic (> 15% organic C) and organo-mineral (> 7.5% organic C) P-fertilisers require a minimum P_2O_5 content of 2% (0.9% P). Therefore, **this assessment focuses on STRUBIAS materials that have a minimum P_2O_5 content of > 2%**.

The agronomic efficiency of fertilisers was assessed using two different **plant response variables** (Figure 4):

i. **The plant dry matter yield (DMY)**: This is the most common response parameter documented in studies. Comparing the absolute values for DMY from F_{prim} and F_{sec} (referring to P fertilisers derived from primary and secondary raw materials, respectively), provides precise information on the different plant biomass responses in function of the fertiliser type.

ii. **The phosphorus use efficiency (PUE)**: Plant P uptake efficiency is calculated as the difference in P uptake between fertilised (PU_F) and unfertilised plants (PU_C), expressed relative to the amount of fertiliser P applied ($P_{applied}$):

$$PUE = (PU_F - PU_C) / P_{applied} = \Delta PU / P_{applied}$$

This parameter takes into account that the consumer valuation of P-fertilisers equals the marginal yield increase relative to an unfertilised treatment. The disadvantage of this parameter is, however, a higher degree of uncertainty due to error propagation because unfertilised treatments have to be subtracted during parameter calculation. Therefore, only pairwise results that documented a significant increase in plant uptake relative to control for P-fertilisers derived from phosphate rock were taken into consideration (see Section 16.2). The exclusion rate, determined as the number of cases that were excluded from the original database, is equal for fertilisers derived from primary and secondary raw materials, safeguarding an unbiased assessment (see Section 16.2 for methodological details).

This study compares plant responses to P fertilisers derived from primary and secondary raw materials (F_{prim} and F_{sec}). For F_{sec}, an assessment has been made for each of the three STRUBIAS materials. The agronomic efficiency of fertilisers containing STRUBIAS materials is expressed relative to mineral P-fertilisers; the resulting ratio is referred to as '**relative agronomic efficiency (RAE)**' (Figure 4):

$$RAE_{DMY} = DMY_{Fsec} / DMY_{Fprim}$$

and

$$RAE_{PUE} = PUE_{Fsec} / PUE_{Fprim} = \Delta PU_{Fsec} / \Delta PU_{Fprim}$$

Note that the P application rate ($P_{applied}$) is levelled by dividing PUE_{Fsec} by PUE_{Fprim} to calculate RAE_{PUE}.

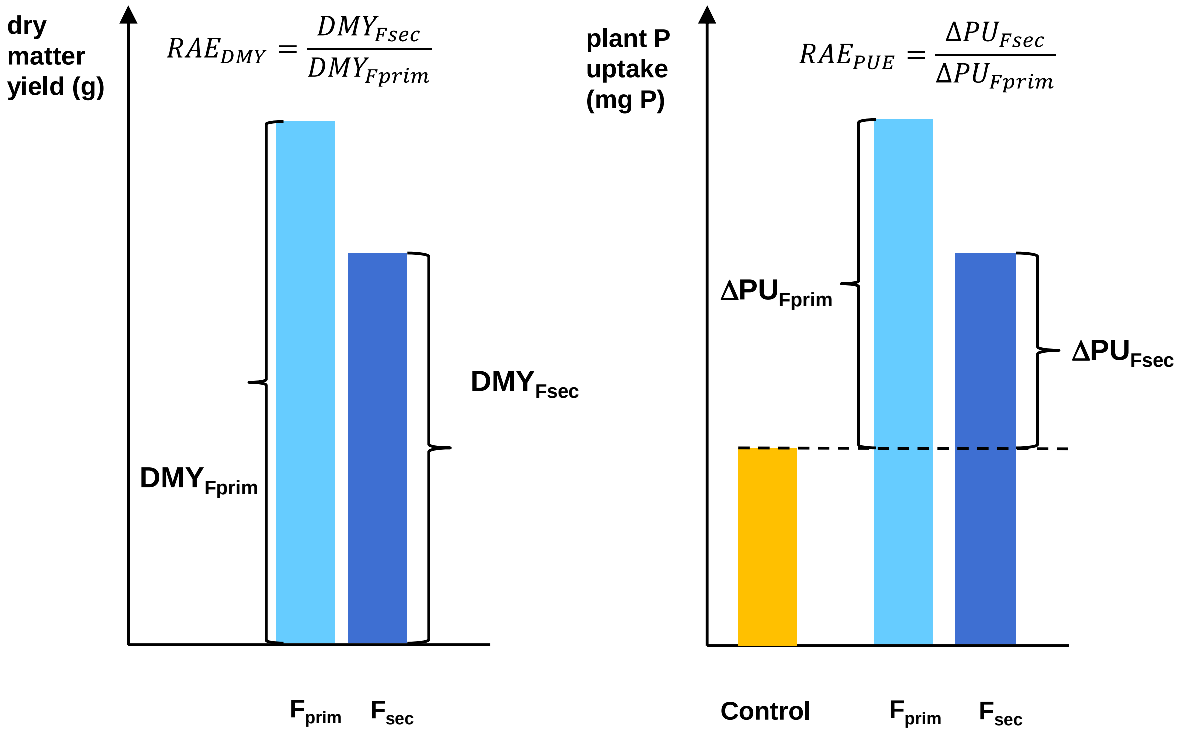


Figure 4: Schematic outline of the plant response variables used to calculate the relative agronomic efficiencies RAE$_{DMY}$ and RAE$_{PUE}$ following treatment with fertilisers from primary materials (F$_{prim}$) or secondary materials (F$_{sec}$)

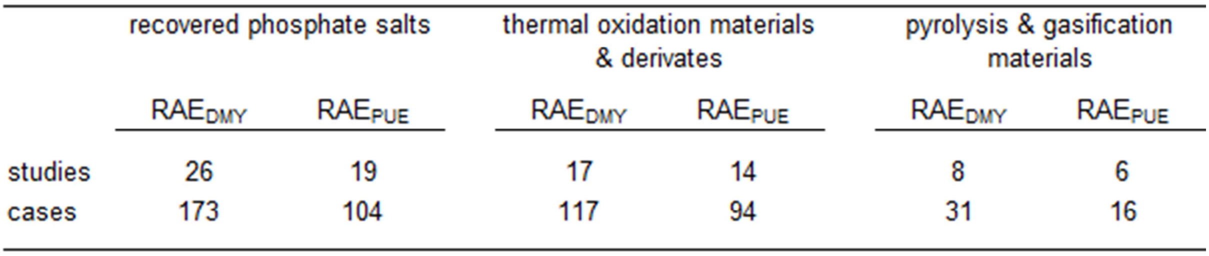
A relative agronomic efficiency value below 1 indicates that that the fertiliser derived from STRUBIAS materials is a less effective plant P-source than a synthetic P-fertiliser derived from mined phosphate rock, and vice versa.

Data were grouped prior to meta-analysis to enable a broad-ranging assessment of the fertilising effectiveness of F$_{sec}$ as a function of soil type, plant group, feedstock used for F$_{sec}$ production, and variables related to the experimental design of the study (e.g. fertiliser regime, pot versus field trial). These parameters that distintinguish between groups were referred to as **grouping variables**. The relative agronomic efficiency for both response variables (RAE$_{DMY}$ and RAE$_{PUE}$) was then calculated for a number of **‘cases’** where all grouping variables (such as soil type and crop grown, crop harvest time, P application rate) are identical for both fertiliser treatments. Hence, the sole divergent variable for each case is the type of P-fertiliser.

Results were collected from the information provided by the STRUBIAS subgroup and from scientific literature. The number of studies and ‘cases’ for precipitated phosphate salts, thermal oxidation materials & derivates and pyrolysis & gasification materials is indicated in Table 8. A significant number of studies were available for precipitated phosphate salts & derivates and thermal oxidation materials & derivates, whereas data coverage for pyrolysis & gasification materials was poor. Therefore, no hard conclusions on the agronomic efficiency of pyrolysis & gasification materials could be made. Hence, the **results for pyrolysis &**

gasification materials provide only a *preliminary* assessment and should be interpreted with the necessary caution. The complete methodology and the references to the original works used for this study are given in Section 16.2.

Table 8: Number of studies and cases included for the meta-analyses on the relative agronomic efficiency of P-fertilisers derived from precipitated phosphate salts & derivates, thermal oxidation materials & derivates and pyrolysis & gasification materials



	recovered phosphate salts		thermal oxidation materials & derivates		pyrolysis & gasification materials	
	RAE_{DMY}	RAE_{PUE}	RAE_{DMY}	RAE_{PUE}	RAE_{DMY}	RAE_{PUE}
studies	26	19	17	14	8	6
cases	173	104	117	94	31	16

Results in the following sections are represented as ‘forest plots’ that graphically indicate the RAE_{DMY} (left figure) and RAE_{PUE} (right figure). The bars cover the 95% confidence interval, so error bars that cross the vertical 1 line indicate that F_{sec} is not significantly different from F_{prim}.

6.2.2 Precipitated phosphate salts & derivates

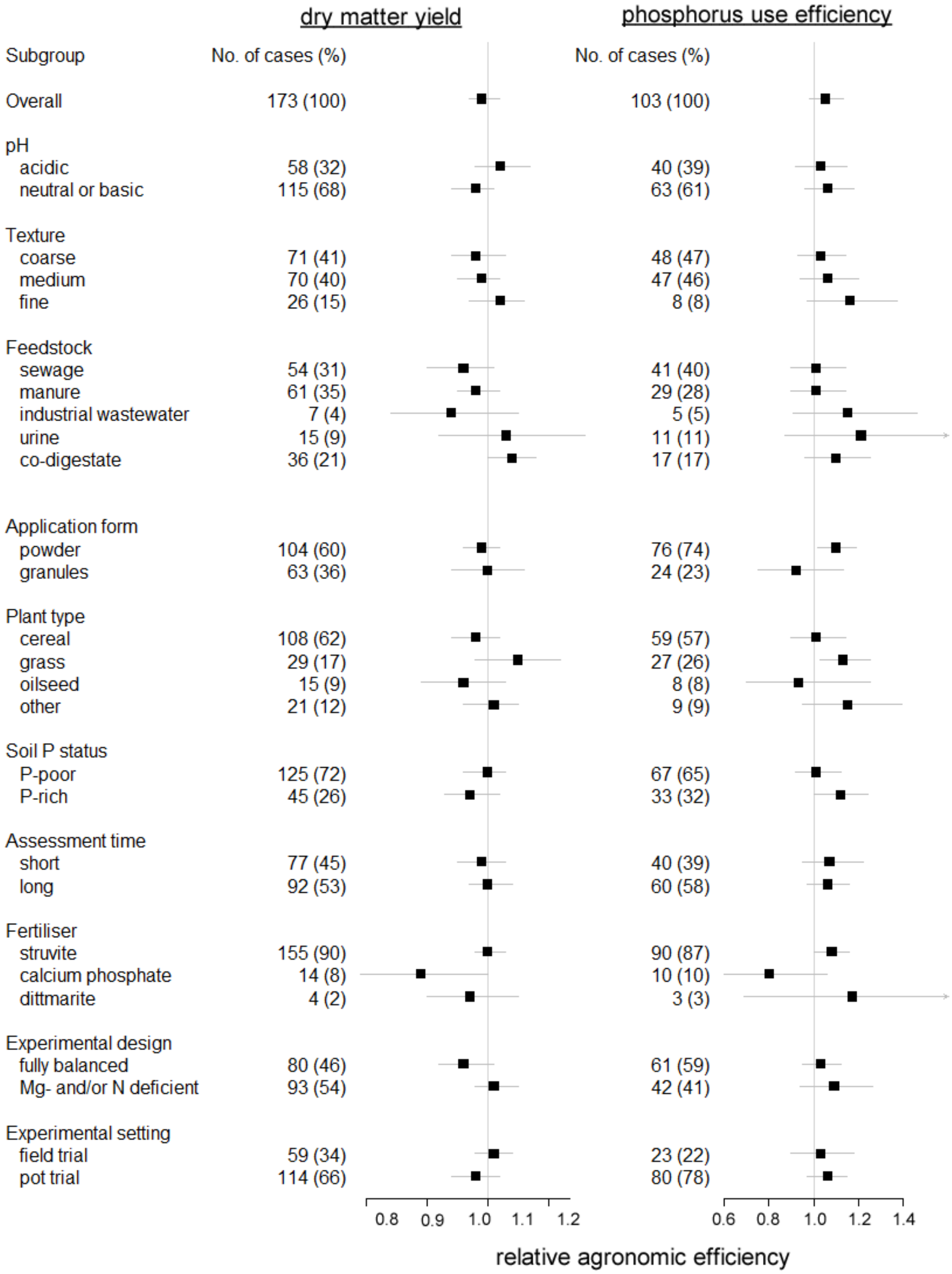
The overall results indicated a similar agronomic efficiency for precipitated phosphate salts & derivates to mined rock phosphate and processed P-fertilisers. The mean values of RAE_{DMY} and RAE_{PUE} equal 0.99 and 1.05, respectively (Figure 5), with the corresponding 95% confidence intervals overlapping the 1 value for both parameters. Regardless of soil pH, soil texture, feedstock, application form, plant type, soil P status, assessment time, and experimental design and setting, RAE_{DMY} and RAE_{PUE} values for precipitated phosphate salts & derivates were not significantly different from 1. The RAE_{DMY} and RAE_{PUE} for struvite and dittmarite were not significantly different from 1, but the 95% confidence interval for RAE_{DMY} of calcium phosphates (grouping variable fertiliser) extended to a value marginally below 1 (0.995; Figure 5). No significant differences across selected groups were observed at the 95% level, although the effect of plant type was marginally significant (P: 0.06; data not shown) for RAE_{DMY}.

The analysis indicated that the **agronomic efficiency of precipitated phosphate salts is equal to that of mined and synthetic fertilisers. These results are consistent and generalisable across different settings, including soil and crop types, relevant for the European agricultural sector.** Although multi-year assessments fall beyond the scope of this meta-analysis, the results of Thompson (2013) and Wilken et al. (2015) confirm the sustained long-term efficiency of precipitated phosphate salts as a P-fertiliser.

Struvite is the most common precipitated phosphate salt, but some P-recovery processes target a **different end material** such as dittmarite or dicalcium phosphates. The crystallisation of calcium phosphates may involve the formation of metastable precursor phases, such as octocalcium phosphate and hydroxyapatite, which are less available to plants, especially at alkaline pH (Wang and Nancollas, 2008). Hence, the RAE of calcium phosphates can vary depending on the exact composition of the calcium phosphate phases included in the end material. After application to the soil, calcium phosphates can also transform into more stable forms (Arai and Sparks, 2007), potentially further contributing to the wider RAE ranges observed for calcium phosphates than for struvite and dittmarite.

Unlike most mined and synthetic P-fertilisers, precipitated phosphate salts are water-insoluble, but their solubility is increased in acid solutions (Wilken et al., 2015). Nonetheless, our results indicated that **soil pH had no significant effect on the relative agronomic efficiency**. Achat et al. (2014a) indicated that isotopically exchangeable P was similar for finely ground struvite to that for triple superphosphate, irrespective of the pH in the range of 5.2-8.1. Talboys et al. (2016) indicated that the short-term (< 42 days) dissolution of granulated struvite, the most common precipitated phosphate salt, shows similar dynamics across a wider soil pH range of 5.0-8.0. Degryse et al. (2017) indicated a 60-day granulated struvite dissolution rate of > 80% in an acid soil (pH 5.9), but < 10% dissolution in a basic soil (pH 8.5). Hence, as most European soils have a pH between 5 and 8 (Reuter et al., 2008), soil pH is not expected to exert a major influence over the dissolution patterns of precipitated phosphate salts and the relative agronomic efficiency. Plants also modify the rhizosphere pH as they exude organic acids from their root biomass in significant quantities that can drastically lower pH in the plant root microenvironment.

Talboys et al. (2016) indicated that organic acids have a major impact on the rate of dissolution of P from struvite, and that plants with root systems that exude large quantities of organic acids are more effective at taking up P from struvite granules. The exudates cause the dissolution of the precipitated phosphate salts in the vicinity of the plant root. Grasses exude significantly more organic acids than common crops; estimates for the total allocation of photosynthates – a proxy for rhizodeposition - to roots are 50-70% higher for grasses than for cereals such as wheat and barley (Kuzyakov and Domanski, 2000). Hence, species-specific patterns of root exudation may explain the variations in relative agronomic efficiencies observed, but the effect of plant type is not significant overall (Figure 5).



NB: Results are presented as weighted mean (square) and 95% confidence intervals (error bars).

Figure 5: The relative agronomic efficiency of precipitated phosphate salts & derivates for the plant response variables DMY (dry matter yield) and PUE (phosphorus use efficiency) as a function of grouping variables

No significant effect of **assessment time and application form** on the relative agronomic efficiency along a single plant growing season was observed for precipitated phosphate salts (Figure 5). Although the slower initial P release rate from the granulated fertiliser could possibly reduce plant uptake of P during the very initial plant growth stages (< 36 days; Degryse et al. 2017; Talboys et al. 2016), studies that applied an assessment time between 36 and 65 days showed good performance when precipitated phosphate salts were applied. For crops subject to struvite fertilisation, it is has been suggested that a reduction in the number of grain heads due to short-term P deficiency is counterbalanced by the crop root system's capacity to take up P in the later plant growth stages (Talboys et al., 2016). Hence, even for studies with an assessment time < 65 days, the sustained P release from precipitated phosphate salts could possibly compensate their lower initial P-availability and their lower P-dissolution rate relative to water-soluble P-fertilisers (Talboys et al., 2016; Degryse et al., 2017). The relative agronomic efficiencies for dry matter yield and P use efficiency were not significantly different from 1 for struvite and dittmarite, but the 95% confidence interval for calcium phosphates (grouping variable fertiliser) extended to a value marginally below 1 for dry matter yield (0.995; Figure 5). Struvite is the most common precipitated phosphate salt, but some P-recovery processes target a different end material such as dittmarite or dicalcium phosphates. The crystallisation of calcium phosphates may involve the formation of metastable precursor phases, such as octocalcium phosphate and hydroxyapatite, which are less available to plants, especially at alkaline pH (Wang and Nancollas, 2008). Hence, the relative agronomic efficiency of calcium phosphates can vary depending on the exact composition of the calcium phosphate phases included in the end material. After application to the soil, calcium phosphates can also transform into more stable forms (Arai and Sparks, 2007), potentially further contributing to the wider relative agronomic efficiency ranges observed for calcium phosphates than for struvite and dittmarite.

In line with the observation that **feedstock** does not have a major impact on the chemical composition of the precipitated phosphate salts & derivates, no input-material-specific impacts on RAE were observed.

6.2.3 Thermal oxidation materials & derivates

The overall mean effects for thermal oxidation materials & derivates were 0.92 and 0.81 for RAE_{DMY} and RAE_{PUE}, respectively (Figure 6). The 95% confidence intervals for both response variables indicated that the agronomic efficiency for thermal oxidation materials & derivates was overall lower than for mined and synthetic fertilisers (Figure 6; 95% confidence intervals do not cross the RAE value of 1). The analyses for the different grouping variables indicated significant effects of feedstock, post-processing groups and assessment time for RAE_{DMY} and RAE_{PUE} ($P < 0.001$; data not shown). Thermal oxidation materials & derivates derived from sewage sludge showed a significantly lower RAE_{DMY} and RAE_{PUE} than for thermal oxidation materials & derivates derived from crop residues and poultry litter (Figure 6), but it should be taken into consideration that sewage-sludge-derived thermal oxidation materials & derivates include both raw ashes and ashes that have been post-processed. Ashes that have been post-processed using wet-digestion and thermal manufacturing steps to improve their plant P-availability and reduce inorganic contaminants showed significantly greater RAE_{DMY} and RAE_{PUE} than raw sewage sludge ashes (Figure 6). The RAE_{DMY} values were 1.03 and 0.93 for materials subjected to wet-digestion and thermal post-processing steps (Figure 6). The RAE of F_{sec} derived from crop residues, poultry litter and pig manure did not differ from F_{prim} (Figure 6), but results for pig manure should be interpreted with caution because of the low number of cases. Thermal oxidation materials & derivates derived from wood and steel slags showed a low RAE_{DMY} and RAE_{PUE}, but the results should be interpreted with caution because of the low number of cases (Figure 6). A significant effect of assessment time on RAE_{DMY} and RAE_{PUE} was observed ($P < 0.001$; Figure 6), with values that are 20% (RAE_{DMY}) and 40% (RAE_{PUE}) lower in the long term (> 65 days) than in the short term (< 65 days). A significant effect of experimental design (*P*: 0.04) and experimental setting (*P*: 0.003) was observed for RAE_{PUE} (Figure 6). No significant effects of soil pH, soil texture, plant type and soil P status were observed, although the effect of soil pH on RAE_{PUE} was marginally significant (*P*: 0.08) (Figure 6).

Significant differences in the relative agronomic efficiency of thermal oxidation materials & derivates were observed, primarily **dependent on the feedstock applied and the possible post-processing steps that were performed**. Thermal oxidation materials & derivates consist of P-fertilisers with heterogeneous properties that control their behaviour and agronomic impacts in soils. Moreover, it should be taken into consideration that this study did not include fertilising products that are F_{sec} ash derivates (e.g. Ecophos® process, ICL RecoPhos® process, acidulation process; see Huygens et al. (2016) and Egle et al. (2016)) with the same chemical composition as that of F_{prim}. For such F_{sec}, an RAE value of 1 can reasonably be expected. The observed RAE results are not affected by soil pH, soil texture, application form, or soil P status, and different F_{sec} groups produced from a variety of feedstocks have an agronomic efficiency that is not significantly different from F_{prim}. Hence, **these observations confirm that thermal oxidation materials & derivates can deliver an effective alternative to mined rock phosphate and processed P-fertilisers in European agriculture, but that the RAE is dependent on the properties of the end material produced.**

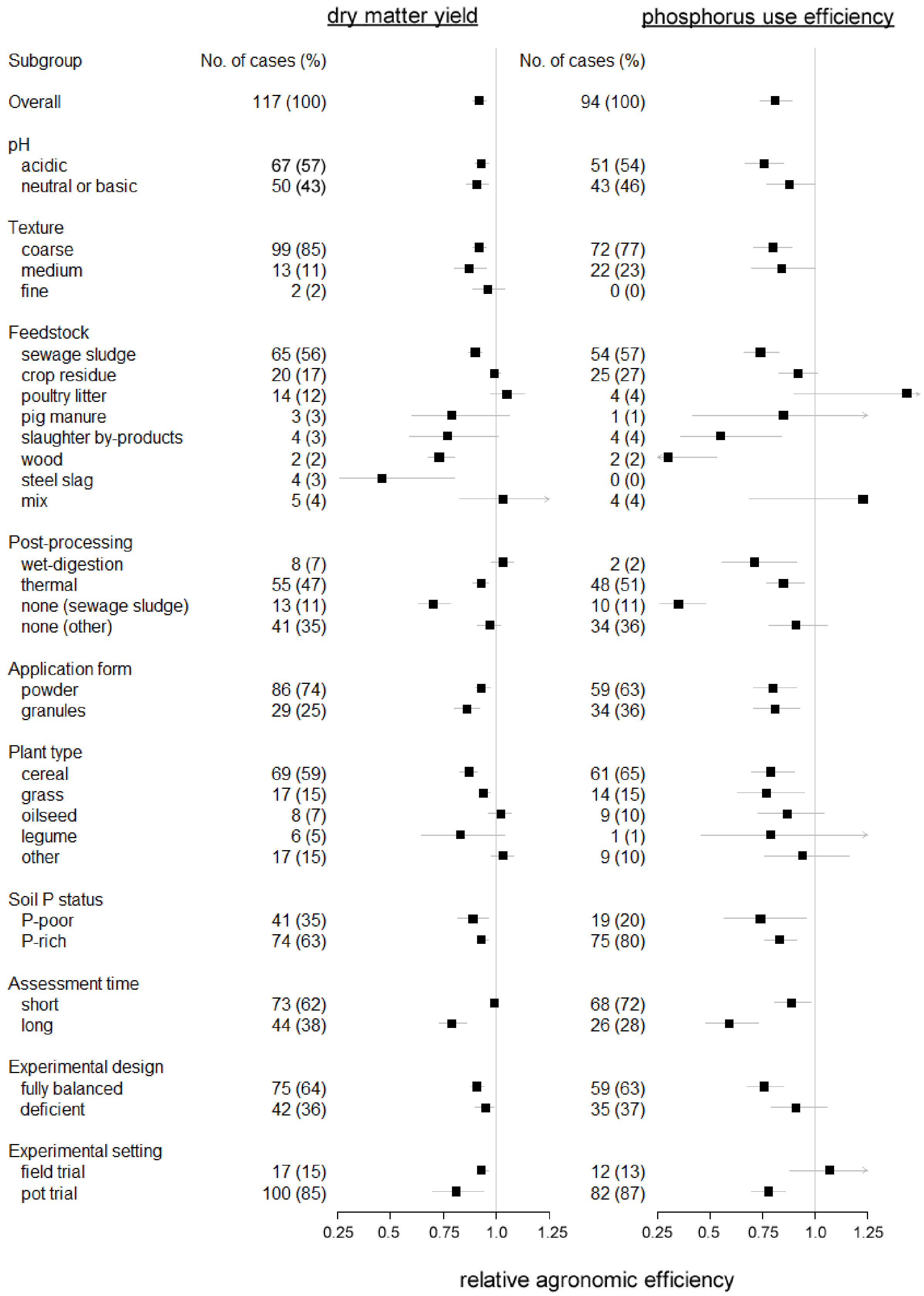
The impact of pH on the P-dissolution depends on the elemental composition of the P-fertiliser because P is strongly bond to Ca at high pH and to Fe and Al at low pH (Hinsinger, 2001; Tóth et al., 2014). Nonetheless, the high basic cation contents of some thermal oxidation materials might **buffer the acidity effect of the soil micro-environment, thus obscuring the effect of the soil pH**. Also, no consistent differences were observed in relative agronomic efficiency across plant types for the response variables, indicating that possible differences in root exudation patterns of organic acids do not meaningfully impact the P-release patterns from thermal oxidation materials & derivates.

The RAE varies considerably as a function of **feedstock, but these results require a cautionary interpretation as sample sizes are low for most groups, other than sewage**

sludge. Crop residues show a high RAE value, but it should be considered that most results are derived from three studies that used a similar soil type (Schiemenz and Eichler-Löbermann, 2010; Schiemenz et al., 2011; Delin, 2016). For sewage sludge ashes, a post-processing step is often applied to increase P-availability, and to comply with legislative limit values for metals and metalloids. This analysis confirms that such **manufacturing processes starting from sewage sludge mono-incineration ashes clearly improve the plant availability relative to unprocessed sewage sludge ashes, and enable the transformation of sewage sludge ashes into efficient P-fertilisers.** Relative agronomic efficiencies close to 1 can reasonably be expected for materials resulting from wet-digestion post-processing, especially for these that have the same chemical composition as that of mined rock phosphate and processed P-fertilisers. Thermal post-processing steps aim at separating P from other elements and to influence the crystal structure of the materials by isomorphic substitution of the PO_4^{3-} ionic group (by for example SiO_4^{2-} or CO_3^{2-}), affecting the reactivity of the final product and therefore plant P availability. The final products show similar characteristics to Thomas phosphate and Rhenania phosphate, and show overall good fertiliser efficiency.

Thermal oxidation materials and derivates **perform better in short-term experiments than in long-term studies** (> 65 days) (Figure 6). The plant-availability of the P in P-fertilisers is likely controlled by the coordinated cations of Ca, Mg Al and Fe to which PO_4^{3-} is bound. All these different ions are abundantly present in thermal oxidation materials & derivates, although their relative abundance varies across end materials. Complexes between phosphate and K, Ca, Mg and S ions are relatively easily decomposed (Hinsinger, 2001; Tóth et al., 2014), and this more labile P-fraction is therefore likely to be released in the short term. Phosphate may, however, be unavailable to plants when strongly bound to particular trivalent cations in a stable matrix (Barrow, 1984; Hinsinger, 2001). The release of P from this more stable fraction could be limited, effectively decreasing the long-term P supply from thermal oxidation materials & derivates. This contrasts with mined and synthetic fertilisers that are of a uniform chemical composition; such fertilisers can be expected to release P readily upon physical disintegration. The released P that is not readily taken up by plants can be adsorbed to soil minerals, with the nature of such reactions dependent on the pH and on the concentration of metal cations such as Ca, Fe and Al as well as organic and inorganic ligands (Hinsinger, 2001; Tóth et al., 2014). At a later time in the plant growing season, desorption of adsorbed P can occur via ligand exchange reactions, especially if the P was bound in more labile soil P-complexes (Hinsinger, 2001). Such desorption processes could effectively contribute to a better long-term effect of mined rock phosphate and processed P-fertilisers compared to thermal oxidation materials & derivates rich in trivalent cations.

Studies that supply primary and secondary macronutrients together with mined rock phosphate and processed P-fertilisers to ensure an equal supply of all different plant nutrients across treatments show a somewhat reduced relative agronomic efficiency, especially when PUE is considered as a response variable. On the other hand, results for the field studies performed in more realistic settings than those of pot experiments show better results. Both effects are potentially related, as field studies often apply a deficient experimental design where the broad range of secondary macronutrients and micronutrients present in thermal oxidation materials & derivates are not added in the mined and synthetic P-fertiliser treatment. Hence, **these results indicate the importance of secondary macronutrients and micronutrients in achieving optimal agricultural yields**. It is often challenging to evaluate the supplementary fertiliser need for particular plant-limiting elements within the broad spectrum of secondary macronutrients and micronutrients. On condition that the excess application of micronutrients is avoided, the application of thermal oxidation materials & derivates as P-fertilisers could provide the complementary benefit of supplying secondary macronutrients and micronutrients to enhance agronomic yields.



Subgroup	No. of cases (%)	No. of cases (%)
Overall	117 (100)	94 (100)
pH		
acidic	67 (57)	51 (54)
neutral or basic	50 (43)	43 (46)
Texture		
coarse	99 (85)	72 (77)
medium	13 (11)	22 (23)
fine	2 (2)	0 (0)
Feedstock		
sewage sludge	65 (56)	54 (57)
crop residue	20 (17)	25 (27)
poultry litter	14 (12)	4 (4)
pig manure	3 (3)	1 (1)
slaughter by-products	4 (3)	4 (4)
wood	2 (2)	2 (2)
steel slag	4 (3)	0 (0)
mix	5 (4)	4 (4)
Post-processing		
wet-digestion	8 (7)	2 (2)
thermal	55 (47)	48 (51)
none (sewage sludge)	13 (11)	10 (11)
none (other)	41 (35)	34 (36)
Application form		
powder	86 (74)	59 (63)
granules	29 (25)	34 (36)
Plant type		
cereal	69 (59)	61 (65)
grass	17 (15)	14 (15)
oilseed	8 (7)	9 (10)
legume	6 (5)	1 (1)
other	17 (15)	9 (10)
Soil P status		
P-poor	41 (35)	19 (20)
P-rich	74 (63)	75 (80)
Assessment time		
short	73 (62)	68 (72)
long	44 (38)	26 (28)
Experimental design		
fully balanced	75 (64)	59 (63)
deficient	42 (36)	35 (37)
Experimental setting		
field trial	17 (15)	12 (13)
pot trial	100 (85)	82 (87)

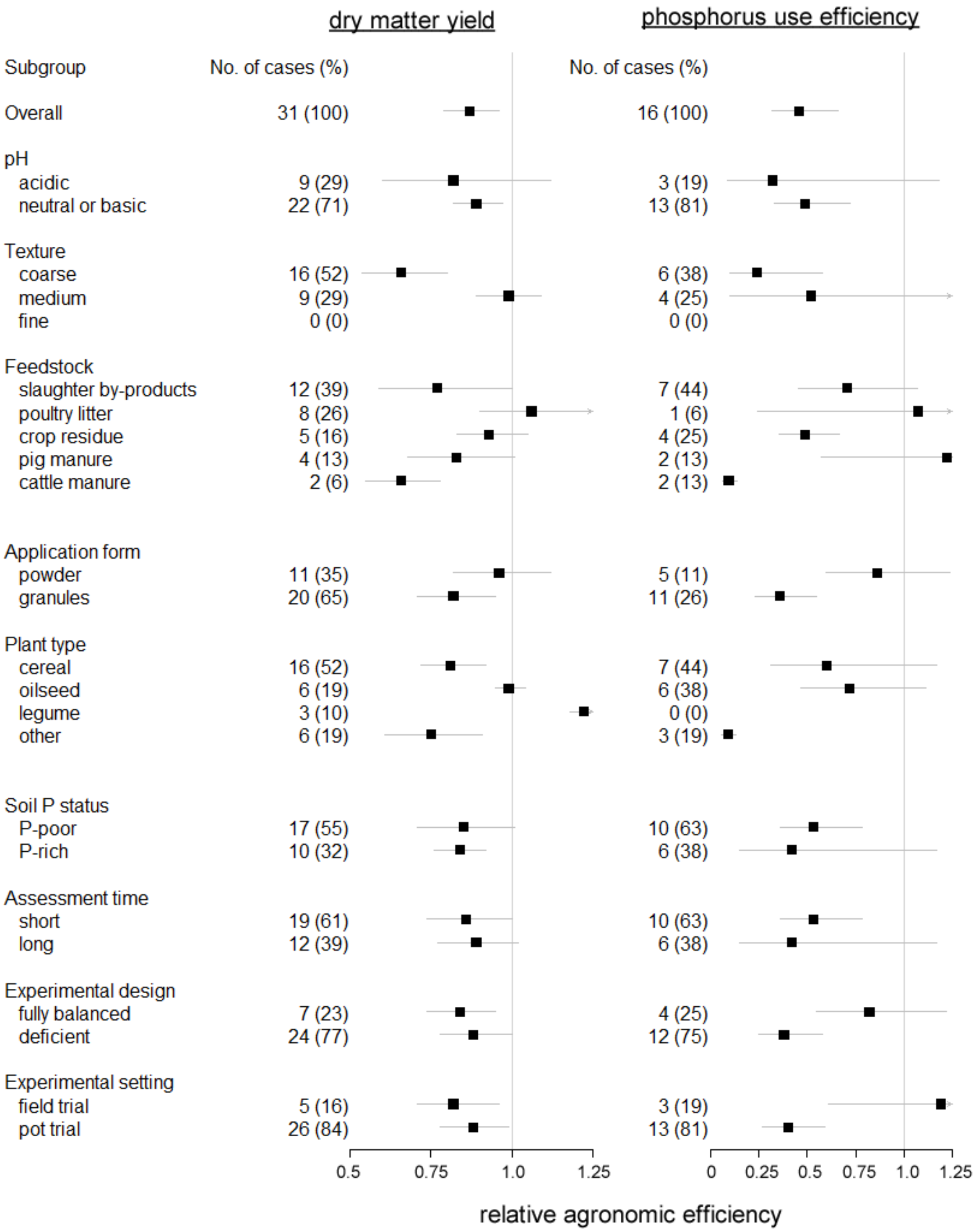
NB: Results are presented as weighted mean (square) and 95% confidence intervals (error bars).

Figure 6: The relative agronomic efficiency of thermal oxidation materials & derivates for the plant response variables DMY (dry matter yield) and PUE (phosphorus use efficiency) as a function of grouping variables

6.2.4 Pyrolysis & gasification materials

The overall mean effects for pyrolysis & gasification materials were 0.87 and 0.46 for $\mathrm{RAE}_{\mathrm{DMY}}$ and $\mathrm{RAE}_{\mathrm{PUE}}$, respectively (Figure 7). The analyses for each of the grouping variables were constrained by the number of studies available; only the RAE values for neutral and basic soils and for pyrolysis & gasification materials that were applied in granulated form were derived from a minimum of 4 different studies and a number of cases greater than 10 for both response variables (Figure 7). For these groups, the $\mathrm{RAE}_{\mathrm{DMY}}$ and $\mathrm{RAE}_{\mathrm{PUE}}$ values pointed towards a significantly lower agronomic efficiency than for mined rock phosphate and processed P-fertilisers. The significant differences of specific groups such as those varying in soil texture ($\mathrm{RAE}_{\mathrm{DMY}}$), feedstock ($\mathrm{RAE}_{\mathrm{PUE}}$), application form ($\mathrm{RAE}_{\mathrm{PUE}}$), plant type ($\mathrm{RAE}_{\mathrm{DMY}}$ and $\mathrm{RAE}_{\mathrm{PUE}}$), experimental design and setting ($\mathrm{RAE}_{\mathrm{PUE}}$) should be interpreted with caution because some of the contrasting groups have a low number of cases, often originating from a few studies. Therefore, only a marginal reduction of the size of the confidence interval of the underlying ‘true effect’ across groups could be achieved, compared to the results from individual studies by applying the meta-analysis techniques. Hence, no conclusions can be drawn on RAE across pyrolysis & gasification materials applied to different soil types, feedstocks, application forms and plant types. Figure 7 enables, nevertheless, a standardised visual assessment of the RAE ranges observed across selected studies.

Given the small sample size for pyrolysis & gasification materials, it is not pertinent to draw overarching conclusions for pyrolysis & gasification materials from the available data. The properties of pyrolysis & gasification materials can vary widely, depending on the interactive effects between production process conditions and feedstock applied. Many groups, including pyrolysis & gasification materials derived from slaughter by-products, poultry litter, crop residues and pig manure, display an agronomic efficiency that is not significantly different from F_{prim}. The sole groups for which a relatively large number of data are available (at least 4 different studies and > 10 cases) are neutral and basic soils and pyrolysis & gasification materials that have been applied in granulated form. For these groups, a lower agronomic efficiency than for mined rock phosphate and processed P-fertilisers is observed. Potentially, some of the documented high agronomic efficiencies after the application of pyrolysis & gasification materials addition could be the result of a liming effect that increases soil P availability (Hass et al., 2012), or the result of the milling of the pyrolysis material which increases the P solubility in the otherwise stable pyrolysis matrix (Ma and Matsunaka, 2013). Therefore, future studies should focus on assessing the mechanisms that underlie documented potential positive plant responses, and evaluate the agronomic efficiency of pyrolysis & gasification materials in the same physical form as will be applied under actual settings in agriculture. It is concluded that the current available data do not enable a comprehensive assessment of the agricultural efficiency of P-rich pyrolysis & gasification materials in relevant European agricultural settings, and that plant responses for P-rich pyrolysis & gasification materials can vary widely depending on the feedstock and production conditions of the pyrolysis & gasification materials, as well as on the soil and plant type under fertilisation.



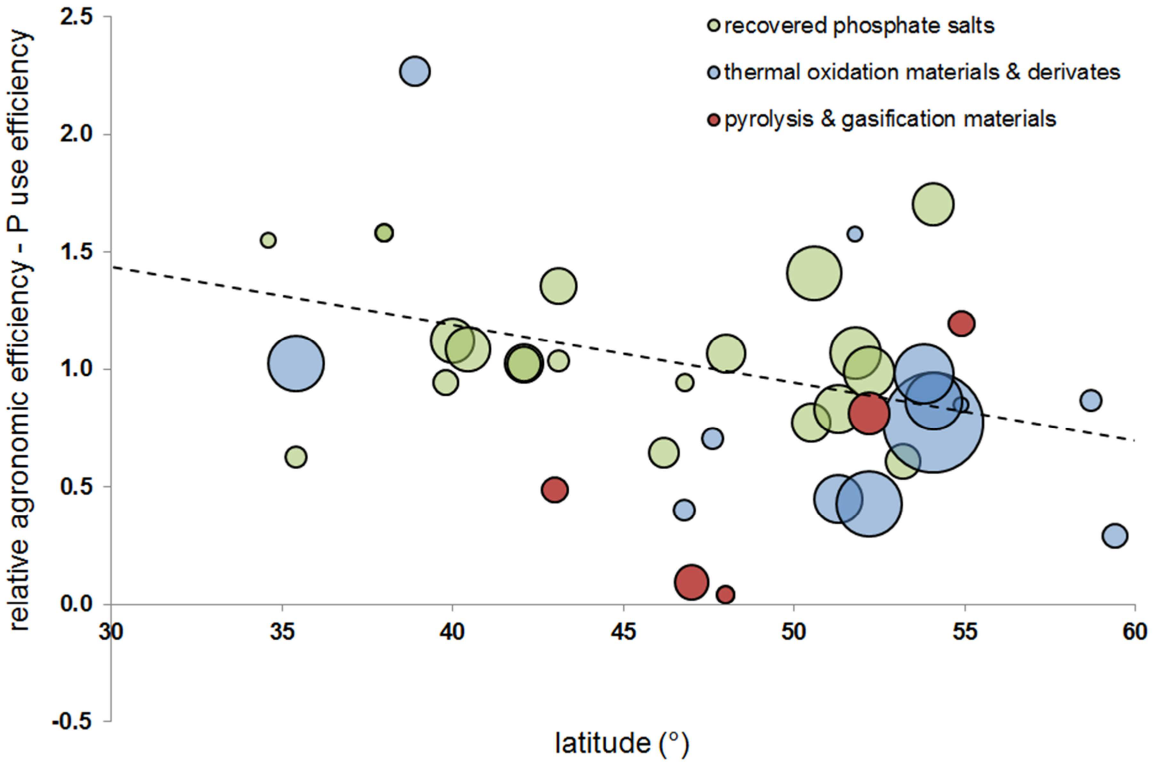
NB: Results are presented as weighted mean (square) and 95% confidence intervals (error bars).

Figure 7: The relative agronomic efficiency of pyrolysis & gasification materials for the plant response variables DMY (dry matter yield) and PUE (phosphorus use efficiency) as a function of grouping variables

6.2.5 Geographic scattering

The results provided give an overview of the relative agronomic efficiency as a function of soil and plant type, but fail to take into consideration the interactions and combinations of those variables that occur in different geographic regions in Europe. The effect of the north-south position (i.e. latitude of the geographic coordinates) is especially relevant, given that climatic conditions (colder and drier soils at higher latitudes), soil texture (sandier at higher latitudes), and soil pH (more basic at lower latitudes) vary significantly across this gradient (Panagos et al., 2012; Ballabio et al., 2016). **A significant negative correlation between geographic latitude on RAE_{PUE} was indicated (*P*: 0.02), with greater RAE_{PUE} values observed in sites of lower latitudes than in higher latitudes** (Figure 8). Latitude explained, nonetheless, only a minor share of the total variance observed (R^2_{adj}: 0.14). It should, however, be noted that the assessment includes both pot and field studies, and that some variables, especially climatic conditions, may not be accurately represented in pot experiments. Therefore, the results should be interpreted with the necessary caution.

Concerns related to the effectiveness of water-insoluble P-fertilisers in semi-arid and Mediterranean regions may exist because some slow-release P-fertilisers, such as phosphate rock and meat and bone meal, do not dissolve readily in such soils (Bolland and Gilkes, 1990; Elliott et al., 2007). The results of our work, however, reject such expectations for P-fertilisers containing STRUBIAS materials in European settings as the RAE_{PUE} correlated negatively to latitude (Figure 8). Therefore, **the effectiveness of F_{sec} for semi-arid and Mediterranean European regions is suggested.** The soil moisture patterns probably have a negligible impact on the solubility of P-fertilisers containing STRUBIAS materials, as these have a low water-soluble P fraction. Their solubility is mainly determined by the extent of the root exudation of the plants grown on the agricultural field. It can, however, be expected that the solubility of water-soluble P-fertilisers is increased in the more northern latitudes characterised by more moist soils due to the increased precipitation. Therefore, the agronomic efficiency of mined rock phosphate and processed P-fertilisers could be higher for the higher latitudes, resulting in decreased RAE ratios in the more northern regions. Other soil properties that vary across latitude, such as soil texture and soil pH, did not have a significant effect on the RAE for the STRUBIAS materials under study.



NB: The size of the bubbles represents the number of cases and relative weight for each data pair The assessment includes both pot and field studies, and that some variables, especially climatic conditions, may not be accurately represented in pot experiments.

Figure 8: Bubble plot indicating the relationship between $\mathrm{RAE}_{\mathrm{PUE}}$ and latitude

6.3 C-rich pyrolysis & gasification materials

The addition of pyrolysis & gasification materials to soils induces a series of changes in the plant mycorrhizosphere that can promote (e.g. augmented soil fertility, increased soil water-holding capacity, physical protection for plant-growth-promoting microorganisms) or reduce (e.g. addition of phytotoxic volatile organic compounds, nutrient immobilisation, water retention) plant growth and nutrient uptake. The interaction between positive and negative effects determines the effect on plant yield and plant nutrient uptake, and thus the possible benefit to the farmer. Meta-analysis results have summarised the effect of the application of pyrolysis materials relative to control soils that were unamended. A recent meta-analysis for C-rich pyrolysis & gasification materials indicated that, on average, C-rich pyrolysis & gasification materials did not increase plant yields relative to unfertilised control sites (Jeffery et al., 2017). The study was based on 598 cases documented in 44 different studies from temperate regions. It was observed that **C-rich biochar amendment to soils in temperate regions significantly decreased crop yield relative to *controls* (i.e. soils that did not receive any fertilising materials)**, averaging approximately 3% at a median biochar application rate of 30 t ha^{-1}. For temperate regions, only positive effects for pyrolysis & gasification materials were observed at an application rate of 31-50 tonnes ha^{-1}. Lower application rates (312 pairwise comparisons; 52% of all data) did not show any significant differences relative to control soils, whereas higher application rates (224 pairwise comparisons, 37% of the data) at times even showed negative impacts on plant yields. These results of Jeffery et al. (2017) are in line with the results found for temperate regions by Biederman and Harpole (2013). Jeffery et al. (2017) indicated that many arable soils in temperate regions are moderate in pH, high in fertility, and generally receive high fertiliser inputs, leaving little room for additional benefits from C-rich pyrolysis & gasification materials. Their work indicated the positive impacts of the application of pyrolysis & gasification materials for soils with an organic C level < 1%. Although such soils are mostly absent in Europe (de Brogniez et al., 2015), this observation may point towards the possible added value of pyrolysis & gasification materials in soil-less growing media.

7 Market aspects and outlook for the year 2030

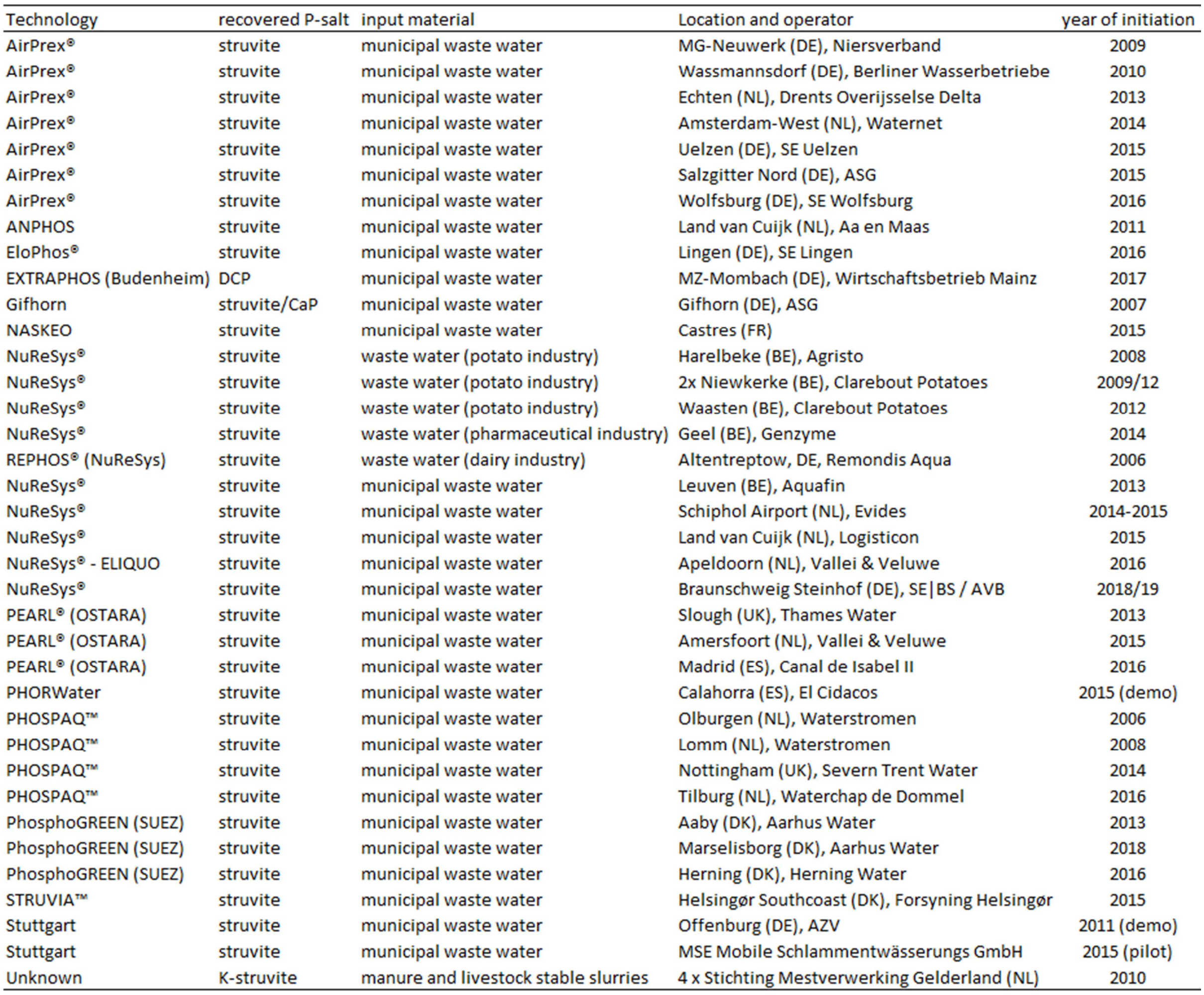
This section gives an overview of the current (Section 7.1) and projected future market (Section 7.3) for STRUBIAS materials, as well as for mined phosphate rock and processed P-fertilisers (Section 7.2). In the EU Fertilising Products Regulation ((EU) 2019/1009), fertilising products are classified in product function categories (PFCs) that are in line with their intended function. Given the different intended uses for these STRUBIAS materials, this document is structured into different **subsections that cover uses of such materials in P-fertilisers (Sections 7.2 and 7.3)** or in **liming materials, soil improvers, growing media and plant biostimulants (Section 7.4)**. Sections 7.2 and 7.3 on P-fertilisers cover materials with a minimum P-content of 3% and 2% P_2O_5, the P-threshold values for inorganic and organic/organo-mineral P-fertilisers, respectively. Market aspects for materials with a lower P-content are covered in Section 7.4.

7.1 Current STRUBIAS market

7.1.1 Precipitated phosphate salts

Currently, best estimates summing production volumes of the different plants suggest that **about 15 000 tonnes of struvite** are produced each year in Europe. Existing facilities mainly use municipal wastewaters as input material, although industrial wastewaters (potato industry, pharmaceutical industry, dairy industry) and manure and livestock stable slurries are also used as input materials (Kabbe et al., 2017; Ehlert et al., 2016a). Additionally, substantial amounts of struvite are produced outside Europe (USA, Japan, China) (Kabbe, 2017).

Table 9: Overview of facilities that produce precipitated phosphate salts in the European Union (data adopted from Kabbe, 2017 and Ehlert et al., 2016a)



Technology	recovered P-salt	input material	Location and operator	year of initiation
AirPrex®	struvite	municipal waste water	MG-Neuwerk (DE), Niersverband	2009
AirPrex®	struvite	municipal waste water	Wassmannsdorf (DE), Berliner Wasserbetriebe	2010
AirPrex®	struvite	municipal waste water	Echten (NL), Drents Overijsselse Delta	2013
AirPrex®	struvite	municipal waste water	Amsterdam-West (NL), Waternet	2014
AirPrex®	struvite	municipal waste water	Uelzen (DE), SE Uelzen	2015
AirPrex®	struvite	municipal waste water	Salzgitter Nord (DE), ASG	2015
AirPrex®	struvite	municipal waste water	Wolfsburg (DE), SE Wolfsburg	2016
ANPHOS	struvite	municipal waste water	Land van Cuijk (NL), Aa en Maas	2011
EloPhos®	struvite	municipal waste water	Lingen (DE), SE Lingen	2016
EXTRAPHOS (Budenheim)	DCP	municipal waste water	MZ-Mombach (DE), Wirtschaftsbetrieb Mainz	2017
Gifhorn	struvite/CaP	municipal waste water	Gifhorn (DE), ASG	2007
NASKEO	struvite	municipal waste water	Castres (FR)	2015
NuReSys®	struvite	waste water (potato industry)	Harelbeke (BE), Agristo	2008
NuReSys®	struvite	waste water (potato industry)	2x Niewkerke (BE), Clarebout Potatoes	2009/12
NuReSys®	struvite	waste water (potato industry)	Waasten (BE), Clarebout Potatoes	2012
NuReSys®	struvite	waste water (pharmaceutical industry)	Geel (BE), Genzyme	2014
REPHOS® (NuReSys)	struvite	waste water (dairy industry)	Altentreptow, DE, Remondis Aqua	2006
NuReSys®	struvite	municipal waste water	Leuven (BE), Aquafin	2013
NuReSys®	struvite	municipal waste water	Schiphol Airport (NL), Evides	2014-2015
NuReSys®	struvite	municipal waste water	Land van Cuijk (NL), Logisticon	2015
NuReSys® - ELIQUO	struvite	municipal waste water	Apeldoorn (NL), Vallei & Veluwe	2016
NuReSys®	struvite	municipal waste water	Braunschweig Steinhof (DE), SE|BS / AVB	2018/19
PEARL® (OSTARA)	struvite	municipal waste water	Slough (UK), Thames Water	2013
PEARL® (OSTARA)	struvite	municipal waste water	Amersfoort (NL), Vallei & Veluwe	2015
PEARL® (OSTARA)	struvite	municipal waste water	Madrid (ES), Canal de Isabel II	2016
PHORWater	struvite	municipal waste water	Calahorra (ES), El Cidacos	2015 (demo)
PHOSPAQ™	struvite	municipal waste water	Olburgen (NL), Waterstromen	2006
PHOSPAQ™	struvite	municipal waste water	Lomm (NL), Waterstromen	2008
PHOSPAQ™	struvite	municipal waste water	Nottingham (UK), Severn Trent Water	2014
PHOSPAQ™	struvite	municipal waste water	Tilburg (NL), Waterchap de Dommel	2016
PhosphoGREEN (SUEZ)	struvite	municipal waste water	Aaby (DK), Aarhus Water	2013
PhosphoGREEN (SUEZ)	struvite	municipal waste water	Marselisborg (DK), Aarhus Water	2018
PhosphoGREEN (SUEZ)	struvite	municipal waste water	Herning (DK), Herning Water	2016
STRUVIA™	struvite	municipal waste water	Helsingør Southcoast (DK), Forsyning Helsingør	2015
Stuttgart	struvite	municipal waste water	Offenburg (DE), AZV	2011 (demo)
Stuttgart	struvite	municipal waste water	MSE Mobile Schlammentwässerungs GmbH	2015 (pilot)
Unknown	K-struvite	manure and livestock stable slurries	4 x Stichting Mestverwerking Gelderland (NL)	2010

The current market for P-salt recovery materials is mainly driven by the increased need to remove P from waste streams (e.g. urban wastewaters, manure, waste from the food-processing industry) to reduce and prevent the leaching of P to water bodies. Given the national and EU legislation and guidance on nutrient management and water quality (Common Agricultural Policy, Water Framework Directive, Nitrates Directive, etc.), tertiary treatment with enhanced P removal is becoming a more common practice in many European municipal and industrial wastewater treatment facilities (European Environment Agency, 2013a). Basically, there are two options to prevent P from ending up in the effluents of wastewater treatment plants: (1) **enhanced biological phosphorus removal (EBPR)**, and (2) **chemical precipitation with metal salts (ChemP)** or a combination of both. In EBPR, microorganisms (P accumulating organisms) 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 aluminium (Al(III)), ferric (Fe(III)) and ferrous (Fe(II)) oxides, and calcium (Ca(II)) salts. Phosphorus nutrient removal initially relied entirely on chemical precipitation, which remains a leading technology today (Wilfert et al., 2015). Nonetheless, EBPR has become firmly established in some European Member States (Wilfert et al., 2015).

Struvite production provides important operational **benefits for the operation of municipal wastewater treatment plants** that apply enhanced biological phosphorus removal, even without retailing struvite as a fertiliser.

- Costs for biological wastewater treatment plants are reduced by the lower maintenance costs due to the **avoided pipe clogging and abrasion of centrifuges**.
- Struvite production processes that precipitate **P from (activated) digested sludges increase the dewaterability** of the sludge, in turn lowering the associated costs for dewatering chemicals (e.g. flocculation agents) and sludge disposal. At present, operating costs for sludge dewatering usually account for up to 25-50% of the total expenses of the entire wastewater treatment process (Mahmoud et al., 2011). 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 cations create bridges between particles whereas monovalent cations tend to deteriorate floc 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 the proper tuning of chemical additions in order to achieve progressive dewatering.
- The **reduction of the N load of the sludge liquor** has a direct effect on the overall treatment capacity of the wastewater treatment plant as well as on its operational costs, since the removal of N from wastewater requires energy, chemicals and tank volume (Ewert et al., 2014).

In most EU Member States, **struvite is not yet legally recognised as a fertiliser,** meaning a special permission from the national government is needed to be relieved of the waste status. This situation may cause a bottleneck in the distribution of the struvite produced as fertiliser to agriculture. Of the full-scale techniques mentioned, only the struvite products Pearl and NuReSys (respectively Crystal Green and BioSTRU) are certified as fertilisers in the United States/United Kingdom and Belgium, respectively. The struvite obtained by the Seaborne process is only used locally. It can be concluded that the outputs produced are mostly used in the countries where production takes place and that **in most cases the existing market and production volumes are very small**.

The installation of P-precipitation recovery processes is a service that is typically carried out by **industrial partners** of the municipalities that operate the plant. The improved operability and the reduced maintenance costs associated with controlled struvite precipitation and removal enable municipalities to justify expenditure for the installation of the recovery facility. The industrial partner can be in charge of the sales of the recovered P-precipitate, or

P-precipitates can be sold directly by the wastewater treatment operators to the fertiliser industry (for further processing) and to farmers (for direct use on the field).

7.1.2 Thermal oxidation materials & derivates

Significant amounts of ashes are produced as **production residues from the biomass energy and paper industry**. In addition, the incineration of **poultry litter** and **meat and bone meal** is an established practice that combines the purposes of energy generation and nutrient recovery. The ashes of those combustion and incineration facilities can be applied as fertilising materials directly on land (**raw ashes, or 'thermal oxidation materials'**), without post-treatment. A second group of ash-based materials (**'thermal oxidation material derivates'**) are P-concentrated fertilisers that have been derived from manufacturing processes on ashes obtained from the combustion/incineration of P-rich input materials with the specific intention to produce P-fertilisers.

7.1.2.1 Ashes and slags

Biomass ashes from the wood and paper industry

The demand for biomass-based heat and electricity is increasing because of targets for generating **energy from renewables** and decreasing the emission of fossil CO_2. **Thus, there is increased interest in biomass ash utilisation.** Also, for the waste generated by the wood pulp and paper industry, incineration with energy recovery is becoming the main waste recovery method because landfills are increasingly being reduced as a final destination for wastes in Europe (Monte et al., 2009). Data on the exact amount of ashes produced are limited; according to the report of the International Energy Agency (van Eijk et al., 2012), about 600 kt of ashes per year are produced from clean wood summing the contributions in Austria, Denmark, Finland, Germany, the Netherlands and Sweden alone. Additionally, substantial amounts of ashes are produced from waste wood (e.g. 270 kt yr^{-1} in Germany) and black liquor (i.e. the waste materials from the kraft process when digesting pulpwood into paper pulp; e.g. 135 kt yr^{-1} in Austria). Hence, the volumes of ash produced are substantial.

Nevertheless, direct use of ashes as fertiliser on agricultural or forest soils is primarily possible for bottom ashes or mixtures of bottom and coarse fly ashes that have lower amounts of contaminants, and only when clean biomass fuels are used. Moreover, it should be considered that plant-based ashes have a low P-content (see Sections 5.4.4.1 and 15.2.1; on average about 0.7% P for bottom ashes), making the potential for P-recovery from such materials intrinsically low. Based on the data by Van Dijk et al. (2016), the combined P losses from the wood and paper industry are about 79 kt P yr^{-1}. Nonetheless, considering the contamination of a substantial fraction of ashes by chemicals (paper industry, waste wood from households, etc.), the existing alternative uses of ashes (e.g. cement industry), and the quality requirements for their use as a fertilising product, **only a relatively small contribution is expected for ash materials from the wood and paper industry for P-recycling in Europe**. These ashes may, however, also contribute to the recycling of other nutrients, such as Ca and K.

Poultry litter and meat and bone meal ashes

About 80% of the **non-edible animal by-products from abattoirs** are processed to meal (bone meal, meat meal, feather meal, blood meal, carcass meal and combinations thereof). Animal meal production is a process that includes bulk slaughterhouse waste mincing and coagulation, followed by the separation of the solid and liquid material by pressing. The solid fraction is then dried, while the wet fraction is heated for the extraction of fats. For processed meat and bone meals (MBM; ~ 5% P), pet food and combustion with energy recovery are the most common fates, and only a small share of the available and sterilised meals are used for direct use as a fertiliser, often in organic farming (Franke-Whittle and Insam, 2013; Moller, 2015). Especially in the UK, MBM are increasingly being processed to fertilisers with a high P-content (6-19%) (ESPP, 2016). EPR (UK) produces more than 2.8 kt P yr^{-1} of their 'P-grow' MBM fertiliser, while Saria (UK) processes around 45 kt of MBM to produce ~ 2 kt P yr^{-1} MBM as the P-fertiliser product 'Kalfos' (mainly calcium phosphate mineral fertiliser, ~ 21% P_2O_5 plus potassium and sulphur). Also, companies like Fibrophos (UK), ACL/Wykes Engineering (UK), COOPERL (FR), Elosato (FI) and ITS SA (PT) process inedible animal by-products and meat and bone meal to straight P-fertilisers or compound PK fertilisers (ESPP, 2016). Van Dijk et al. (2016) estimated the total P-recovery through the production of fertilisers from slaughterhouse waste at 16 kt P yr^{-1}.

The incineration of **poultry litter** with energy recovery is currently being performed by commercial companies such as BMC Moerdijk (NL), Fibrophos (UK), BHSL (IE) and others. Those companies alone process > 1500 kt of poultry litter yearly, leading to an estimated recovery of about 30 kt P yr^{-1} (and similar quantities of K). The poultry litter ash end material has a P content of about 7.6% (median value; Ehlert, 2017).

7.1.2.2 Thermal oxidation material derivates

Ashes can only be applied on land as fertilisers when derived from input materials with a low content of inorganic metals and metalloids. **The post-combustion techniques can remove the inorganic contaminants present in ashes, enabling the use of more contaminated input materials, and simultaneously increase the plant-availability of the nutrients in the ashes**.

Both thermochemical and wet-digestion techniques are applied in piloting and operational facilities in Europe (see Section 14.2). The most suitable input materials for these processes are ashes that have been produced from **P-rich input materials** (e.g. mono-incinerated sewage sludge ashes from EBPR and Chem-P plants, animal bones, meat and bone meal, possibly poultry litter). These facilities have recently been established in Europe, and some operators hope to replace a large share of their phosphate rock with secondary phosphates in the near future (see Section 14.2.2 for a non-exhaustive list of such processes).

7.1.3 Pyrolysis & gasification materials

The **International Biochar Initiative (IBI),** a trade and advocacy group for the nascent industry that focuses exclusively on for-profit pyrolysis production enterprises, made a conservative estimation of 827 tonnes of pyrolysis & gasification materials produced worldwide in 2013, by a total of 175 companies. The 2015 IBI report highlights that the number of active pyrolysis companies rose from **200 in 2014 to 326 companies in 2015**. The steady increase is most likely indicative of both new companies entering the marketplace and more information being readily available regarding pyrolysis companies around the world (International Biochar Initiative, 2016b).

According to IBI, the pyrolysis & gasification materials industry is in a fledgling state, comprised largely of enterprises selling relatively **small volumes** of pyrolysis & gasification materials, with a limited package size, **locally for end uses such as gardening and tree care.** Pyrolysis has yet to make a substantial entry into large-scale agricultural operations (International Biochar Initiative, 2016b).

An **overall assessment of the specific situation for Europe is not available**. Nevertheless, based on the information available for specific pyrolysis facilities and retailers in the EU, it is concluded that the current market is relatively small (actual production volumes < 10 000 t material yr^{-1}). It should be noted that most manufacturers and producers focus on the production of plant-based pyrolysis & gasification materials with a low P-content, which explains why the current contribution of pyrolysis to the market of P-fertilisers derived from secondary raw materials is low:

- The **developed 3R technology** integrates pyrolysis, catalytic and biotechnological processes to produce plant-based and animal bone pyrolysis materials (3R AgroCarbon, 2016). The technology is owned by the company **Terra Humana Ltd.**, with a staff of 12 people, and is the only medium pyrolysis facility that produces materials intended for agricultural use with a > 1 000 t yr^{-1} throughput capacity. Recently the company also received authority permits for the full-scale industrial installation and operation of a pyrolysis plant in Kajászó, Hungary. For 2018/2019, a **production (output) capacity of 12 500 t material yr^{-1} is targeted.** The current state of technology readiness level is high (TRL 8-9).
- The German company **Pyreg (PYREG, 2016)** sells pyrolysis plants, but does not operate any plant.
- **Carbon Terra** has a production capacity of about 1 000 t yr^{-1} and relies on the Schottdorf Technology (under patent) and is also based in Germany. The input materials are not specified, but it is stated the company only relies on surplus biomass, and that the technology can process over 100 different kinds of biomass. The process is certified according to the EBC, and the quality management of Carbon Terra is based on the DIN ISO 9001 *Standard*. The pricing ranges from EUR 25 for a 30 L package to EUR 900 for 1 400 L.

- The German company **Regenis GmbH** has a pyrolysis plant with an annual production capacity of 500 tonnes, but no further information is currently available on pricing (Regenis - Bio Energie Technologie, 2016).
- **Biomacon GmbH (Germany) and Black Carbon (Denmark)** are producers of pyrolysis plants. Biomacon produces machineries with production capacities ranging from 6.2 kg to 34.2 kg hour^{-1} (540-3 000 t yr^{-1}), while an annual production capacity of 300 tonnes is planned for Black Carbon (BIOMACON, 2016; Black Carbon, 2016).
- Moreover, there are a number of companies based in the EU that produce or sell small volumes of pyrolysis & gasification materials: Biogreen/EDT (FR), EM-Chiemgau (Germany), Sonnenerde (Austria), AWN Abfallwirtschaftsgesellschaft des Neckar-Odenwald-Kreises mbH (Germany), Geiger Pflanzenkohle und Energie UG (Germany), FETZER Rohstoffe + Recycling GmbH (Germany), Lixhe Compost SA (Belgium) and Carmagnola Energie SRL (Italy).

7.2 Market aspects and outlook for P-fertilisers

7.2.1 Phosphate rock as the major source for P-fertilisers

The current market for P-fertilisers other than manure is **dominated by processed mineral materials that have been derived from phosphate rock**. Specific materials such as unprocessed phosphate rock, meat and bone meal, and guano can be used as alternative P-fertilisers, but their use is limited in terms of the quantities applied and agricultural sectors covered. The future outlook of P-fertilisers is based on the evolution of supply and demand of mined fertilising products that currently dominate the P-fertilisers market.

There are three classical routes for the **production process of P-fertiliser from phosphate rock** (European Phosphate Fertilizer Alliance, 2017):

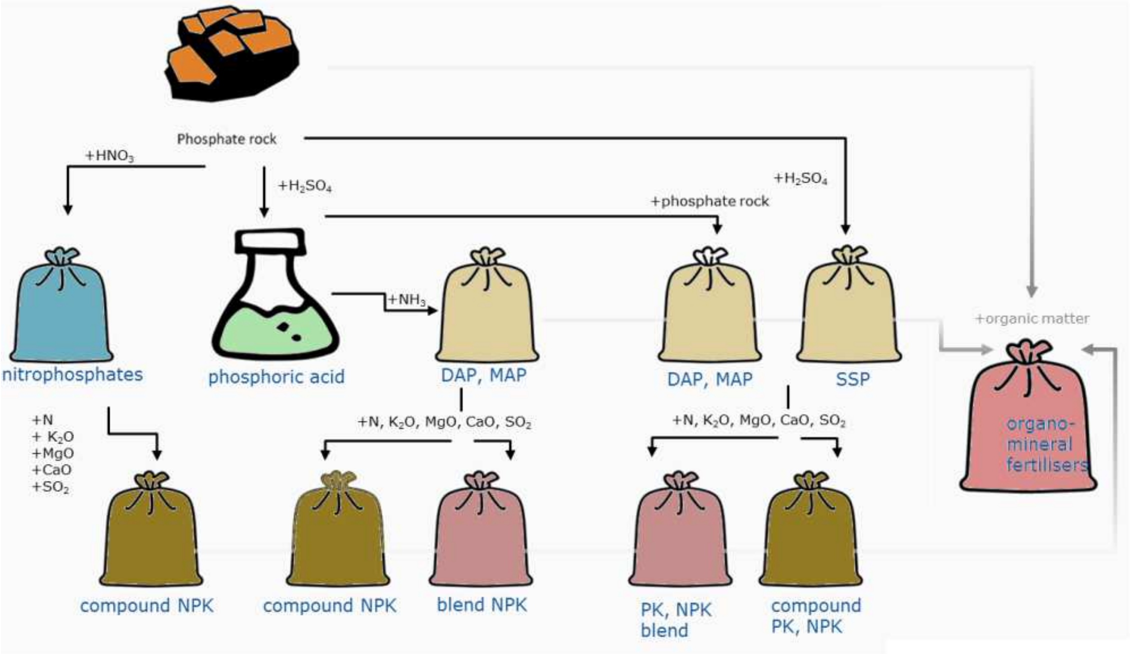
- The first route treats phosphate rock with sulphuric acid to create phosphoric acid, which is then used to produce **diammonium phosphate (DAP) or monoammonium phosphate (MAP)**. DAP is typically 18-46-0 (i.e. contains 18% N, 46% P_2O_5 and 0% K_2O), whereas MAP can be between 10-50-0 and 11-55-0. MAP and DAP can be used as the final fertiliser or as input for NPK fertilisers. The main downside of this process is that some phosphogypsum is created as a residue, which raises environmental concerns and also leads to some P being lost. About two thirds of the phosphate fertiliser production is currently performed through this route.
- The second route also uses sulphuric acid, but in smaller quantities in relation to phosphate rock, which results in superphosphate (**SSP**, typically between 0-16-0 and 0-22-0) or triple phosphate (**TSP**, typically ranging between 0-44-0 and 0-48-0). These can be used as final fertilisers, or as inputs for NPK fertilisers.
- The third route is the so-called **nitrophosphate** route, which consists of treating phosphate rock with nitric acid to obtain compound fertilisers. Phosphoric acid and calcium nitrate are formed as intermediary products, which then react. There are generally two variations of this process (the ‘Odda’ process or the mixed-acid process).

The resulting materials from these three routes can then be used for the production of NPK fertilisers:

- **Complex NPK fertilisers** are produced by an intentional chemical reaction in a chemical plant. For example, a NPK fertiliser can be made by reacting nitric acid with phosphate rock to make an ammoniated phosphate, and adding potassium chloride (KCl) and granulating. The resulting product will have the same proportions of N, P and K in each granule. Complex plants are expensive to build, but can produce the lowest-cost NPK products. They are relatively inflexible, being best operated making a small number of large-volume grades.
- **Compound NPK fertilisers** are physical mixes. Ingredients such as ammonium nitrate (AN), monoammonium phosphate (MAP) and KCl can be mixed in the correct proportions for the amounts of N, P and K required. This mix is then ground down to

a fine powder, thoroughly mixed, and usually granulated using for example steam granulation. This ensures that the desired N, P and K proportions are present in each granule, but without the involvement of wet chemistry. Compound plants require moderate investment, and, whilst still benefiting from economies of scale, are generally smaller and more flexible than complex plants, making a wider variety of products.

- **NPK blends** are physical mixes of different fertilisers, such that the proportions of N, P and K correspond to the desired value in each bag, but each pellet will be of AN, or MAP, or KCl for example. Blending plants are the least expensive, requiring only storage, physical mixing and bagging operations, and have the highest flexibility in terms of products offered.



NB: DAP: diammonium phosphate; MAP: monoammonium phosphate; TSP: triple superphosphate; SSP: single superphosphate; NPK: Fertiliser containing nitrogen, phosphorus and potassium)

Figure 9: Production routes for P-fertilisers that are derived from mined phosphate rock (adapted from European Phosphate Fertilizer Alliance (2017))

7.2.2 Market drivers for P-fertiliser demand

The main **long-term macro-economic drivers** for P-fertilisers are **population growth**, changing diets, determining how many people need to be fed, and **per capita income**, determining how much that population has to spend on food and therefore the quantity and quality of food they can afford.

In addition to the macro-economic drivers, both annual and regional demand for P-fertilisers will vary according to the influences of a range of factors, with crop planting as the most important one (in turn influenced by crop prices). The price of fertilisers can also have an influence, as – increasingly - does the weather. Government policy also plays a significant role in the demand for fertiliser, as can local agricultural practices.

Different crops require differing quantities of N, P and K to produce optimum yields. IFA has been collecting and monitoring fertiliser use by crop at the global level for a number of years. In its most recent report (IFA, 2013), a breakdown of P-fertiliser demand by crop was given for Europe (Figure 10). More than 50% of the P fertilisers were applied to arable farmed land planted with cereals in 2010-11.

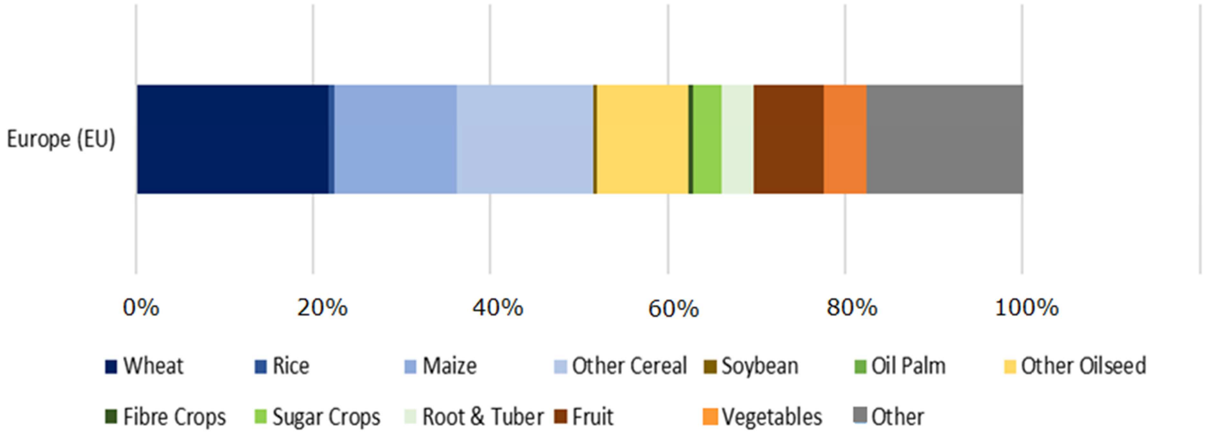


Figure 10: P-fertiliser use by crop (%) in Europe for the year 2010/2011 (Source: Fertilisers Europe & IFA)

The main driver of the reduction in apparent P-fertiliser consumption in Europe has been the significant changes to agricultural policy implemented from 2003. The most relevant of these has been the decoupling of subsidies from production, which was agreed in June 2003, for implementation from May 2005. Prior to 2005, the more farmers produced in volume terms the more subsidies they received. **Since 2005, the subsidy has been as a single farm payment, which is subject to meeting a number of conditions relating to factors such as environmental compliance and quality, food safety and animal welfare**. There is discretion across the EU in terms of implementation but over time the reform, by promoting factors other than production, led production and fertiliser use to decline, or at best stabilise, in the period to 2008. **Since then, as the impact of the revised subsidy faded, production has grown modestly: based on rolling 5-year averages for 2008 and 2015 production has grown at 1.04% per year over the period.**

The impact of **climate change and the weather** will probably play an increasing role in affecting annual volumes. Although the full impacts of climate change on the weather are not yet clear, it does seem that more extreme weather patterns are emerging – bigger storms, more severe droughts and floods etc. (climate variability). The impact of this will be an increased volatility of demand – if there is a significant risk of losing a crop farmers will either not apply P or reduce the amount added to mitigate the risk.

In conclusion, the main long-term global drivers for phosphate demand will be population growth and the continued economic prosperity of each country. At a regional and national level, and on an annual basis, the mix of crop planting, crop

prices, the weather, government policy and fertiliser prices will all influence how demand develops.

7.2.3 Production and consumption volumes

7.2.3.1 Raw materials for P-fertiliser production

The primary raw material for the P-fertiliser industry is **phosphate rock**. Phosphate rocks can be igneous (volcanic – e.g. the rock deposits in the Kola Peninsula in Russia, at Phalaborwa in South Africa, and in a number of locations in Brazil amongst others), but most commonly are sedimentary, being made up from the bones (calcium phosphate) laid down in shallow seas over millions of years. Most sedimentary rocks contain some phosphate, but economically viable extractable deposits of phosphate rock occur where there are one or more seams of rock containing generally more than 15% P_2O_5, which have a uniform texture and composition. Morocco has the largest proven reserves of phosphate, but the International Fertilizer Association (IFA) noted that **commercial production of phosphate rock took place in 29 countries in 2015**. The most important commercial resources are given in Figure 11.

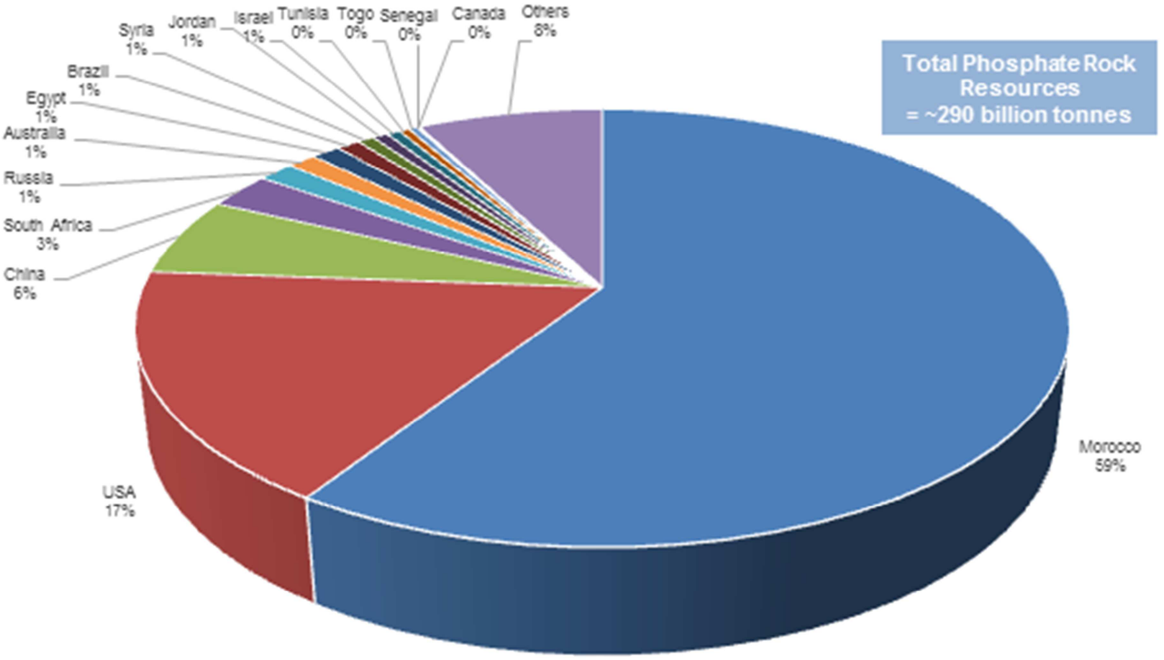


Figure 11: Global phosphate rock sources (Source: IFDC, 2010)

Europe has only one active phosphate rock mine, owned and operated by Yara, and located at Siilinjärvi in Finland (Davidson and Sheldon, 1986). Most of this rock is used by Yara at its manufacturing sites in Finland, or elsewhere in the Nordic region. It therefore follows that most of the rock used in Europe is imported from outside the region. There are other phosphate resources in Europe, including:

- igneous carbonatite outcrops in Sweden, Norway, Scotland and Spain;
- sedimentary deposits in Belgium, France and Germany;
- metamorphic deposit in Serbia.

None of these are currently deemed to be commercially viable. Yara has a project to develop a second resource in Finland at Sokli, north of the Arctic Circle, but the project is currently stalled as there may be alternative long-term sources for the company that are more commercially attractive.

Phosphate rock is thus mined mostly outside the European Union and either imported into the European Union **as rock**, or further **processed abroad and brought into the European Union as a semi-product or product**: phosphoric acid, diammonium phosphate (DAP), monoammonium phosphate (MAP), triple superphosphate (TSP), single superphosphate (SSP). The main phosphate-rock-importing countries in the EU are Lithuania, Poland, Belgium, Bulgaria, Romania and Spain. Most EU imports come from Morocco (1.6 Mt), Russia (1.4 Mt), Algeria (702 kt), Israel (506 kt), South Africa (429 kt) and Tunisia (unknown tonnage).

Imports of rock into Europe declined from around 9.6 million tonnes to 6.0 million tonnes in the 10 years between 2005 and 2015. The phosphate content of the rock imported has remained fairly constant at between 31.9% and 32.9% P_2O_5 (13.9% to 14.3% P). Imports have declined in both the original EU countries (EU-15) and the more recent accession countries (EU-13), but much more so in the former than the latter. The compound average decline in imports between 2005 and 2015 in the EU-15 was 7.9% per year, whereas in the EU-13 it was a much more modest 1.2% per year decline. With only very modest extraction of rock in Europe (0.7 Mt phosphate rock from Finland), apparent consumption follows the same pattern as imports. For the year 2015, Fertecon estimated that **the total apparent P-consumption as phosphate rock in the EU-28 equalled ~ 936 kt P** (6.7 Mt material x 0.32 kg P_2O_5/kg phosphate rock x 0.437 kg P/kg P_2O_5).

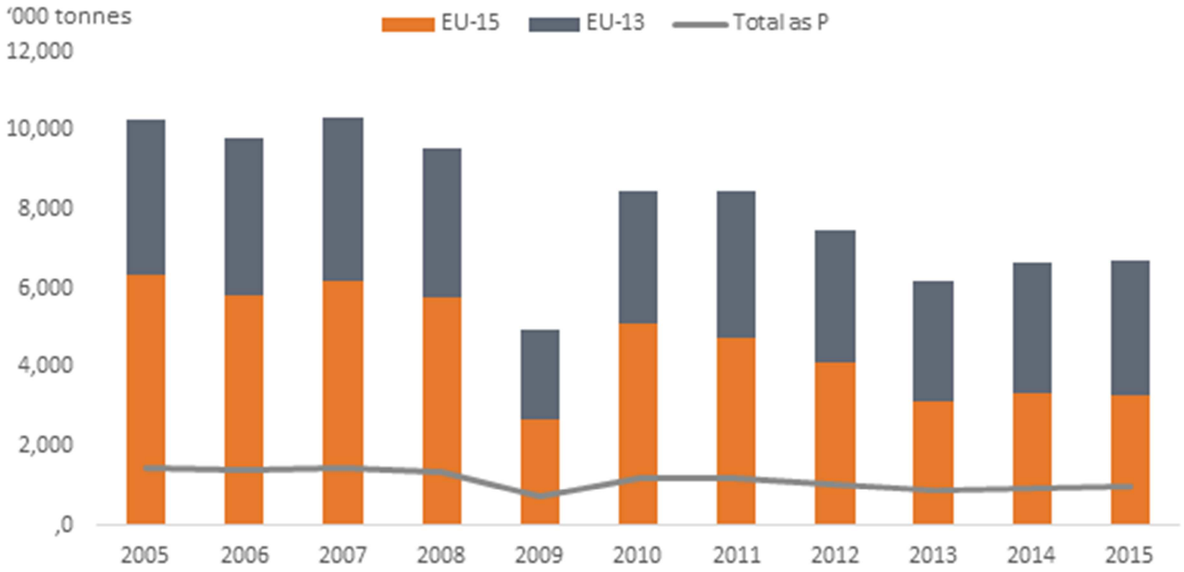


Figure 12: Phosphate rock apparent consumption for all applications in Europe, 2005-2015 (Sources: IFA, GTIS, Fertecon)

Since 2005, apparent phosphate rock **consumption** in the EU-15 countries has declined by 6.3% per year, compared with a more modest 1.4% per year in the EU-13 countries (Figure 12). Since 2010, the apparent consumption for the EU-13, when seasonal fluctuations are removed, seems stable, whereas in the EU-15 countries it has reduced from levels typically above 5.0 million tonnes of product to levels below 3.5 million tonnes of product (Figure 12).

The other main source of phosphate raw material for Europe is **phosphoric acid**. Acid is imported in two main forms, Merchant Grade Acid (MGA, otherwise known as green acid or fertiliser acid), which is an impure acid containing around 54% P_2O_5, and purified phosphoric acid (PPA, otherwise known as white acid), typically containing around 61% P_2O_5. MGA is used as a feedstock for fertiliser and animal feed products; purified acid is used for industrial applications including some speciality fertilisers (frequently, but not exclusively fully soluble products). **For the year 2015, it is estimated that the total P-imports as phosphoric acid in the EU-28 equalled ~ 524 kt P** (Figure 13). The EU-15 accounts for over 95% of all phosphoric acid imports in the EU (Figure 13). This might overstate reality, depending on the accuracy of data on re-exports form key import ports, e.g. in the Netherlands and Belgium, but there is no doubt at all that the EU-15 will account for the majority of the use of imports. Imports to the EU-15 declined by 1.3% per year in the 2005-2015 period, whereas imports grew at 1.8% per year in the EU-13.

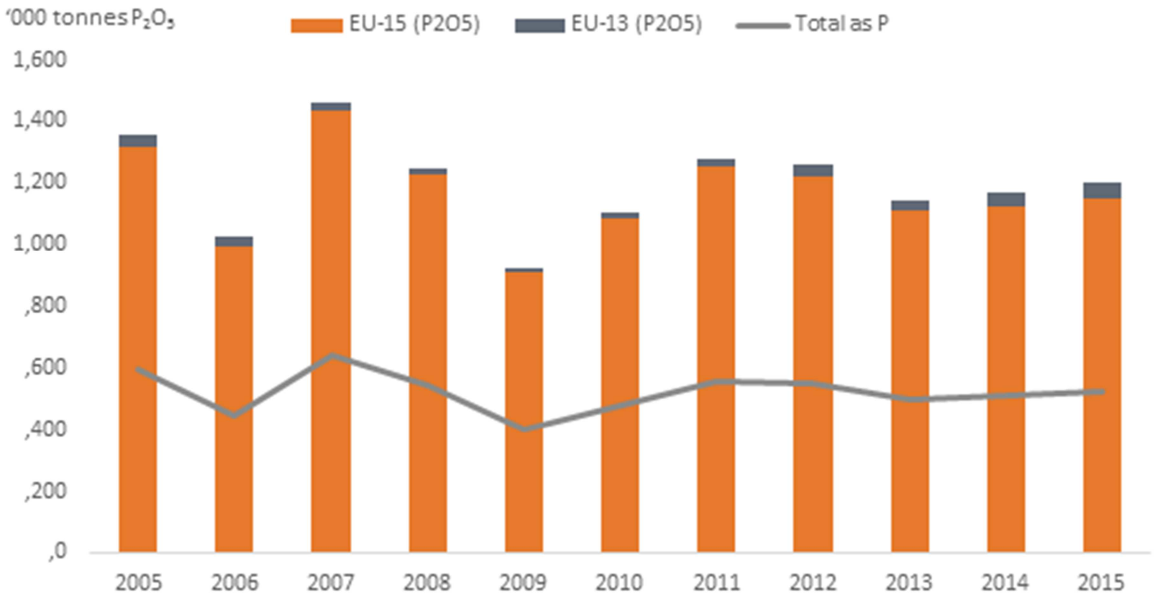


Figure 13: Imports of phosphoric acid into Europe by region (Source: GTIS, IFA)

7.2.3.2 Finished P-fertilisers

Eurostat publishes two data sets on inorganic fertilisers use. The first one is collected from Member States and is an estimate of the N and P use in agriculture. However, data on fertiliser consumption is available in many countries from country-specific data sources (surveys, trade/production statistics) that are not always reliable. **As indicated by Eurostat itself, the quality of the data cannot be sufficiently verified** due to different data sources used (farmer surveys vs trade/production statistics) and inherent problems of data sources used (for instance inclusion of non-agricultural use in statistics based on trade and production). This involves significant limitations as, for instance, the reliability and accuracy of farmer surveys depend amongst others on the sampling design and size.

The other data set is estimated consumption based on the sales of mineral fertiliser in the EU-28 from **Fertilizers Europe**. The figures estimated by the trade association Fertilizers Europe based on sales of mineral fertiliser mostly correspond with the estimates of N and P use reported by countries although they cannot be directly compared due to methodological differences. This is a harmonised data source. Data are available at NUTS0 level for the EU-27 (i.e. does not include Croatia). Data from Fertilizers Europe relate to crop years (t-1/t) which are reported for year t-1. These data have been used in this assessment, also because they are largely agreement with the Member State data documented by Eurostat and the FAO.

Fertilizers Europe estimated the apparent P-consumption for fertilisers in the EU-28 at 1 090 kt P in 2015 (Figure 14). Eurostat estimated the total P-consumption as mineral fertilisers at 1 133 kt P for the year 2015, whereas FAOSTAT documents 1 070 kt P for the EU-28 in 2015. The data from Fertilizers Europe indicate that mineral P-fertiliser consumption was higher in the EU-15 (70%, 765 kt P yr^{-1}) than in the EU-13 (30%, 325 P yr^{-1}) (Figure 14).

Looking forward, Fertecon forecasts phosphate consumption for fertilisers to grow across the EU-28 at a CAGR (Compound Annual Growth Rate) of 0.8% for the next decade. Growth in the EU-15 is stable within any reasonable margin of error (0.09% per year). Fertecon, however, forecasts a growth in the EU-13 countries at a CAGR of 2.2% (Figure 14). Future estimates on mineral P-fertiliser consumption were obtained by projecting these CAPR growth rates, resulting in a 2030 estimate of 1 220 kt P yr^{-1}. The EU-15 still had a dominant share of the total mineral P-consumption (63%; 769 kt P yr^{-1}), but the contribution of the EU-13 increased to a total value of 37% (450 kt P yr^{-1}). Fertilisers Europe forecasts total P-fertiliser consumption at 1 178 kt P yr^{-1}, thus reasonably in line with the Fertecon estimate for the year 2030.

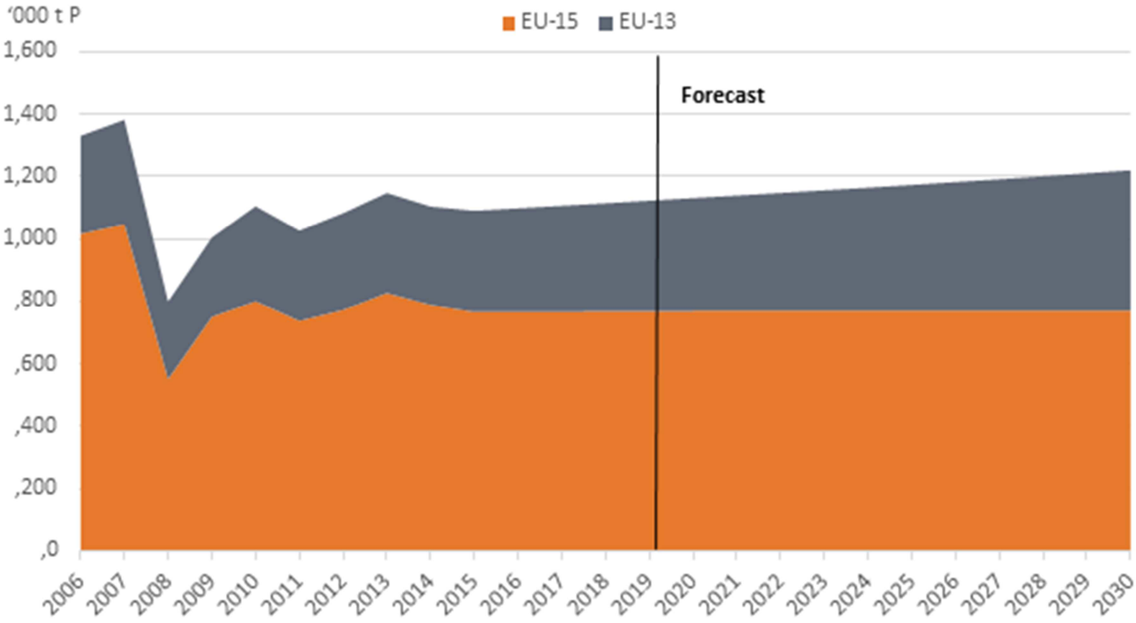
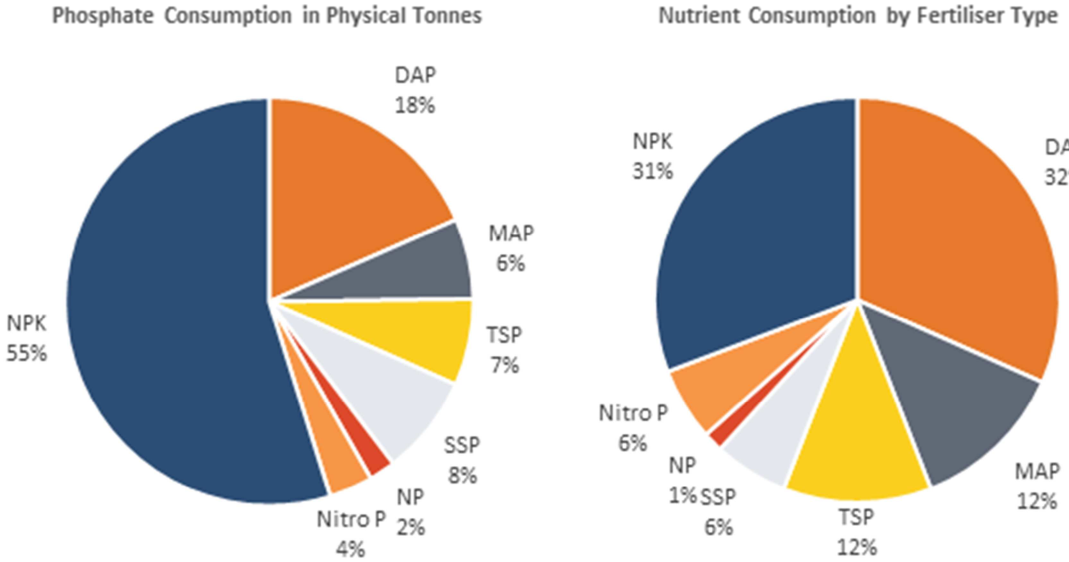


Figure 14: Apparent consumption of mineral P-fertiliser in the EU-28 (kt P yr^{-1}) (Source: Fertilizers Europe; forecasts based on projected Fertecon's Compound Annual Growth Rates)

Although traditionally P-fertiliser and NPK fertiliser producers purchase phosphate rock to produce fertilisers within Europe, in recent years some of the producers have – for numerous reasons (price, environmental or others) – **suspended or abandoned purchases of phosphate rock and have chosen instead to purchase phosphoric acid**. Some of them have even temporarily decided to move even further downstream and purchase MAP or DAP instead (European Phosphate Fertilizer Alliance, 2017).

The apparent consumption of phosphate in the EU-15 both in terms of the physical tonnes of products containing P, and the total P-content delivered through those products is given in Figure 15. The analysis shows that in terms of product types, NPK fertilisers accounted for 55% of the tonnes consumed, followed by DAP (18%) and SSP (8%) (Figure 15). In terms of **actual P delivered** however, because NPK fertilisers contain less P than MAP, DAP or TSP, the proportions change. **DAP has the largest share at 32%, followed by NPK fertilisers at**

31%, and MAP with 12% (Figure 15). The chemical composition for the most important mineral P-fertilisers is given in Box 1.



NB: The left figure indicates the actual tonnes of material; the right figure indicates the same data expressed on a P-basis.

Figure 15: Apparent consumption of P-fertilisers in the EU-28 for the year 2015 (Source: Fertecon)

The data from Fertilizers Europe indicate that France is the largest consumer of phosphates for fertilisers, with a share assessed at 17% of the EU-28 in 2015. Spain is the second largest market (15%) followed by Poland (13%). The top seven markets accounted for just over 76% of apparent consumption of phosphate for fertilisers, with the balance spread over the remaining EU countries (Figure 16).

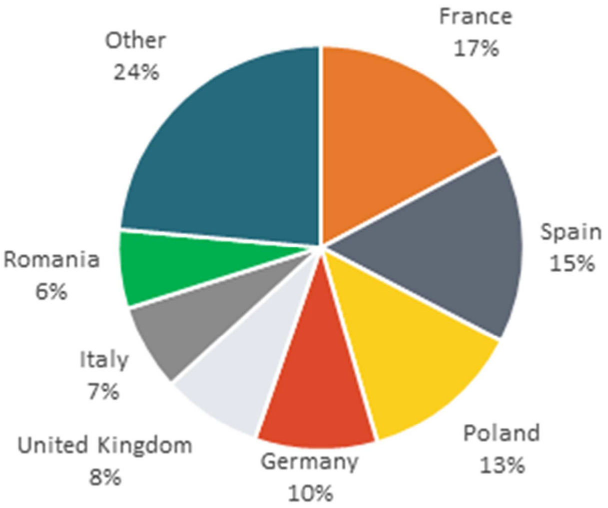


Figure 16: EU-28 Top 10 markets for apparent consumption of mineral phosphate fertilisers for the year 2015 (Source: Fertilizers Europe)

Box 1: Production of DAP, MAP, TSP, SSP and NPK in the EU (AEEP, 2017; Fertecon)

- **Diammonium phosphate (DAP, $(NH_4)_2HPO_4$):** DAP is typically 18-46-0 (i.e. contains 18% N, 46% P_2O_5 and 0% K_2O). It was one of the first fertilisers to have a standardised content, which in part explains why it is the best-selling phosphate fertiliser. Annual production of DAP in the EU-28 is just under 1.0 Mt or 200 kt P, with Lithuania by far the largest producer, followed by Poland, Spain and Belgium.
- **Monoammonium phosphate (MAP, $NH_4H_2PO_4$):** MAP can be between 10-50-0 and 11-55-0. Annual production of MAP in the EU-28 production is a modest 78 kt (18 kt P), with Belgium, Poland, Bulgaria and Spain the main producers.
- **Single superphosphate (SSP; $Ca(H_2PO_4)_2 + CaSO_4$):** SSP is typically between 0-16-0 and 0-22-0. Annual production of SSP in the EU-28 accounts for just over 1.0 Mt or 88 kt P. No information is available on imports/exports.
- **Triple superphosphate (TSP; $Ca(H_2PO_4)_2$•H2O):** TSP is the most concentrated straight phosphate fertiliser, typically ranging between 0-44-0 and 0-48-0. Annual production of TSP in the EU-28 accounts for just over 250 kt or 53 kt P, mostly in Bulgaria.
- **NPK fertilisers:** No direct data are available on the total production of NPK fertilisers in Europe.

7.3 Market outlook for P fertilisers derived from STRUBIAS materials for the year 2030

7.3.1 General considerations

The market potential of STRUBIAS materials is based on estimates of plant-available P. The agronomic efficiency of fertilisers containing STRUBIAS materials is expressed relative to mineral P-fertilisers based on the 'relative agronomic efficiency' (see Section 6.2.1). A relative agronomic efficiency value below 1 indicates that that the fertiliser derived from STRUBIAS materials is a less effective plant P-source than a mineral P-fertiliser derived from mined phosphate rock, and vice versa.

The market for P-fertilisers derived from STRUBIAS materials is dependent on the **technological readiness** and potential production limitations of the production processes, **market and consumer readiness**, and the **impacts** along the STRUBIAS material life cycle (Figure 17). Legislative and policy impacts due to the possible beneficial impacts of STRUBIAS materials relative to alternative fertilising products that are available on the internal market are considered.

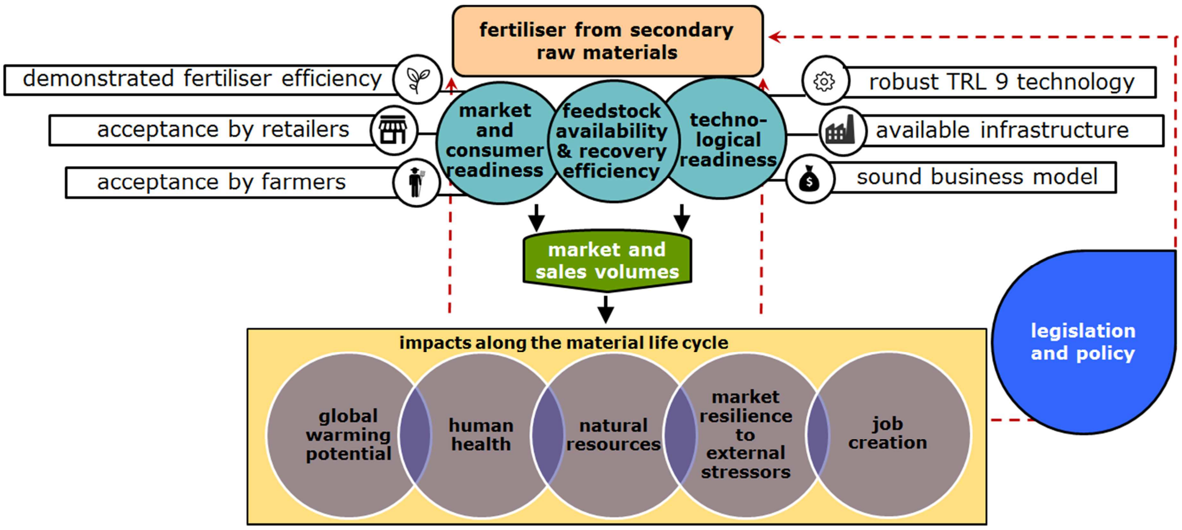


Figure 17: Schematic overview of the market drivers for P-fertilisers derived from secondary raw materials

The market for STRUBIAS materials will not only depend on the nutrient recovery rules as laid down in the fertiliser product legislation, but also on national and EU legislation related to specific **eligible input materials** (e.g. Directive 2008/98/EC on waste and by-products, Regulation (EC) No 1069/2009 as regards animal by-products, Regulation (EC) 86/278 as regards sewage sludge), **nutrient use and management** in crop and livestock production, and **prevention and reduction of water pollution** (Buckwell and Nadeu, 2016). Moreover, links exist with the **bioeconomy initiatives and related legislation**, and more specifically Directive 2009/28/EC on the promotion of the use of energy from renewable sources. The existing legal framework and policy incentives vary considerable as a function of eligible input materials. When relevant, the relation of existing EU legislation to STRUBIAS market aspects will be discussed for the eligible materials. **In the event that STRUBIAS materials are associated with a positive impact on the environment, human health, or the circular economy in general, positive feedback loops due to legislative and policy initiatives could possibly further stimulate the market** (see Section 7.3.8.2).

Sections 7.3.4 to 7.3.8 discuss market estimates under the anticipatable EU and national legislation for all process pathways with a high technological readiness level (TRL 7-9). A broad spectrum of new STRUBIAS production processes are emerging and developing, some of them already being at the TRL 6 level (‘System/subsystem model or prototype demonstration in a relevant environment’). **It is, nonetheless, challenging to predict a 2030 market for production pathways that might still face technological and market failures during the remaining TRL stages ahead.** Therefore, the market assessment of TRL 6 processes is discussed together with possible legislative drivers in Section 7.3.8.2.

7.3.2 Market and consumer readiness

7.3.2.1 Conventional agriculture

STRUBIAS materials are used to produce a variety of fertilising products (see Section 14). On the one hand, **STRUBIAS materials can be used for the production of traditional P-fertilisers with a well-known chemical composition such as DAP, MAP, TSP, SSP**, etc. On the other hand, STRUBIAS production processes may expand the variety of P-fertilising products on the market by producing P-fertilising products **with a different chemical composition**, with P present in a water-insoluble form. The development of new fertiliser products requires the determination of key properties of the materials that affect storage and spreading, soil behaviour, and agronomic efficiency. **Knowledge of these properties is of prime importance to increase market and consumer readiness for new fertilisers** (Antille et al., 2013).

STRUBIAS materials should preferentially be **available in a physical form that enables their homogeneous distribution across the agricultural field and their application using conventional application equipment**. With the increased need and readiness of the sector to increase nutrient-use efficiency, it is important that the fertiliser characteristics enable the accurate distribution of fertilisers within the field. At the same time, there is a strong preference to use existing machinery that applies fertilisers in a time- and energy-efficient manner, which is why the physical form of the fertilisers should be compatible with existing machinery. This implies, for instance, that fertilisers that will be broadcast should preferentially comply with physical characteristics such as bulk density, grain size distribution, sphericity, hardness, brittleness, dust rate, resistance to humidity, resistance to weight increase, etc. (Antille et al., 2013). Fertiliser materials which have moderately high crushing strength can better resist handling, storage and spreading without significant shattering, dust formation, or caking. Density properties are related to the volume needed for storage and transport, and are required to calibrate fertiliser-spreading equipment. Particle size and size distribution affect the uniformity of distribution during field application, and it is well documented that uneven spreading of fertilisers can increase nutrient losses to the environment, reduce fertiliser use efficiency and crop profit margins (e.g. Horrel et al., 1999). Alternatively, materials could be applied using lime spreaders that homogeneously distribute fertiliser materials on the field, albeit in a more labour-intensive manner due to the reduced distribution width relative to modern fertiliser broadcasters.

Fertiliser blending companies could incorporate STRUBIAS materials as part of a physical or chemical blend together with traditional mineral fertilisers on condition that they meet certain characteristics (e.g. with respect to material purity and granulometry (Formisani, 2003)). Also, certain combinations of molecules should be avoided due to possibly occurring chemical reactions in the granulator that cause nutrient loss or reduce the water-solubility of specific elements in the blend. Certain fertiliser companies are in the process of **testing and/or partially including struvite and poultry litter as part of compound mineral P-fertilisers**. ICL Fertilisers has successfully tested partial struvite incorporation in their compounding process (20% of the total P in the blend). Given that the P in good-quality precipitated phosphate salts & derivates is already in plant-available form, there is no need for acidulation; ICL tests indicated that in quantities < 20%, precipitated phosphate salts & derivates can be placed directly in a granulator with acidulated phosphate rocks (Six et al., 2014). In such a process, struvite is physically mixed or blended with phosphorus fertilisers and possibly with N, K and other nutrient sources. It is not chemically converted. Struvite will in this case be used as a P, N and Mg source and will require identical application machinery to traditional P-fertilisers. Some fertiliser blending companies formulate poultry litter ashes as powder with KCl or TSP.

There is an increased acceptance for innovative fertilisers within the European agricultural sector, but market and consumer readiness is enhanced for products with **proven agronomic benefits and comparable fertiliser efficiencies relative to traditional P-fertilisers expressed on a monetary basis**. Such evidence could soon be available after **comprehensive agronomic testing** under different climate and soil conditions are complete, which will require a minimum of 3 or 4 years of field trials, and probably more. With the exception of struvite and poultry litter ashes, such assessments have not yet started for most recovered products that have a different chemical composition to those products currently dominating the market (DAP, MAP, TSP, SSP, nitrophosphate, etc.).

In conventional European agriculture, the P-fertilisers MAP and DAP (with a P_2O_5 content of 48-61% and 46%, respectively) or physical blends (NPK) make up more than 75% of the total mineral P-fertilisers applied, expressed on a P-basis (see Section 7.2.3.2). The straight P-fertiliser TSP (45% P_2O_5) and SSP (16-20% P_2O_5) only make up 18% of the total share of all P-fertilisers, expressed on a P-basis (see Section 7.2.3.2). A major reason for this observation is the ease and efficiency of fertiliser application and distribution logistics. **The nutrient content of the fertiliser directly impacts upon the logistic cost for the transport, distribution and application of fertilisers.** Therefore, nutrient-dense fertilisers will provide clear benefits for the downstream transport, distribution and application by retailers and farmers.

Some recovered fertiliser products could also enter the market in **specific segments that deliver the highest margins**. Struvite and poultry litter ashes are, for instance, currently already sold in specific niche segments of the fertiliser market. In the early stages of market development, it may lead fertiliser producers to choose physical and chemical forms that are specifically adapted to these markets. The STRUBIAS application potential in niche markets includes for instance use in **growing media** for greenhouse farming or the fertilisation of **grasslands** where some STRUBIAS materials show a high potential. Moreover, STRUBIAS materials could be marketed as alternatives to mined rock phosphate and processed P-fertilisers, possibly providing market opportunities for customers attentive to alternatives derived from waste materials within a circular economy framework.

Assuming a similar price setting to traditional P-fertilisers, it is concluded that fertilisers derived from STRUBIAS materials should meet numerous conditions in order to provide a substantial spot-on alternative for mined rock phosphate and processed P-fertilisers in conventional farming. The fertiliser end-product should preferentially be presented in a **physical form that enables its efficient application** and its **agronomic efficiency should be well demonstrated**. Additionally, it is advisable that P-fertilisers derived from STRUBIAS materials have characteristics that enable the use in fertiliser blends together with other plant macronutrients to increase nutrient density. **Therefore, STRUBIAS materials contained in compound P-fertilisers with a chemical composition that is already available on the market and acid-soluble P-fertilisers that are well advanced in agronomic efficiency testing and are compatible for blending are associated with the highest degree of market and consumer readiness, at least for conventional agriculture.** Phosphorous fertilisers that do not meet these conditions may possibly be relevant for specialised niche markets or be traded at a lower price.

7.3.2.2 Organic farming

Depending on the input materials and process pathways employed, certain STRUBIAS materials could be used as fertilisers in organic farming**.** Agricultural practices, including fertiliser management, are regulated under the existing legislation (**Council Regulation (EC) No 834/2007 on organic production and labelling of organic products**). The EU organic farming sector mainly uses organic fertilising materials, but under specific circumstances natural or naturally-derived substances and low-solubility mineral fertilisers can also be applied. **Synthetic resources and inputs may only be permissible if there are no suitable alternatives.** Such products, which must be scrutinised by the Commission and EU countries (the Expert Group for Technical Advice on Organic Production – EGTOP, see below) before authorisation, are listed in the annexes to the implementing regulation (Commission Regulation (EC) No. 889/2008).

The manufacturing of many STRUBIAS fertilisers from secondary raw materials is in line with the objectives, criteria and principles of organic farming and the responsible use of natural resources. Given the limited availability of concentrated P-fertilisers that comply with the principles of organic farming, P-fertilisers derived from STRUBIAS materials could potentially fulfil an important role as fertilising material for the sector, especially in organic systems that rely on biological N fixation for N inputs (Seufert and Ramankutty, 2017). P-fertilisers derived from STRUBIAS materials could provide an alternative for meat and bone meal, meat and bone meal ashes and lower concentrated P-fertilising products such as manure and compost that are currently the major P-sources in organic farming (Nelson and Janke, 2007). In 2015, the percentage of the total utilised agricultural area within the EU for organic farming was 6.2%, of which 42% was planted with arable crops. **The expanding organic farming sector could, however, become a more significant agricultural market in the near future.** The Common Agricultural Policy (CAP) (2014-2020) recognises the role of organic farming in responding to consumer demand for more environmentally friendly farming practices: under the first pillar, organic farms will benefit from the green direct payment without fulfilling any further obligations because of their overall significant contribution to environmental objectives.

The EU's **'Expert Group for Technical Advice on Organic Production' (EGTOP)** has **positively evaluated two dossiers** proposing authorisation of recycled phosphate products as fertilisers in organic agriculture (under the EU Organic Farming Regulation (889/2008)). The dossier for struvite was submitted by the UK in 2014 and concerns struvite (magnesium ammonium phosphate) recovered in sewage works or from animal waste processing. The dossier for calcined phosphates was submitted by Austria in 2011 and concerns recovery from ashes of sewage sludge, meat and bone meal, or other biomass ash. **The committee concludes that for Ostara Pearl struvite (the submitted dossier) there is no hygiene risk (organic pollutants or pathogens), but that this is not proven for other struvite production methods and struvite-like end materials.** The EGTOP concluded that struvite recovery conforms to environmental objectives (reduces N and P losses to surface waters, recycles nutrients, reduces consumption of non-renewable P resources) and that struvite should be authorised for organic farming **'provided that the method of production ensures hygienic and pollutant safety'.** For calcined phosphates, the committee also concludes that recovery from ashes conforms to environmental objectives (but with some concerns about energy consumption) and that calcined phosphates should be authorised for organic farming subject to being recovered from sewage sludge incineration ash and that the heavy metal content should be limited. The EGTOP also concluded that these two products cannot be authorised under the Organic Farming Regulation until they are authorised under the EU Fertilisers Regulation. **Properly formulated nutrient recovery rules integrated into the EU Fertilising Products Regulation ((EU) 2019/1009) could address the above considerations from the EGTOP and hence facilitate the market entry of STRUBIAS materials into the organic farming market.**

It should be noted, however, that STRUBIAS materials for organic farming are most likely to replace currently used (organic) P-fertilising materials as the use of mined rock phosphate and processed P-fertilisers is negligible in this agricultural sector. Regardless of the volumes used in organic farming, **the use of STRUBIAS materials as P-sources in organic farming is unlikely to contribute significantly to the substitution of mined rock phosphate and processed P-fertilisers** unless a significant expansion of organic farming occurs prior to the year 2030 at the expense of traditional farming.

7.3.3 Technological readiness level and potential production limitations

Due to the combination of diverse input materials and output materials that can be produced, STRUBIAS production processes span a broad range of technological readiness levels. For the assessment of the market outlook for P-fertilisers derived from STRUBIAS materials for the year 2030, only STRUBIAS production processes with **technological readiness levels (TRLs) 6-9 are considered**:

- TRL 6 – technology demonstrated in relevant environment (industrially relevant environment in the case of key enabling technologies);
- TRL 7 – system prototype demonstration in operational environment;
- TRL 8 – system complete and qualified;
- TRL 9 – actual system proven in operational environment (competitive manufacturing in the case of key enabling technologies; or in space).

TRL 6 processes are covered in Section 7.3.8.2 whereas TRL 7-9 processes are split up as a function of eligible input material (see Sections 7.3.4 to 7.3.7).

The underlying justification for excluding TRL 1-5 processes is the unlikeliness of technologies with a lower technological readiness level to:

- surpass TRLs 6-9;
- comply with administrative arrangements (e.g. obtain a waste treatment permit, REACH registration of CE fertilising product containing STRUBIAS materials, arrangements with downstream distributors and retailers);
- build the necessary infrastructure; and
- perform extensive product testing on agronomic efficiency,

in order to ensure market and consumer confidence (see Section 7.3.2) prior to the year 2030. Although TRL 1-5 STRUBIAS materials are not covered in this section, the possibility of such emerging process pathways contributing to the substitution effect for mined rock phosphate and processed P-fertilisers in the mid to long term is not excluded.

An additional point that should be taken into consideration to derive the substitution potential for the year 2030 is the **availability of infrastructure** required to produce STRUBIAS materials. STRUBIAS production processes often include different phases and specific configurations within the production process: struvite can be precipitated in wastewater treatment plants that use biological nutrient removal techniques, K-struvite can be precipitated from manures after anaerobic digestion, DAP can be produced from mono-incinerated sewage sludge ashes, etc. As a matter of fact, on most occasions, STRUBIAS production takes place as part of a material cycle that also serves other objectives, for instance the handling and recycling of wastes and by-products in a bio-economy context, the production of a different primary material (e.g. clean water effluents, steel), etc. Therefore, the substitution potential will be partially dependent on the trade, market, needs and infrastructure to deal with co-products that are formed as part of the STRUBIAS material cascade. An additional aspect related to infrastructure that is especially relevant to consider for thermal oxidation materials & derivates is the **configuration of the incinerator** (mono- versus co-incineration). This choice has a large impact on the suitable further use of the ashes from sewage sludge and meat and bone meal for nutrient recovery. It is preferable that P-rich input materials are mono-incinerated because co-incinerated materials are less suitable for P-recovery due to P-dilution. Even in Member States such as Germany and the Netherlands that incinerate a dominant share of their sewage sludge, mono-incinerators currently only process 50% or less of the current volumes (STOWA, 2011; Wiechmann et al., 2013a).

A major challenge also lies in increasing the production volumes of P-fertilisers derived from secondary raw materials. With the exception of manure ashes, struvite production plants, EcoPhos and 3R agrocarbon (animal bone biochar), **P-recovery facilities are at present still piloting or producing minor P-fertiliser volumes compared to the apparent fertiliser P-consumption in Europe**. Other promising technologies, such as RecoPhos, Budenheim, Green Mineral Factory and many others, will start to build full-scale installations in Europe. The extent to which current P-fertiliser production processes through the acidulation route could be adapted to use other P-sources besides phosphate rock (for instance P-rich ashes) should be evaluated. Leading P-fertiliser companies aim at decreasing their reliance on phosphate rock, but further technical process refinements may be required. Therefore, the recovered P-fertiliser volumes will only gradually become more abundant in the coming years. ICL Fertilisers has, for instance, expressed the ambition of substituting 25 kt of P yr^{-1} by the year 2025.

In the following sections, the market potential for STRUBIAS materials derived from different input materials will be discussed, considering possible limitations on feedstock, P-recovery efficiency and available infrastructure.

7.3.4 STRUBIAS materials from crop residues, manure, and bio-waste

7.3.4.1 Introduction

In Europe, most **agricultural crop residues arise** on farms in the form of straw, maize stover, residues from sugar beet, oilseeds, grass cuttings, and pruning and cutting materials from permanent crops, and in the crop processing sector in the form of olive pits, seed husks and nutshells. By far the largest source of crop residues is the straw and stover from grain crops (wheat, barley and maize) (Kretschmer et al., 2013). There are essentially two overarching challenges to mobilising crop residues (Kretschmer et al., 2013). **Transport costs are high** because the residues are highly dispersed and have high bulk volumes and low value. This limits the range over which they can economically be collected for processing and makes it important that processing plants are optimally located. This requires appropriate investment in machinery and equipment, which may be beyond individual farmers' reach and necessitates cooperative action or specialised contractors. Harvesting costs can also be high in relation to the value of the material. Secondly, **many crop residues have existing uses and established practices, particularly for recycling organic materials and nutrients back to the soil**. There is poor awareness of sustainable extraction rates in relation to local conditions. There are therefore real risks that overextraction could cause detrimental reduction of soil organic matter with knock-on effects for wider soil functionality, soil biodiversity and erosion risk. Taking these issues into account, the realistic potential derived from the technical-sustainable potential for agricultural crop residues – excluding grasslands – 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, maize and barley (Iqbal et al., 2016). The total P content in those crop residues would be around 110 kt P yr^{-1} (see Section 13.2).

Much of the off-land technology for dealing with biomass is well understood and long established. Generally, the biomass-based raw materials will require some physical **pretreatment**, for example to separate components, dry, chop and pelletise. Then, the processing will either follow a **biochemical pathway** (based on digestion, transesterification, or fractionation, the latter also serving as a type of pretreatment) or a **thermochemical process** (based on hydrogenation, gasification or pyrolysis). The review of a wide range of life cycle assessments (LCAs) for different treatments for crop residues shows the superiority of the anaerobic digestion pathway over other pathways for energy recovery and other bio-based applications from crop residues (Kretschmer et al., 2013). Thermochemical conversion technologies are less suitable for the direct processing of crop residues, characterised by a relatively high moisture content.

Manure is defined as a category 2 Animal By-Product material according to Regulation (EC) No 1069/2009. Pursuant to this Regulation, **it can be applied to land without processing**. Most European countries have similar supplementary regulations regarding livestock farming including (i) licensing required for housing animals, (ii) storage of manures and slurries to enable a better agronomic utilisation and (iii) prohibited periods for landspreading (usually the winter months of November to February). There are, however, differences between countries - and even between regions of the same country - as a consequence of the local situations and locally defined Nitrate Vulnerable Zones. A common pollution concern is nitrate contamination of water, but in most countries there are other pollution issues including ammonia emission (in the Netherlands) and odour nuisance (in the UK and Greece).

In 2010, about **7.8% of the livestock manure production in the EU was processed, equal to a total manure volume of 108 million tonnes/year, and containing 556 kt N and 139 kt P** (Foget et al., 2011; Flotats et al., 2013). At least **45 different manure treatment technologies** are available (Foget et al., 2011). The highest levels of livestock manure processing were recorded in Italy, Greece and Germany, with 36.8%, 34.6% and 14.8% of the manure production being processed, respectively. The **objectives** for manure processing include **increasing the handling and management** for storage and transport of nutrients (viscosity, greenhouse gas emissions, storage of reduced volumes, etc.), the **selective removal of nutrients** (especially N), and **incentives received for renewable energy production** (biogas).

A clear tendency identified is that anaerobic digestion is the 'door-opener' for the introduction of nutrient recovery technologies for cattle and pig manure with a high moisture content (Foget et al., 2011). Several EU Member States have no other type of manure processing besides anaerobic digestion (Foget et al., 2011). The reason for this is that most of the manure nutrient processing technologies are complementary to anaerobic digestion, either as pretreatment technologies that can enhance the biogas production, or as post-treatments, which can help to convert the digestate into products with particular properties (Foget et al., 2011). Solid manure fractions, such as poultry manure, can also be used for direct incineration and thus the manufacturing of thermal oxidation materials & derivates, as currently already performed in different EU Member States.

Bio-waste means biodegradable garden and park waste, food and kitchen waste from households, restaurants, caterers and retail premises and comparable waste from food processing plants. Across the European Union, somewhere between 118 million and 138 million tonnes of bio-waste arise annually, of which currently only about 25% is effectively recycled into high-quality compost and digestate (European Commission, 2010a). The total P content of this fraction is unknown, but solid residues from the food processing industries (e.g. brewery, sugar industry) make up at least 36 kt P (see Section 13.6).

Co-digestion of animal manure with solid or slurried bio-waste with high methane potential, such as oily residues and by-products, alcohol residues, digestible organic wastes from agri-processing and the food industry or food waste, produces more gas from the digester than manure only (Al Seadi and Lukehurst, 2012). Co-digestion can therefore improve the profitability of biogas plants, and is a common practice in many biogas facilities in Europe (Foget et al., 2011). In addition, co-digestion of animal manure and slurry with suitable organic wastes from food industries utilises the huge amounts of organic wastes that are produced annually and in many places otherwise dumped into landfills. The liquid fraction of co-digestates varies in phosphate content depending on the composition of the input materials; digestates that contain a dominant share of manure and food industry residues (e.g. sugar production, residues from the brewery industry) typically have phosphate contents above 500-2 000 mg L^{-1} $PO_4^{3-}P$, whereas liquid digestate fraction produced from predominantly plant-based materials have much lower phosphate contents (~0-500 mg L^{-1} $PO_4^{3-}P$) (Akhiar et al., 2017).

7.3.4.2 Policy and legal framework

The field extraction of crop residues and manure for STRUBIAS nutrient recycling processes manifestly result in the loss of nutrients and other valuable agronomic assets, such as soil organic matter. Therefore, it is clear that STRUBIAS nutrient recycling processes from these materials will only take place for reasons other than nutrient recycling close to the site of extraction of manures and crop residues (hygienisation, transport logistics, synergies with energy recovery, etc.). Fundamentally, it is assumed that **significant materials from the agricultural sector might be used as input materials for STRUBIAS processes bearing in mind two different rationales**:

i. **Pollution control**: in the case of manure, in European regions characterised by nutrient excess and the need for the long-distance translocation of fertilising products to nutrient-poor regions or storage under appropriate conditions.

ii. **Synergies with other bio-based materials**: secondary raw materials from the agricultural sector are used for STRUBIAS production processes as part of a cascading approach; in this case, the input materials for the STRUBIAS production processes are the residues of a process aimed at producing a different primary raw material from crop residues and manure, often renewable energy.

Both rationales are not mutually exclusive and measures related to pollution control are mostly accompanied by additional benefits obtained through energy recovery.

As outlined in the Waste Framework Directive (Regulation (EC) No 2008/98), Member States shall take measures to encourage (a) the **separate collection of bio-waste with a view to the composting and digestion** of bio-waste; (b) the treatment of bio-waste in a way that fulfils a high level of environmental protection; (c) **the use of environmentally safe materials produced from bio-waste**.

Pollution control

International conventions of relevance to P use in agriculture include *inter alia* UNEP/MAP (**United Nations Environment Programme/Mediterranean Action Plan**), CBD (**Convention on Biological Diversity**) and OSPAR (**Oslo & Paris Convention to prevent pollution**). Such international treaties often give an impetus to harmonise standards amongst all Member States of the European Union. Despite the significant off-site impact that diffuse contamination of P from agricultural land poses, there is no specific legislation that is directly concerned with the use of P in agriculture at European level. There is a lack of appropriate institutional arrangements specific to the environmental pollution of P. Aspects of the P problem are, however, integrated in several policy areas and related legal instruments at European level. This section provides an overview of existing regulations and directives dealing with farm-level nutrients, including P, use and production at **European level**.

The **Water Framework Directive** (Directive 0060/2000) is a legal obligation to protect and restore the quality of waters across Europe. Measures applied under the Water Framework Directive affecting the use of P in agriculture relate to best environmental practices and include the reduction of nutrient application, the modification of cultivation techniques, the proper handling of pesticides and fertilisers, and the prevention of soil erosion through erosion-minimising soil cultivation. The P balance surplus is a commonly used indicator for identifying areas vulnerable to nutrient pollution in the pressures and impacts analysis.

The **Nitrates Directive** (Directive 0676/1991) established in 1991 aims to reduce water pollution caused or induced by nitrates from agricultural sources and to prevent further nitrate pollution. The Water Framework Directive explicitly refers to the Nitrates Directive for information on diffuse pollution of nitrates from agricultural activities and extends this to phosphates. Under the Nitrates Directive, Member States had to establish action programmes for Nitrate Vulnerable Zones with the purpose of meeting the objective of reducing and preventing nitrate pollution. It is the responsibility of each Member State to set limits appropriate to their vulnerable zones; there are no specific limits set in the Directive. However, the action programmes must include measures to ensure that, for each farm or livestock unit, the amount of livestock manure applied to land each year, including processed forms of manure and direct excretions by animals, shall not exceed 170 kg N per hectare. The measures established within the action programmes aim to control diffuse and direct water pollution and also to influence the use of P in farm practice. For instance, by limiting the annual application of N fertiliser and livestock manure, defining legally binding maximum concentrations of nitrates in drinking water and designating periods when the application is prohibited, the Directive clearly aims at establishing and maintaining the natural balance of nutrients in soils. Through these measures, a massive influx of nutrients to ground- and surface water and thus potential eutrophication is prevented, while excess nutrients, oversaturation and a possible ensuing soil quality degradation are avoided at the same time. The monitoring of waters for nitrates and the review of the eutrophic state of waters must be repeated every 4 years.

Pollution by P is also partially covered by the **Directive on Bathing Water** (Directive 0007/2006), which together with the Nitrates Directive and the Industrial Emissions Directive, has been linked since 2000 in the Water Framework Directive.

The **7th Environmental Action Programme** encourages the full implementation of the Water Framework Directive, in order to achieve levels of water quality that do not give rise to unacceptable impacts on, and risks to, human health and the environment.

The Rural Development Programme has established various agri-environmental measures throughout the European Union directly or indirectly addressing diffuse contamination by P. Some of these measures are directed at mitigating soil erosion such as crop rotations, mulch seeding, retaining stubble after harvest and ploughing restrictions. Other measures tackle the problem of excess nutrients through reduced fertiliser use. All measures that impact soil erosion and nutrient balances ultimately result in a reduction of diffuse contamination by phosphates from agricultural land.

The **Industrial Emissions Directive (IED, 2010/75/EU)** introduces an integrated cross-media approach, aiming to prevent or minimise emissions to air, water and land, as well as to avoid waste production with a view to achieving a high level of environmental protection as a whole. The IED also concerns potentially polluting industries in the agricultural sector, among which are intensive pig and poultry farms. A single permit based on the concept of Best Available Techniques (BAT including limit values) must include all arrangements made, including emission limit values for pollutants, for water, air and land, and may, if necessary, contain requirements for the protection of the soil and the groundwater as well as measures or waste management (Article 9(3)) in order to continuously prevent and reduce pollution. The purpose of the IED was to achieve integrated prevention and control of pollution arising from several categories of industrial activities. The indicative list of the main polluting substances to be taken into account if they are relevant for fixing emission limit values includes oxides of N and substances which contribute to eutrophication (P and N).

The main purpose of the **Habitats Directive** (Directive 0043/1992) and **Birds Directive** (Directive 0147/2009) is to ensure biological diversity through the conservation of natural habitats and wild flora and fauna within the European territory, while taking into account economic, social, cultural and regional requirements. Farmers who have agricultural land in

Natura 2000 sites and face restrictions due to the requirements of the Habitat Directive are eligible to receive payments for the management of these sites by the Rural Development Regulation, which helps promote environmentally friendly farming. Depending on the specific conditions of a certain area, these include measures to reduce the use of pesticides and fertilisers, measures to mitigate the effects of soil compaction, e.g. limitations on the use of machinery or the setting of stocking limits, or measures aiming to regulate the irrigation of agricultural land.

Specific EU Member States have put in place **national or regional regulations** to control for P use on farmland (Buckwell and Nadeu, 2016). Belgium (Flanders), Estonia, Finland, France (Brittany), Germany, Ireland, Luxembourg, Northern Ireland, Sweden and the Netherlands apply a regulation system that limits maximum P application rates. Limits are mostly dependent on crop type, soil P status and yield.

In spite of significant improvement in potential P surpluses within the EU, the **latest progress report on the implementation of the Water Framework Directive** (European Commission, 2015a) indicated that Member States need to strengthen their basic measures to tackle diffuse pollution caused by agriculture. Notwithstanding the fact that there is still a long way to go to achieving ‘good status’ (as defined in the Water Framework Directive), many Member States rely only on voluntary measures. Moreover, they need to ensure that their measures target the sources and chemicals that cause water bodies to fail to achieve ‘good status’. While these can effectively close a fraction of the remaining gap, significant improvement can only be achieved through compulsory basic measures. Member States should tackle the sources of pollution by fully implementing the Water Framework Directive measures and water-related legislation, especially the Nitrates Directive, Industrial Emissions Directive and Urban Wastewater Treatment Directive. This is far preferable to using end-of-pipe treatment, for instance to ensure the high quality of drinking water while avoiding high treatment costs and protecting the environment. Member States are encouraged to continue extending the establishment of safeguard zones to protect areas used for the abstraction of drinking water, in particular as regards surface waters.

The **last report on the Nitrates Directive** (European Commission, 2013b) points to a slight improvement in groundwater nitrate pollution while stressing the need for further action to reduce and prevent pollution. This is confirmed by the analysis of ‘Programmes of Measures’ reported by the Member States. Despite the fact that 63% of river basin districts reported that implementation of the Nitrates Directive is not enough to tackle diffuse pollution to the level needed to meet the Water Framework Directive’s objectives, necessary measures have not been added to address the remaining shortcomings. Diffuse pollution still affects 90% of river basin districts, 50% of surface water bodies and 33% of groundwater bodies across the EU. The agricultural sector is the primary source of diffuse pollution. **The report indicates that there are still many gaps in the basic measures put in place by Member States to address agricultural pressures, including a lack of measures to control phosphate and nitrate emissions outside Nitrate Vulnerable Zones established under the Nitrates Directive**.

Supplementary measures reported in agriculture are largely voluntary, including advice schemes and agri-environmental measures of the **Common Agriculture Policy (CAP)** such as **farm extensification** and **organic agriculture**.

Renewable energy

Renewable energy sources contribute to **climate change mitigation** through the reduction of greenhouse gas emissions, achieve sustainable development, protect the environment and improve citizens' health. Moreover, renewable energy is also emerging as a driver of inclusive economic growth, creating jobs and reinforcing energy security across Europe.

These aspects are enshrined in Article 194 of the Treaty on the Functioning of the EU, which has conferred the Union competences to promote renewable energy. The EU has long been a world leader in the promotion and development of renewable energy, steering the effort to combat climate change, encouraging the shift to a low-carbon economy and stimulating high-potential economic growth.

The current 2020 framework sets an **EU 20% target for energy consumption** which relies on legally binding national targets until 2020. **National Renewable Energy Action Plans (NREAPs)** and the biennial monitoring provided for by Directive 2009/28/EC on the promotion of the use of energy from renewable sources have been effective in promoting transparency for investors and other economic operators. This has favoured the rapid increase in deployment in the share of renewables from 10.4% in 2007 to 17.0% in 2015.

In October 2014, the European Council agreed the 2030 framework for climate and energy reaffirming the Union's long-term commitment to the ambitious EU strategy in renewable energies. **The new framework sets out the European Union target of at least 27% for the share of renewable energy consumed in the EU in 2030.** This target is binding at EU level and will be fulfilled through individual Member States' contributions guided by the need to deliver collectively for the EU. In addition, the new framework also enables the collective delivery to be done without preventing Member States from setting their own, including more ambitious, national targets. Member States can support renewable energy, subject to State aid rules.

To qualify for the Renewable Energy Directive targets, materials consumed in the EU must comply with strict **sustainability criteria** provided in Article 17 of the Directive, in order to be eligible for financial support and to count towards the EU renewable energy target. Rigorous requirements are set in the Renewable Energy Directive on the minimum level of greenhouse gas savings, appropriate land use, as well as monitoring requirements for any potentially adverse effects. Agricultural raw materials produced within the EU must be produced in accordance with the minimum requirements for good agricultural and environmental practices that are established in the common rules for direct support schemes under the Common Agricultural Policy. **Using manure and agricultural residues as feedstock is highly advantageous as it does not augment pressure on land and water**

resources and offers very high greenhouse gas mitigation gains (European Environment Agency, 2013b).

Materials from the agronomic sector can contribute to renewable energy production through the production of biogas and the thermochemical conversion of solid biomass. The resulting materials (i.e. digestates and ashes) are suitable intermediate or end materials of STRUBIAS production processes, including precipitated phosphate salts & derivates, thermal oxidation materials & derivates and pyrolysis & gasification materials.

The application of anaerobic digestion for biogas production remains widespread as a useful **bioenergy production** route due to the robustness of its main design configurations and pathways. Anaerobic digestion serves multiple purposes. It provides a treatment platform for decreasing large amounts of complex organic materials, converting the majority of such molecules into monomers, i.e. methane and carbon dioxide (biogas) utilisable in the energy sector in multiple pathways. The EU Common Agricultural Policy (CAP, Pillar II - Rural Development Policy) mechanisms provide support for capital investments on farm or as part of local renewable energy initiatives, including providing grant aid for the installation of anaerobic digesters.

The digestate produced can be used as fertiliser for crops without any further processing. However, the need for efficient nutrient management, required by restrictions on manure applications in areas with a high livestock density, make recovery and recycling of plant nutrients from manure and bio-based waste streams increasingly important for farmers. **The anaerobic treatment serves to improve the efficiency for nutrient recycling as solid-liquid separation procedures can be applied to separate the solid from the liquid digestate** (Möller and Müller, 2012). The solid phase may be characterised as an organic fertiliser comparable with solid animal manure with highly available N and P contents, best suited to application on arable land in order to increase soil humus reproduction and to substitute P-losses via harvested P-rich biomass such as grains. Separated liquid digestates are characterised as liquid NK fertilisers. A digestate solid-liquid separation, with a target-oriented separate application of the liquid and solid phase, is, therefore, a technique for further improvement of the nutrient use efficiency upon return to the field (Möller and Müller, 2012).

The traditional substrates for anaerobic digestion plants in Europe are **agricultural/livestock residues** (manure and slurries), **biogenic waste** (food waste, municipal organic waste, etc.), **energy crops** (maize whole crop silage, sugar beet, grass silage), as well as **residues from food and agro-industries** (animal by-products from abattoirs, brewers' spent grains and solubles, etc.) (Drogs et al., 2015). More recently, residues from the **bioethanol** and the **biodiesel** industries have been used. Producing biogas from dedicated energy crops, such as maize, sugar beet or wheat, requires careful analysis due to their land use implications. The emissions of greenhouse gases and acidifying gases such as ammonia from these systems are substantial. The use of specifically grown energy crops for biogas has thus been questioned due to sustainability concerns. Such energy cropping patterns are not 'environmentally compatible' according to the criteria of a 2006 EEA study (European Environment Agency, 2006). Where manure or organic residential wastes are used, the greenhouse gas performance of biogas pathways is far better (European Environment Agency, 2013b). **Wastes and crop residues are currently underused and can contribute significantly to reaching EU bioenergy targets** (European Environment Agency, 2013b). The latter report projects that agricultural residues and organic waste would contribute 44% of the total supply for meeting the NREAP bioenergy targets for agriculture. Therefore, introducing new, cost-competitive and sustainable feedstock, such as manure, straw and even grass, will be increasingly important for the biogas sector to deliver its full potential in the future energy scenario.

Biogas is a diverse energy source, suitable as a flexible and storable energy form. Between 2000 and 2013, the production of biogas in Europe increased six-fold, from 2.2 Mtoe to 13.5 Mtoe, with the main producers being Germany, the UK and Italy, followed by the Czech Republic, France and the Netherlands. Germany is the leader in biogas production from biomass, with more than 65% of the EU production in 2013 (Flach et al., 2015). For Germany, it was estimated that in the year 2011, approximately 20% of the animal wastes and the biomass harvested from an area of approximately 1.1 million ha was used as feedstock in biogas plants (Möller and Müller, 2012). Italy, the Czech Republic and the Netherlands followed with a production share of 14%, 5%, and 2%, respectively (Flach et al., 2015). The incentive for farmers in Germany to invest in biogas digesters is a guaranteed feed-in price for the electricity generated, which is considerably higher than that of electricity generated from fossil fuels, natural gas, coal, or nuclear sources. This feed-in price is guaranteed for 20 years from the erection of the plants. However, changes to the German renewable energy law in 2012 and 2014 and similar policy changes in Italy **reduced the attractiveness of investing in new plants (Flach et al., 2015)**. As a result, the further increase in biogas plants will be minimal. Instead, investments will focus on rejuvenating existing plants. Biogas production is increasing in the Czech Republic (driven by feed-in tariffs for the derived electricity) and Denmark (driven by the goal to use 50% of livestock manure for biogas production in 2020) (Flach et al., 2015). In France, the government seeks to increase the number of biogas facilities by means of investment support. However, administrative burden and a lack of profitability for investors limit the expansion. The development is also stagnant in Slovakia and Hungary. Slovak energy distribution companies announced a blanket stop on connecting new electricity production facilities (over 10 kW) to the grid until further notice. Hungary reports problems with green energy feed-in systems and the complicated non-harmonised investment licensing. In addition, low electricity purchase prices make further investments in biogas facilities economically unattractive. In the Netherlands, the low electricity prices have even led to a decline in biogas production.

Landfilling of biodegradable waste

The **Landfill Directive (1999/31/EC)** obliges Member States 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%.

7.3.4.3 Anaerobic digestion followed by P-precipitation

The European Environment Agency forecast a reduced growth rate for biogas production in Europe, with a predicted **annual growth rate of 3% for the period 2013-2020 based on the NREAPs** (European Environment Agency, 2016).

The anaerobic digestion process does not affect the phosphate content in digestate, which is completely dependent on the content in the substrate. Nonetheless, the technique is a door-opener for manure solid-liquid separation processes (Foget et al., 2011).

Box 2: Precipitated phosphate salts & derivates from the liquid fraction of anaerobically digested materials

To derive the substitution potential of P-fertilisers containing STRUBIAS materials from the eligible input materials that have undergone an anaerobic digestion process, the following assumptions and calculations were made:

1) **Digestates from manure and specific food industries** are more suitable for P-recovery than digestates from crop residues and other bio-waste materials due to the higher P content in the liquid digestate fractions. Although co-digestion of other organic materials (food waste, crop residues) is common practice, the liquid fraction of such digestates is typically low in phosphates that can be precipitated (Akhiar et al., 2017). Therefore, this assessment only takes into account the P present in anaerobically digested manures and solid residues of food processing industries.

2) In 2010, about 88 million tonnes of **manure**, excluding other organic materials as co-substrates, were anaerobically digested in the EU (Flotats et al., 2013). Anaerobic digestion is also a door-opener for separation processes and treatment of the liquid manure fraction (Foget et al., 2011). Using the average P-contents of Foget et al. (2011), the total manure P that is subjected to anaerobic digestion was estimated at 114 kt for the year 2010. The European Environment Agency (2016) indicates that biogas production from anaerobic digestion would roughly double in the period 2010-2020. After 2020, an annual 3% growth, similar to the period 2013-2020, is assumed. Therefore, it is assumed that the total volume of materials from the agricultural sector would increase by a factor 2.69 relative to the amounts processed by anaerobic digestion in the year 2010. Hence, **a total amount of 237 million tonnes of manure would be processed through anaerobic digestion, with an estimated P content of 306 kt**. This number is considered realistic; it equals the excess P that accumulates in soils from six livestock-dense Member States: Belgium, Denmark, Germany, France, the Netherlands and the UK (based on data for the year 2005 by van Dijk et al., 2016).

3) P-rich residues from the **food processing industry** are taken into account (36 kt P yr^{-1}, mainly from sugar residues and brewery industries; for the reference year 2005). These P-estimates are assumed to remain steady for the year 2030.

5) It is assumed that **precipitated phosphate salts & derivates are recovered from all anaerobic digesters that process manure and solid processing residues**. The possible non-compliance with this assumption is presumed to be counteracted by the fact that liquid manure fractions can also be obtained by solid-liquid separation techniques on raw manures, thus not after anaerobic digestion.

4) At present, the P-recovery process of **Stichting Mestverwerking Gelderland (SMG)** is the only process at TRL level 7-9 in Europe that processes P-rich digestate liquids and that complies with the draft proposals of the STRUBIAS nutrient recovery rules. In this process, a mineral phase predominantly composed of K-struvite is precipitated from the liquid digestate fraction. The assessment of the process indicates the business case for the plant in the Netherlands where manure treatment is associated with a gate fee. It is assumed that more operators might emerge in other livestock-dense EU regions in the near future.

5) Following solid-liquid separation of the digestate, between 10% and 30% of the total P content will end up in the liquid digestate fraction, and about 80-95% of this P in the liquid fraction can be recovered as precipitated phosphate salts & derivates. Therefore, the total P-recovery efficiency from anaerobically digested manure varies between 8% and 28.5%. The exact numbers are highly dependent on the separation technology applied (Drosg et al., 2015). In order to estimate the P-recovery potential, **a P-recovery efficiency of 13.3% as indicated by SMG** is used.

6) The **relative agronomic efficiency (RAE_{PUE})** of precipitated phosphate salts & derivates relative to mined rock phosphate and processed P-fertilisers is assumed to be 1.05 (see Section 6.2.2).

7) Considering 1) to 6), **the 2030 P-recovery from anaerobically digested eligible input materials is estimated at 48 kt P yr^{-1}** ((306 kt P yr^{-1} (manure) + 36 kt P yr^{-1} (food processing) * 0.133 (recovery efficiency) * 1.05 (RAE)). It is estimated that these materials will be brought onto the market directly as P-fertilisers or as part of a physical blend, and that their further processing in recovered phosphate salt derivates (e.g. MAP, DAP, TSP, nitrophosphate) is unlikely.

7.3.4.4 Thermochemical conversion processes of solid fractions from the agricultural sector

The **poultry manure** fractions and other manure fractions that have undergone **a solid-liquid separation process** might be suitable for thermochemical conversion processes due to their relatively low moisture content. Both **thermal oxidation and pyrolysis spectrum techniques** are considered in the STRUBIAS project.

Thermal oxidation

Thermal oxidation processes have been widely used for the production of energy (electricity and power). **Poultry litter incinerators** in the UK (Fibrophos), Ireland (BHSL), the Netherlands (BMC Moerdijk) and some Scandinavian Member States produced approximately **150 000-200 000 tonnes of poultry litter ash** per year in the year 2015, equalling an estimated recovery of about **12-16 kt P yr^{-1}** (assuming a P content of 8%). A best estimate of 14 kt P yr^{-1} is assumed.

The aim of bioenergy technologies is to convert biomass into different forms of energy including power, heat, combined heat & power (CHP) and liquid biofuels. **The primary aim of poultry litter combustion is the production of renewable energy** (Billen et al., 2015). The calorific power of poultry litter when used as a fuel is about half that of coal (Moore, 2013). In order to meet NREAP expectations, **a compound annual growth of 7% over the period remaining up to 2020 would be necessary (European Environment Agency, 2016).**

The CE Delft research firm established the environmental effects of nine different ways that poultry litter – from the chicken to the field – can be used (Ehlert and Nelemans, 2015d; Ehlert and Nelemans, 2015c; De Graaff et al., 2017). The study concluded that the production of electricity through thermal oxidation is the most **attractive and sustainable way to process poultry litter from an environmental perspective.**

In Section 6.2.3, a relative agronomic efficiency for the response variable phosphorus use efficiency (RAE_{PUE}) of 1.49 was observed. The accuracy of this value is, however, uncertain, because this value was based on only four cases. Data from a research study performed by Alterra Wageningen UR, cited in De Graaff et al. (2017), indicated a relative fertiliser efficiency varying from 37% to 100%. These data were not included in the meta-analyses of Section 6.2.3 due to confidentiality issues at the time of writing. Therefore, the lower end estimate of **0.90 for RAE_{PUE} for poultry litter ashes was retained for further calculations**; this value corresponds roughly to the weighted mean of the data from Section 6.2.3 and the results given in De Graaff et al. (2017).

Box 3: Thermal oxidation materials & derivates from solid manure fractions

To derive the substitution potential of P-fertilisers containing STRUBIAS materials from solid manure fractions from the agricultural sector, the following assumptions and calculations were made:

1) Only poultry litter is considered as input material for thermal oxidation materials & derivates as no TRL 7-9 thermal oxidation processes from other solid manure fractions have been described that produce P-fertilisers.

2) In 2015, a total amount of 150 000-200 000 tonnes of poultry litter ash was produced, with an estimated P content of 14 kt P.

3) An annual growth rate in energy production from solid biomass of 7%, forecasted by the European Environment Agency (2016) for the period 2013-2020, has been assumed for poultry litter and extended until the year 2030. Accumulated over the time period 2015-2030, this would imply an increase of renewable energy production from poultry litter manure by a factor 2.76. Expressed on a P basis, the total P recovered as poultry litter ash would equal 39 kt P yr^{-1} for the year 2030.

4) The agronomic efficiency of poultry litter ash (RAE_{PUE}) relative to mined and synthetic fertilisers is assumed to be 0.90.

5) Considering 1) to 4), the estimated P-recovery from solid material from the agricultural sector is estimated at 35 kt of P.

Pyrolysis spectrum techniques

Gasification and pyrolysis can potentially convert a range of biomass types with high input-output efficiency and these are **therefore interesting technology options to convert residues and wastes that do not occur in large and geographically concentrated volumes** (Kretschmer et al., 2013). One of the advantages of pyrolysis spectrum techniques is that processing facilities can operate on a relatively small industrial scale, enabling the establishment of regional facilities to process high-volume, geographically dispersed materials, such as excess manure, without excessive transport costs. Moreover, the syngas generated during the pyrolysis process is mostly used as an energy source to dry the manure input material as pyrolysis processes of the raw input materials are often not autothermal. Starting from input materials that have not undergone a solid-liquid separation, this process pathway would therefore only marginally contribute to the production of renewable energy, but would principally be performed as a measure for pollution control.

Although no commercial thermochemical technologies using biomass have been identified in Europe at the time of writing, several commercial plants are close to piloting and operationalisation, both in Europe and rest of the world. The emerging technologies especially focus on **the treatment of the solid pig manure fraction**, obtained after solid-liquid separation treatment. This issue is particularly relevant given that highly diluted pig manure is associated with high transport costs, and most farmers have only a small surface area within which to dispose of the slurry produced (Foget et al., 2011).

The EU-funded **BioEcoSim** project (https://www.bioecosim.eu/) plans to process 2 000 000 m^3 of pig manure per year, resulting in a production of 62 kt of pyrolysis & gasification materials derived from pig manure (3 kt P, assuming a 5% P content in the resulting output material). The Japanese company **Hitachi Zosen Corporation** also aims to turn pig manure from pig farmers, in particular in the Netherlands and Belgium, into P-rich pyrolysis & gasification materials. The latter company has gained experience with the process from their operations in Japan. Other initiatives also exist and are emerging (e.g. Ecochar).

The **market outlook for this STRUBIAS pathway is associated with a large degree of uncertainty** as the resulting pyrolysis material is the only output material that will be brought onto the market. In contrast to anaerobic digestion techniques, no energy is typically recovered from pyrolysis and the increase in demand for renewable energy will thus not stimulate this STRUBIAS pathway. The economic valuation of pyrolysis & gasification materials is at present, however, largely unknown, as is the degree of consumer confidence associated with the end material due to the lack of long-term test results on agronomic efficiency for this material. Pyrolysis & gasification materials from solid manures have a P content of about 4-6%, organic C (~ 30-60%) and the presence of other nutrients in smaller quantities such as N, Mg and K. Therefore, the **nutrient density** of the resulting pyrolysis material is much **lower**, resulting in a higher land application cost for pyrolysis & gasification materials relative to more concentrated P-fertilisers. **It is assumed that the market potential of pyrolysis material in the conventional agricultural sector is indeterminate in the short term (before 2030)**. Given the lack of availability of concentrated P-fertilisers for organic farming, pyrolysis & gasification materials could potentially make an entry into the organic farming sector. **Due to the limited availability of P-dense fertilisers that meet the principles and requirements for organic farming, it is believed that this STRUBIAS material might be demanded and traded in this sector.** Here, the higher price setting of P-rich pyrolysis & gasification materials relative to mined rock phosphate and processed P-fertilisers could potentially be compensated by the higher sales prices of organically grown food products. Manure-derived pyrolysis & gasification materials could thus potentially replace current organic P-inputs such as manure, compost, meat and bone meal and meat and bone meal ashes in the organic farming sector.

7.3.5 STRUBIAS materials derived from slaughter residues

7.3.5.1 Introduction

Animal by-product activities covered under this section include the by-products obtained from the treatment of entire bodies or parts of animals at slaughterhouses. Rendering activities include the treatment of animal by-products both intended for and not intended for human consumption. The animal by-products industry handles all of the raw materials that are not directly destined for human consumption. **The use and disposal routes permitted are governed by Regulation (EC) No 1069/2009 laying down health rules as regards animal by-products and derived products not intended for human consumption**. Animal by-products are classified in three different categories, of which categories 2 and 3 can be used for the production of fertilisers. **The transformation of category 1 material into fertilisers is effectively prohibited by Regulation (EC) No 1069/2009, even after incineration, as any ashes produced should be disposed of as**

waste. The ban on the use of processed animal protein in feed for farmed animals has led to the diversification of the animal by-products industry.

The **rendering industry** processes most of the **animal by-products not intended for human consumption** as well as a significant fraction of the **catering waste**, which also has a legal status as an animal by-product (European Commission, 2005). There appears to be a trend towards fewer slaughterhouses with increasing average throughputs, favouring the **central collection of the slaughterhouse residues** for possible further processing to STRUBIAS materials (European Commission, 2005). In the year 2016, the rendering industry processed about 12.4 million tonnes of animal by-products of category 2 (0.8 million tonnes) and category 3 (11.7 million tonnes) (EFPRA, 2017).

7.3.5.2 Sector outlook for the rendering industry and material use

The total meat consumption in the EU is expected to remain stable within the EU for the period 2016-2026 (European Commission, 2016a). After the recent recovery, the EU-28 per capita consumption of meat products is expected to rise only slightly over the projection period, mainly as a result of the increase in meat consumption in the EU-13. However, developments in total meat consumption depend both on evolutions in per capita consumption and in total population numbers. The population in the EU-13 is projected to decline in the years to come at such a rate that the per capita increase in meat consumption is almost completely flattened out by the shrinking population, resulting in only 35 000 t additional meat consumed by 2026. In the EU-15, by contrast, the population is still increasing and therefore, with a stable per capita consumption, total meat consumption is expected to grow by 600 000 t by 2026. Combined, total EU meat production is expected to expand only slightly in the next 10 years, reaching 47.6 million tonnes. This can be mainly attributed to an increase in poultry meat production and to a lesser extent to pig meat production, while beef and veal production will decrease substantially. Production of poultry is expanding (+ 4.5% for the period 2016-2026, to a total value of about 15 million tonnes of meat produced), driven by a favourable domestic market. Pig meat production is expected to increase slightly (+0.1%, to a total value of about 23 million tonnes of meat produced), despite the environmental concerns. After a few years of increase, beef production is expected to return to its downward trend in the coming years (-8.6% for the period 2016-2026, to a total value of about 7.5 million tonnes of meat produced). By contrast, production of sheep and goat meat is likely to remain relatively stable after years of decline. **As EU consumption will not entirely absorb the moderate increase in production, the EU balance is stable due to the somewhat increased export prospects** (European Commission, 2016a).

Based on these meat production numbers, **a 2030 outlook of 11.5 million tonnes and 0.8 million tonnes of animal by-product processing is predicted for categories 3 and 2, respectively** (i.e. 98.5% of the volumes processed in the year 2016). The degreasing process then results in the production of ~ 2.9 million tonnes of protein-rich materials of categories 2 and 3 that contain the overall share of the P derived from animal bones, feathers, hairs, blood, etc. (Dobbelare, 2017). This material is known as meat and bone meal (category 2 materials) or processed animal proteins (PAP, category 3 materials). The P-concentrated bones (~ 10.5% P on a dry matter basis) contain the overall share of the P in the meat and bone meal, because the other animal fractions are characterised by a low P content (0.1-1.7%). Assuming an average P-content of 5.3% (Moller, 2015), **it is projected that the protein fraction of category 2 and 3 materials contains about 154 kt P yr^{-1}**.

At present, the meat and bone meal of **category 2 material is already largely used for the production of fertilisers, mostly as meat and bone meal without any significant further processing** (Dobbelare, 2017). Minor amounts of category 2 materials are sent to incineration (for energy recovery), landfill or composting. Its use in pet food, feed or food materials is not permitted according to the provisions of Regulation (EC) No 1069/2009. **category 3 materials are used to a lesser extent for the production of fertilisers (~ 17%). Most of the materials are used in pet food (~ 65-70%), fish feed (~ 10%) and fur animal feed (5-10%).** Less than 5% of the category 3 materials are used for the production of terrestrial animal food, human food (gelatin) or incinerated (Dobbelare, 2017).

The proteins derived from processed animal by-products are thus intensively used for the production of different materials. As a consequence of the many different competing uses, **meat and bone meal and processed animal proteins are traded at a price of about EUR 50-70 per tonne, or about EUR 943-1 320 per tonne P**. The bone fraction is even traded at a higher price, expressed on P-basis.

7.3.5.3 Thermochemical conversion processes

Meat and bone meal has a heating value ranging between 13 000 MJ and 30 000 MJ per tonne (Conesa et al., 2003). Therefore, thermal treatment by combustion, gasification or pyrolysis could potentially be used to generate energy.

Incineration

When incinerated, the ashes derive mostly from the bone component and contain high amounts of Ca and P, mainly hydroxyapatites and tricalcium phosphates. The combustion induces a wide range of structural modifications, reducing the P-solubility relative to the unburnt meat and bone meal (Moller, 2015).

Co-incineration of meat and bone meal is at present the foremost treatment for category 1 materials that should be disposed of. However, according to Article 32 of the Animal By-Products Regulation (EC No 1069/2009), category 1 derived-materials cannot be placed on the market as organic fertilisers or soil improvers.

The **fertiliser industry has already successfully tested the use of category 2 and 3 meat and bone meal ashes in the acidulation process** for the production of mineral P-fertilisers (Langeveld and Ten Wolde, 2013). Meat and bone meal ashes can effectively substitute phosphate rock in the process due to their high P content similar to phosphate rock, their consistency, their low Al and Fe content and their low levels of metals/metalloids (especially the Cd content is much lower than in phosphate rock). The chemical composition of the fertilisers (partly) produced from incinerated protein-based animal by-products is equal to that of fertilisers that are currently on the market (DAP, MAP, TSP, DCP, SSP, nitrophosphate, etc.; depending on the choice of the production process). In any case, the contaminant profile of the fertiliser end material will be different, with metals/metalloids – especially Cd – being present in lower contents in P-fertilisers derived from meat and bone meal ashes relative to most mineral P-fertilisers derived from phosphate rock (Kratz et al., 2016). Also, the environmental impact of producing fertiliser using these secondary phosphate sources suggests the emissions of phosphate and fluorine are lower than when using regular phosphate rock (Langeveld and Ten Wolde, 2013).

From an economic point of view, meat and bone meal and processed animal proteins of category 2 and 3 material (EUR 943-1 320 per tonne P, see Section 7.3.5.2) are more expensive than phosphate rock (on average EUR 649 per tonne P; see Section 9.1.3). However, in contrast to phosphate rock, the combustion of meat and bone meal and processed animal proteins will enable some energy recovery. **Thermochemical P-fertiliser production** processes such as the Mephrec process, the FEhS/Salzgitter process, and the thermo-reductive RecoPhos process (see Section 14.2.2) use meat and bone meal as an input material, **effectively reducing the energy demand of such processes**. Meat and bone meal is used as a fuel and as a P-source in the above-mentioned STRUBIAS production processes that produce phosphoric acid and ashes/slags that can be used directly as a P-fertiliser. Moreover, benefits for the protection of the environment and human health could be achieved relative to alternative scenarios (see Section 8.8). Nonetheless, mainly due to the strong competition for value-added category 2 and 3 animal by-products (feed industry, pet food industry, etc.), **the market potential for STRUBIAS materials produced via this pathway remains indeterminate**.

Pyrolysis of animal bone materials

Animal by-products of Categories 2 and 3 could also be processed via pyrolysis spectrum techniques, **resulting in a P-rich material that is embedded in a matrix of stabilised C**. At present, the only TRL 7-9 production pathway is the so-called 3R pyrolysis process, where animal bone materials are heated further up to 850 °C. During the pyrolysis process, all volatile and protein-based substances are removed from the mineral matrix, and a highly macro-porous apatite-type mineral material is produced, composed of hydroxyapatite (70-76%), $CaCO_3$ (7-13%) and carbon (9-11%), with a P content of 13% (30% P_2O_5) (3R AgroCarbon, 2016). The composition of the bone char material is, however, dependent on the production process applied; the bone chars produced as in Morshedizad et al. (2018) contained on average 60% hydroxyapatite, 26% dicalcium phosphate and 14% calcium phytate.

Similar to the market for pyrolysis & gasification materials derived from manure, the **market outlook for this STRUBIAS pathway is associated with a large degree of uncertainty** as the resulting pyrolysis material is the only output material that will be brought onto the market. The **economic valuation of pyrolysis & gasification materials is at present,**

however, largely unknown, as is the degree of consumer confidence associated with the end material. Expressed on a P-basis, the producers of pyrolysis & gasification materials claim a higher sales price for pyrolysis & gasification materials than for traditional P-fertilisers based on claims of high agronomic efficiencies (> 100%, expressed on a P-basis relative to mined rock phosphate and processed P-fertilisers) and additional benefits from increased soil fertility. Nonetheless, at present, scientific and experimental evidence is lacking that demonstrates the superiority of pyrolysis material from slaughter residues compared to traditional P-fertilisers in terms of fertiliser efficiency. Preliminary test results seem to point to a relative agronomic efficiency (RAE_{PUE}) that is, at a maximum, close to a value of 1 (thus equivalent to mined rock phosphate and processed P-fertilisers; see Section 6.2.4).

It is assumed that the market potential of animal-bone-derived pyrolysis material in the conventional agricultural sector is indeterminate in the short term, i.e. before the year 2030. Given the lack of availability of concentrated P-fertilisers for organic farming, pyrolysis & gasification materials could potentially make an entry into the organic farming sector. It is reiterated that the use of mined and synthetic fertilisers is heavily restricted under the existing legislation (Council Regulation (EC) No 834/2007 on organic production and labelling of organic products). Synthetic resources and inputs may only be permissible if there are no suitable alternatives. Such products, which must be scrutinised by the Commission and EU countries before authorisation, are listed in the annexes to the implementing regulation (Commission Regulation (EC) No. 889/2008).

Animal bone biochar would be the most P-dense fertiliser, having a P content similar to phosphate rock, but with an improved plant P-availability. Therefore, the higher price setting of animal bone biochar relative to mined rock phosphate and processed P-fertilisers could potentially be compensated by the higher sales prices of organically grown food products. Here, animal-bone-derived pyrolysis & gasification materials could potentially replace current organic P-inputs such as manure, compost, meat and bone meal and meat and bone meal ashes in the organic farming sector. **Given the limited availability of P-dense fertilisers that meet the principles and requirements for organic farming, it is believed that there might be a potential demand for animal-bone-derived pyrolysis & gasification materials in the organic farming sector.** A market entry in this sector may also enable further long-term product testing under realistic circumstances.

7.3.6 STRUBIAS materials from municipal wastewaters

7.3.6.1 Introduction

Historically, the P present in materials resulting from municipal wastewater treatment plants was largely returned to the agricultural field as sludge. Due to the physico-chemical processes involved in the wastewater treatment, the sludge tends to concentrate heavy metals and poorly biodegradable trace organic compounds as well as potentially pathogenic organisms (viruses, bacteria, etc.) present in wastewaters. Sludge is, however, rich in nutrients such as N and P and contains valuable organic matter that may prove useful when soils are depleted or subject to erosion. Sludge is usually treated before disposal or recycling in order to reduce its water content, its fermentation propensity or the presence of pathogens. Several treatment processes exist, such as anaerobic digestion, thickening, dewatering, stabilisation and disinfection, and thermal drying. The sludge may undergo one or several treatments. Once treated, **sludge can be recycled or disposed of using three to four main routes: recycling to agriculture (landspreading), composting, incineration or landfilling**.

The direct or indirect return of sewage sludge to agricultural land through these routes might be associated with emissions of pollutants into the soil, and indirect emissions into air and water. Other emissions into the air include exhaust gases from transportation and application vehicles. Although the EU Sludge Directive (86/278/EEC) enables the return of treated sludge, some **Member States have adopted national regulations with provisions that go beyond the requirements of the Directive** (see Section 7.3.6.2). **Therefore, the fraction of sewage sludge that is disposed of by incineration is increasing in Europe**. The disposal of sludge is considered by most stakeholders to have the disadvantage of wasting the fertilising value of the sludge.

STRUBIAS materials can be produced from nutrient-rich materials originating from wastewater treatment plants, and thus provide an option for the safe return of P present in sewage to agricultural land. Specifically, municipal wastewaters and sewage sludges are listed as eligible input materials for precipitated phosphate salts & derivates and thermal oxidation materials & derivates.

7.3.6.2 Policy and legal framework

European legislation

The legal framework established by the European Commission and regulating the various sludge routes is mainly composed of directives which have to be transposed into national legislation by Member States. The ones most relevant to sludge management are the following:

- Council Directive of 21 May 1991 concerning urban wastewater treatment **(91/271/EEC), known as the Urban Wastewater Treatment Directive**, is aimed at protecting the environment from the adverse effects of wastewater discharges. This Directive sets minimum sewage treatment standards to be achieved in stages by the end of 2005, and provides for advanced wastewater treatment for the removal of nitrogen and phosphorus from sensitive areas. Sensitive areas are defined as: areas particularly susceptible to eutrophication, surface waters intended for the abstraction of drinking water with high nitrate levels, and other waters that require a higher standard of treatment to satisfy the requirements of other directives. Directive 91/271/EEC supports the use of sewage sludge in Article 14: ‘sludge arising from wastewater treatment shall be re-used whenever appropriate’.
- Council **Directive 86/278/EEC on the protection of the environment when sewage sludge is used in agriculture**. This Directive sets minimum quality standards for the

soil and sludge used in agriculture, and defines monitoring requirements when sludge is spread on agricultural land. The limit values defined in this Directive concern heavy metal concentrations for sewage sludge as well as for soil when sewage sludge is used on land and maximum annual heavy metal loads through the application of sewage sludge.

- Sludge applications in agriculture must also comply with limits set by other EU legislation on nutrients in the environment, such as the **Nitrates Directive (91/676/EEC)** that limits the amounts of nitrates in water.
- The **Waste Framework Directive** (**Directive 2008/98/EC**) confirms the waste management hierarchy; preference has to be given to waste prevention followed by waste reduction, reuse, (nutrient) recycling, and energy recovery. This Directive establishes principles for the use and disposal of waste, waste management plans, approval procedures and monitoring.
- The Directive on the **landfill of waste (1999/31/EC)** restricts disposal of sludge to landfill, by gradually reducing the allowed quantities of biodegradable waste going to landfill and prohibiting the landfilling of both liquid and untreated wastes.
- **Directive 2010/75/EU on industrial emissions** sets limit values for emissions of pollutants to air from waste incineration.

Outlook for sewage sludge treatment and management options in Europe

The current trend in sewage sludge treatment is to reduce the amount of sewage sludge that is landfilled due to the increasing costs and increasing legislative restrictions on the landfilling of biodegradable waste (Milieu Ltd - WRc - RPA, 2010c). **This leaves two mid- to long-term options for the fate of sewage sludge: (1) landspreading – possibly after treatments such as anaerobic digestion and composting, and (2) incineration**.

The provisions of the Waste Framework Directive (Directive 2008/98/EC) oblige the Member States to expressly apply the **waste hierarchy** in the order of priority given in legislation and policy. In accordance with Article 4(2) of Directive 2008/98/EC, when applying the waste hierarchy, **Member States must take measures to encourage the options that deliver the best overall environmental outcome**.

There is scientific evidence that the **landspreading of sewage sludge to agricultural land can provide agronomic benefits,** in particular the recycling of plant nutrients such as N and P. Indeed, one of the most commonly recognised environmental benefits is the recycling of P in the food chain. This contributes to the conservation of P reserves and also reduces external inputs of Cd originating from phosphate rock. Sludge also provides other plant macronutrients, such as K and S, and micronutrients such as Cu and Zn. The beneficial effects of sludge application on soil organic matter status, structural properties and soil moisture retention are also well documented. **The accumulation and recycling of contaminants present in the sewage sludge is, however, a major concern when determining environmental impacts**, especially in population-dense regions where limited land is available for the spreading of sewage sludge close to wastewater treatment plants. The environmental impacts of the transportation of the high-volume sludge should also be considered. **Benefits in terms of climate change and greenhouse gas emissions** from recycled sewage sludge to agriculture are claimed, in particular that a portion of the C in sludge used in agriculture will be sequestered in the soil. However, to extent to which the long-term effect is **scientifically substantiated is unknown and, at present, national inventories of greenhouse gas emissions do not consider sequestered carbon from sludge used in agriculture (Milieu Ltd - WRc - RPA, 2010b).**

The environmental impacts associated with incineration are dependent on whether the sludge is combusted in mono- or co-incinerators. In simple terms, **co-incineration will score better on impacts related to global warming potential and energy recovery, whereas mono-incineration enables the potential recovery of P at the expense of a lower energy recovery potential**. The energy content of sludge is better used when it is co-fired at a coal-fired power plant compared to mono-incineration (de Ruijter et al., 2015). This is caused by a different design of the plants, where the advantage of a coal-fired plant comes from a larger scale and use of inputs with a lower corrosive effect. Ashes from a coal-fired plant, however, are not suitable for P recovery because of their low P content. Requirements for flue-gas cleaning are stricter for mono-incinerators compared to coal-fired power plants. The oven of a mono-incinerator is designed for a specific type of input and including another type of input would require another oven. This means that current co-incinerated inputs cannot simply be diverted to existing mono-incinerators.

The fact that recycling of nutrients is a higher priority than energy recovery in the waste hierarchy suggests **that mono-incineration followed by P-recovery from the ashes may be the preferred incineration option**. This observation is even more important given that P is a finite element, and phosphate rock is itemised on the EU list of critical raw materials. Moreover, **benefits for environmental protection and human health safety are assured** as thermal oxidation materials & derivates will have to comply with contaminant limits in the EU Fertilising Products Regulation ((EU) 2019/1009) that are stricter than the provisions of the Sludge Directive (86/278/EEC). The complex technologies and operational costs required to extract and produce materials from sewage sludge make this route, however, **less cost-efficient in comparison to landspreading**. The cost for landspreading of semi-solid or digested sludge is about EUR 126-185 per tonne sludge (dry matter basis), whereas the cost for the mono-incineration of sewage sludge is estimated at EUR 411 per tonne sludge (dry matter basis) (Milieu Ltd - WRc - RPA, 2010c). Mono-incineration of sewage sludge thus increases treatment costs with a factor 2.2-3.2. The supplementary cost is, however, only 3% of the total costs for wastewater treatment and disposal (Nattorp et al., 2017). Significant investments in mono-incinerators would be required in specific Member States. At present, only Austria, Belgium, Denmark, Germany, the Netherlands and the UK - all of them EU-13 countries - have operational dedicated sewage sludge mono-incinerators (European Commission, 2017b). A total of 43 mono-incinerators are present in Europe, of which the majority (23) is located in Germany (European Commission, 2017b). For cities and regions where infrastructure with mono-incinerators is available and P-rich ashes are already generated, **recovering P from ashes with the best available ash-processing techniques is**

associated with a relatively small cost (compared to ash disposal) if revenues from P-fertiliser sales are taken into consideration (see Section 8.10). This is exemplified by the agreement between EcoPhos and Dutch mono-incinerating facilities (60 000 tonnes of sewage ashes per year), indicating the commercial viability of transforming P-rich mono-incineration ashes into high-value P-sources such as pure DCP.

There is an ongoing debate on the best practices for the recycling of sewage sludge to the field, and **Member States continue to differ widely in the weighing of environmental, social and economic impacts associated with sewage sludge management** (Milieu Ltd - WRc - RPA, 2010b). The uncertainty related to future sewage sludge handling routes makes it highly challenging to provide an accurate estimate of the market for STRUBIAS materials derived from sewage sludge.

Germany and Austria have already taken a clear position and are currently transposing the provisions of the Waste Framework Directive into national legislation that **makes the recovery of P from mono-incinerated sewage sludge ashes mandatory**. According to the final version of the Bundes-Abfallwirtschaftsplan (Federal Waste Plan) in **Austria**, two options for P-recovery have been put forward, (i) mono-incineration of the sewage sludge and P-recovery from the resulting ashes, or (ii) at least 45% P recovery of the P present in the incoming municipal wastewater, if P is recovered from the digester supernatant and/or the sewage sludge. The draft Federal Waste Plan focuses on wastewater treatment plants with capacities of over 20 000 to 50 000 people equivalents (depending on the results of a feasibility study), within a transition phase of 10 years. In **Germany**, the new sewage sludge ordinance will make P recovery from sewage sludge obligatory for all German wastewater treatment plants with a capacity larger than 50 000 person equivalents (p.e.). They will have to recover 50% of the P from the sludge, if the sludge contains more than 2% P of the sludge dry matter content, or will have to achieve an end concentration of < 2% P of the sludge dry matter content. Otherwise the sludge with a high P load can be mono-incinerated, with the obligation to recover 80% of the P contained in the ashes. Land application of sludge will only be allowed under strict conditions for wastewater treatment plants < 50 000 p.e. The ~ 500 plants that are predicted to be subject to the P-recovery requirement represent roughly 66% of the total P removed from German wastewater and transferred into sludge.

The **Swedish government** has recently announced the launch of an inquiry aimed at introducing new laws to ensure that P can be recycled from sewage sludge in order to accelerate the country's transition towards a circular economy. Sweden's environment ministry also said the same inquiry will look at a proposal to ban the spreading of sewage sludge on farmland.

The **Swiss Ordinance on the Avoidance and Disposal of Waste** (VVEA) came into force in 2016. The recovery of P from phosphorus-rich waste, i.e. effluent sludge and animal and bone meal not recovered as feed, has become legally binding at ordinance level, with a 10-year transitional period. Given a 10-year grace period, P-recovery will have to be implemented by 2026. Current studies are ongoing to determine the on-the-ground implementation (e.g. minimal recovery rates).

The implementation of the **EU Directive on wastewater treatment and sludge handling** in the other Member States varies considerably. Certain Member States have implemented stricter provisions for the landspreading of sludge than those of Council Directive 86/278/EEC, especially in terms of the maximum allowed levels of heavy metals and organic compounds. **In particular, in countries such as Austria, Belgium, Denmark, Finland, France, Germany, Luxembourg, the Netherlands and Sweden, the limit values in national regulations are extremely stringent**, although use in agriculture at present still remains an important outlet for sludge in these countries. The landspreading of sewage sludge is already banned or highly restricted in the Netherlands and Belgium (Flanders), and – after a transition period – in Austria and Germany. The reasons for restricting landspreading relate to concerns on pollutants and the objective to reduce competition for the return of organic materials to the available land in the form of manure. **In the United Kingdom, Portugal, Spain, and the EU-13 countries, farmers' associations and authorities support the agricultural use of sludge, both for economic and for agronomic reasons** (mainly in terms of providing an economically feasible option for the return of organic matter and P to the agricultural field). Spain, for instance, has issues related to soil erosion and desertification, and so the recycling of sewage sludge to agricultural land is the preferred option, as indicated in the National Sewage Sludge Plan of wastewater treatment plants 2001-2006: 'As long as sewage sludge complies with legal requirements, including those which might be established in the future (...) it is considered that the most sustainable option is the recycling of nutrients and organic matter by agricultural land application' (Article 1.3.). Italy, Greece and Ireland have not taken a strong position on sludge recycling in agriculture, as far as can be judged from the available information.

STRUBIAS materials could be recovered in the form of precipitated phosphate salts & derivates and thermal oxidation materials & derivates from municipal wastewaters and the resulting sludges. The 2030 outlook for the latter could be estimated as follows:

- The total amounts of wastewaters that will be treated in the EU and sludges that will be generated are expected to increase in the future due to the increased implementation of the Urban Wastewater Treatment Directive (91/271/EEC). An annual increase of 13% in sewage sludge volumes has been estimated for the period 2010-2020 within the EU-28 (Milieu Ltd - WRc - RPA, 2010b). If this growth rate is extrapolated to the **year 2030, the total incoming P in municipal wastewater treatment plants would be estimated at 360 kt of P** (based on the sum of P in communal sewage sludge and urban wastewater treatment effluent for the year 2005; van Dijk et al., 2016).
- Milieu Ltd – WRc - RPA (2010b) indicated a decadal 18% increase in incinerated sludge for the period 2010-2020. When this growth rate is extrapolated to the year 2030, **the share of sludge disposal through incineration would equal 37%.** This value corresponds reasonably to the sum of the sewage sludge amounts that are currently already incinerated (27.3%) and landfilled (9.2%) (based on available data

from Eurostat averaged for the period 2010-2012; this period corresponds to the latest EU-28 representative reporting period). In line with the Waste Framework Directive provisions that prioritise nutrient recycling over energy recovery and the additional technical challenges to recover P from co-incinerated sludge, **it is assumed that mono-incineration is the single route for incineration.** Potentially, the high costs of investment and operation required for a mono-incinerator can be partly offset by the increased revenues from the exploitation of mono-incinerated ashes and higher gate fees for sewage sludge ashes. The possible non-compliance with the highly ambitious 100% mono-incineration assumption is presumed to be counteracted by sludge shifts from landspreading towards the incineration routes. The reduced acceptance for landspreading and possible stricter (national or EU) legislation on contaminant levels in landspread sewage sludge, and the possible synergies of incineration with energy recovery could drive shifts in sewage sludge management.

7.3.6.3 P-precipitation at municipal wastewater treatment plants

The Urban Wastewater Treatment Directive (91/271/EEC) requires more stringent treatment for discharges into sensitive areas (cf. Article 5 of the Directive). According to the latest summary report on urban wastewater treatment (European Commission, 2011), a total of 72.8% of EU-27 territory was identified as a sensitive area/catchment of sensitive area and therefore requires more stringent treatment. The territory of sensitive areas in the EU-28 increased by 4.8% compared to the last Implementation Summary of the year 2001 (i.e. 4.8% increase per decade). In 2011, tertiary treatment was in place for 89% of the load for the EU-15 and for 27% of the generated load for the EU-12 (European Commission, 2011). As the infrastructure in place cannot always achieve quality standards in line with the Directive’s requirements, 79% of the total generated load for the EU-15 and 24% of the total generated load for the EU-12 were reported to work adequately (European Commission, 2011). For the 2030 market estimate, it is assumed that the growth in sensitive areas will lead to an equivalent growth rate for wastewater treatments that rely on tertiary treatment (i.e. 4.8% per decade for the period 2010-2030). Therefore, **the total generated load for adequately operating wastewater treatment plants with more stringent treatment in the year 2030 is estimated at 83% and 26% for the EU-15 and the EU-13, respectively**. Given that ~ 75% of the disposed of sludge is generated in the EU-15 (Eurostat, 2017), **a weighted average for the EU-28 of the relative wastewater load connected to more stringent treatment is estimated at 69%.**

The most popular techniques to remove P from wastewaters at wastewater treatment plants that apply more stringent treatment are chemical phosphorus removal (Chem-P) using iron or aluminium salts and enhanced biological phosphorus removal (EBPR). The capacities of municipal wastewater purification in the EU have increased in recent years. Especially many new plants were constructed in medium-sized and small towns in eastern Europe. **Already existing plants in large cities were equipped with biological wastewater treatment and additional treatment technology.** Most EU-15 Member States have completed the development of their municipal wastewater treatment systems. Thus, maintenance and renewal investments dominate in countries such as Germany and the Netherlands as well as in Scandinavia. Although no exact data are available for the EU as a whole, Wilfert et al. indicated **that in western Europe, exclusive use of Chem-P is the dominant configuration for P removal in wastewater treatment (around 60%;** expressed in terms of sludge produced or people equivalents). The remaining 40% of plants rely on EBPR or EBPR with Chem-P support, configurations that would enable P-recovery as precipitated phosphate salts from the digested sludge or the sludge liquor (see Chapter 14).

Given the substantial operational benefits of the implementation of phosphate salt precipitation units for EBPR wastewater treatment plants, it is assumed that all EBPR plants will be equipped with such P-recovery units. **First-generation phosphate salt precipitation units** show a typical recovery of about 10-20%, and mainly involve the installation of the reactor in existing configurations (e.g. after the digester or after the decanter). **Second-generation P-recovery units** that are integrated in the wastewater treatment plant typically rely on additional treatment units in the sludge line prior to the anaerobic digester (e.g. WASSTRIP, thermal hydrolysis). Such processes lead to high P-recovery efficiencies of up to 50%, and provide substantial economic benefits through the reduction of sludge volumes and sludge dewatering capabilities. Due to the higher investment costs of second-generation P-recovery units, it is assumed that those would only be installed in wastewater treatment plants of agglomerations that treat > 150 000 people equivalents.

Box 4: Precipitated phosphate salts & derivates at EBPR wastewater treatment plants

To derive the substitution potential of P-fertilisers containing STRUBIAS materials via this route, the following assumptions and calculations were made:

1) For the year 2030, the total incoming P in municipal wastewater treatment plants is estimated at **360 kt of P**.

2) 69% of the EU-28 urban wastewaters will be processed with **more stringent treatment**. Struvites are produced at 100% of the wastewater treatment plants that apply more stringent treatment and rely on unique on EBPR, or on EBPR with Chem-P support. Such plants process 40% of the incoming P in municipal wastewaters. For the year 2030, the current **share of the plants with tertiary treatment that partly rely on EBPR configurations is maintained**. At present, there is no clear techno-scientific or economic evidence that justifies a radical change in wastewater treatment plant configurations, which is why only minor shifts can be expected in sludge that is treated in the year 2030.

3) Given 1) and 2), **EBPR wastewater treatment plants will treat about 99 kt P yr^{-1}.**

4) **First-generation P-precipitation** with an assumed P-recovery rate of 15% is applied to 60% of the incoming municipal wastewaters at EBPR plants (59.4 kt P yr^{-1}). **Second-generation** P-precipitation is set in place for the remaining 40% of the incoming municipal

wastewaters at EBPR plants (i.e. 39.6 kt P yr^{-1}), with an assumed P-recovery potential of 50%. Total P recovery is thus equal to 28.8 kt P yr^{-1}.

5) The **relative agronomic efficiency** (RAE_{PUE}) of precipitated phosphate salts & derivates relative to mined rock phosphate and processed P-fertilisers is assumed to be 1.05 (see Section 6.2.2).

6) Considering 1) to 5), **the P-recovery in the form of precipitated phosphate salts** at EBPR wastewater treatment plants **is estimated at 30 kt of P.** It is estimated that these materials will be brought onto the market directly as P-fertilisers or as part of a physical blend, and that their further processing into recovered phosphate salt derivates (e.g. MAP, DAP, TSP, nitrophosphate) is unlikely.

7.3.6.4 Sewage sludge mono-incineration

Egle et al. (2016) indicated the techno-economic **viability of P recycling from mono-incinerated sewage sludge ashes**. The P-recovery from Chem-P wastewater treatment configurations is somewhat more challenging than from EBPR sludges. Nonetheless, specific chemical and thermochemical processes can handle both ashes produced at EBPR and ChemP wastewater treatment plants (see Section 14.2.2). **Processes with a high TRL such as the acidulation process, the Ecophos process and thermo-reductive RecoPhos process (ICL) all produce inorganic intermediates (H_3PO_4, P_4) that can be used to produce conventional, mineral P-fertilisers (MAP, DAP, DCP, SSP, TSP, nitrophosphate, etc.)**. Given that these end materials are associated with a high degree of market and consumer readiness in the conventional sector, the manufacturing of such end materials is a highly probable route. Other routes that produce P-fertilisers of a different chemical composition are associated with a higher production cost and lower market readiness (lack of comprehensive testing on agronomic value, P present in less concentrated form resulting in higher transport and application costs, etc.). Therefore, it is unlikely that such materials will conquer a significant share of the conventional agricultural market, but minor volumes of such slow P-release fertilisers could possibly be traded in niche markets and/or organic farming sectors.

The bottleneck that will limit the volumes of recovered P-fertilisers derived from sewage sludge is the relative share of sludge that will be subject to (cost-intensive) mono-incineration.

As indicated in Section 7.3.6.2, **it is estimated that 37% of all generated sludge will be treated via mono-incineration followed by P-recovery in the year 2030.** It is assumed that downstream P-recovery from the ashes is performed on all incinerated ashes, regardless of any P already recovered in the form of precipitated salts at the wastewater treatment plant. After all, specific P recovery processes (e.g. EasyMining Ash2Phos) can recover P from sludge ashes with low P content.

Box 5: Thermal oxidation materials & derivates from mono-incinerated sewage sludge ashes

To derive the substitution potential of P-fertilisers containing STRUBIAS materials via this route, the following assumptions and calculations were made:

1) The calculations are based on the **implementation of the provisions of the Waste Framework Directive** (Directive 2008/98/EC) with P being effectively recovered from P-rich sewage sludge, either through landspreading of the sludge or through the return of STRUBIAS materials to the agricultural field.

2) The predicted P-content in sewage sludge corresponds to 95% of the P present in the incoming municipal wastewater. It is assumed that 37% of all generated sludge will be incinerated in the year 2030. The P that was already recovered upstream through first-generation P-precipitation in reactors integrated in wastewater treatment plants (Section 7.3.6.3) is subtracted from this figure to avoid double-counting. Hence, the total P in the sewage sludge ashes corresponds to (0.95 x 360 kt P yr^{-1}) x 0.37 – 29 kt P yr^{-1} = 98 kt P yr^{-1}.

4) The recovered P will be used to produce water-soluble mineral P-fertilisers that have, by definition, a **100% relative agronomic efficiency** relative to their mined counterparts.

5) Given 1) to 4), **a P-substitution potential of 98 kt of P yr^{-1} via this route is estimated with thermal oxidation material derivates (e.g. TSP, DAP) the end material of the production process**.

7.3.7 STRUBIAS materials derived from food processing industry wastewaters

According to the assessment of van Dijk et al. (2016), the P-losses from this sector were estimated at a moderate 44 kt P for the year 2005, of which about 8 kt P occurs as wastewater effluents.

Wastewaters from certain food processing industries are rich in P, either because of the properties of the source material (e.g. dairy industry, sugar beet industry, meat processing, fish and seafood processing) or due to the P-rich additives applied to the production processes (e.g. potato industry). At present, STRUBIAS materials are recovered as P-fertilisers from the potato industry and dairy industry as struvite, included in the STRUBIAS category ‘precipitated phosphate salts & derivates’. To the best of the authors’ knowledge, these are currently the only TRL 7-9 processes that manufacture P-fertilisers meeting the conditions outlined for PFC 1. Nutrient recovery processes from other residual wastewater streams from food processing industries have not been documented. The underlying reasons for this observation remain unknown, but may be associated with the

spatial scattering of the food processing facilities (i.e. lack of central collection possibilities for the wastewaters and sludges), **the chemical composition of the wastewater** (a phosphate content < 50 mg P L^{-1} making them unsuitable for precipitation as Ca or Mg phosphates), **the transfer to urban wastewater treatment plants**, or the **direct return to agricultural land of the uncontaminated sludges and wastewaters**. With the exceptions indicated for specific food industries, wastewaters from other food processing industries are mostly characterised by relatively low phosphate contents, so they are unsuitable for P-recovery in the form of precipitated phosphate salts & derivates. **Therefore, production processes in food processing industries, other than those from the dairy and potato industry, are not considered in this 2030 market assessment.**

The P-losses from the dairy industry are more difficult to estimate. The phosphate contents in the wastewaters vary largely between facilities, but most plants rely on the use of Fe coagulants in order to meet effluent quality requirements in line with the Water Framework Directive (Crittenden et al., 2008). **The suitability of the wastewater for P-recovery in a more plant-available form (e.g. struvites) relative to ChemP sludges will vary with wastewater characteristics across dairy plants.**

Box 6: Precipitated phosphate salts & derivates from the food processing industry

To derive the substitution potential of P-fertilisers containing STRUBIAS materials via this route, the following assumptions and calculations were made:

For the year 2030, there is a potential for P-recovery in the form of precipitated phosphate salts & derivates from **wastewaters from the potato industry and the dairy industry**. P-precipitation in other food processing industries is not considered due to a lack of current TRL 7-9 processes. Other STRUBIAS pathways are not considered as the thermochemical conversion of wastewaters from the food processing industry will not produce sludges that can be converted to P-fertilisers via incineration or pyrolysis.

A. Potato industry

At present, more than 8.3 million tonnes of potatoes are processed in Belgium and the Netherlands. In theory, 1 tonne of recovered phosphate salt can be precipitated as struvite from 500 tonnes of potatoes (Schuurmans Stekhoven, 2015), resulting **in a total maximal recovery potential of about 16.6 kt of struvite per year (~ 2 kt P yr^{-1})**.

B. Dairy industry

About 2-3% of the incoming milk is lost during cleaning operations, via washing steps and through occasional milk spills (Hach company, 2017). Typically, milk contains 1 kg of P per tonne. In 2015, about 150 million tonnes of milk were collected by dairies in Europe (Eurostat, 2017). By multiplying these numbers, the total P recovery potential from the dairy sector would equal 3-4.5 kt of P. On the assumption that the precipitation of recovered phosphate takes place with a recovery efficiency of 75%, **the maximal potential P-recovery from the dairy industry is about 2-3 kt of P**.

The increased monetary return of precipitated phosphate salts & derivates will provide a competitive advantage over chemical P-removal, at least for the largest facilities that process large amounts of wastewaters with a high P-content. Assuming that 25-75% of all wastewater treatment plants from the potato and dairy industry would rely on Ca and/or Mg salts for precipitation **results in a total P-recovery potential of 1-3 kt of P yr^{-1}, with a best estimate of 2 kt of P yr^{-1}**. Note that, at present, about one third (640 t P yr^{-1}) of this value is recovered.

7.3.8 Conclusion

7.3.8.1 Market estimate for TRL 7-9 processes under the anticipatable 2030 legislative framework

The estimates given in Sections 7.3.4 to 7.3.7 are subject to many uncertainties and as a result should be interpreted with the necessary caution. This is due to uncertainties regarding existing data sources, the future development in technology, the legislation in place for the year 2030, consumer confidence in STRUBIAS materials, the development of alternatives for dealing with eligible input materials, and the preliminary cost assessments. The results are based on the information gathered, including the responses from consultations, **and therefore represent the best estimate currently possible with the information available.** For selected STRUBIAS production processes, the JRC will evaluate costs and impacts on the environment and human health. These techno-economic analyses may serve as a basis to validate the information provided, and make the necessary corrections on market potential, where relevant.

Summing up the best estimates for the production of STRUBIAS materials from manure, municipal wastewaters, slaughter residues, the food processing industry**, a total of 213 kt of plant-available P could be recovered as precipitated phosphate salts & derivates and thermal oxidation materials & derivates** (Table 10). Although there could be a market for **pyrolysis & gasification materials**, the 2030 market potential **remains indeterminate,** mainly due to the lack of comprehensive test results on agronomic efficiency for this STRUBIAS group. Therefore, it remains unclear whether farmers are willing to pay for a material that is traded at prices that exceed those of mineral P-fertilisers, expressed on a monetary basis per unit of P present in the material. The estimates of production pathways for which the market potential remains indeterminate (pyrolysis & gasification materials; thermal oxidation materials & derivates from slaughter residues) have not been included in the total market potential for STRUBIAS materials.

STRUBIAS materials expected to be on the market in 2030 are **almost exclusively derived from municipal wastewaters (60%) and manure (39%)** (Table 10). Although significant progress has been made, the current handling of P in the sewage sludge and manure is not fully in line with the principles of sustainable use and recycling, because large P amounts from these sources are accumulating in soils, landfilled or removed from the biogeochemical P cycle in the form of construction materials (Schoumans et al., 2015). The applicable legislative framework and policy impacting upon water and soil quality are therefore strong determinants of the STRUBIAS market and trade potential.

It is estimated that **thermal oxidation materials & derivates will be the dominant STRUBIAS output material group with a total estimated production volume of 133 kt P yr^{-1}** or 62% of all STRUBIAS materials (Table 11). It is estimated that a major share of thermal oxidation materials & derivates will be brought onto the market in a **chemical form that is already available on the market** (DAP, MAP, SSP, TSP, nitrophosphate). Such materials may have a competitive advantage because of the high consumer confidence associated with these products in the conventional agricultural market. It is estimated that the market for **precipitated phosphate salts & derivates (80 kt P yr^{-1}**; 38% of the total plant-available P recovered as STRUBIAS) will also grow exponentially in the coming years (Table 11), mainly due to **process benefits at wastewater treatment plants that rely on enhanced biological P-removal techniques and synergies between P-recovery and renewable energy production in anaerobic digesters.**

For specific fertiliser materials, and especially those derived through pyrolysis processes, the organic farming sector is a potential trade market. The use of high-quality struvite and calcinated ashes for the organic farming sector has already been positively evaluated by the sector, and possibly more STRUBIAS materials can be authorised under the existing legislation on organic farming in the EU (Council Regulation (EC) No 834/2007 on organic production and labelling of organic products). In this market, materials are often traded at a higher price and few P-rich fertiliser alternatives are authorised in this sector. At present, about 6% of the utilisable agronomic area is destined for organic farming in the EU. A market entry into organic farming could enable comprehensive testing of the material quality, especially for agronomic efficiency. These results could potentially open an avenue towards expansion within the conventional agricultural sector, where STRUBIAS materials could replace and complement other P-sources of an organic nature. Additionally, smaller volumes of STRUBIAS materials could also be exploited in specific niche sectors, other than organic farming, such as the horticulture market and actors that highly value the circular economy framework.

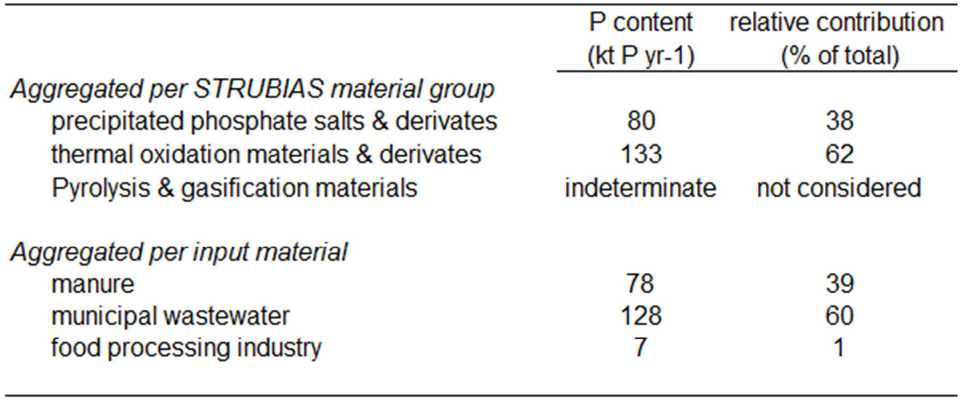
Most P-recovery in the form of STRUBIAS materials will take place in western Europe. The regional distribution of STRUBIAS production pathways is mainly a result of issues related to renewable energy production through digestates, high livestock densities that lead to P-excess in soils and possible issues related to water eutrophication, and reduced acceptance for the landspreading of sewage sludge (Table 10). With thermal oxidation materials & derivates as the dominant STRUBIAS pathways, EU Member States that have mono-incineration facilities also have a competitive advantage for STRUBIAS production (reduced infrastructure investments). Nonetheless, incineration ashes will also be shipped to facilities in other European regions (e.g. EcoPhos manufacturing site in Bulgaria).

Table 10: Market estimate for TRL 7-9 processes under the anticipatable 2030 legislative framework, the importance of the different processes across EU regions, and the major market drivers that will positively stimulate the market

process pathway	section	recovered STRUBIAS material	recovered total P (kt P yr^{-1})	relative agronomic efficiency (%)	recovered bio-available P (kt P yr^{-}	share of total bio-available P recovered (%)	regional distribution in Europe	major market drivers that will stimulate the market
P-precipitation after anaerobic digestion, dominantly from manure	7.3.4.3.	precipitated phosphate salts & derivates	45	105	48	23	livestock dense regions having soils showing P-excess and water bodies subject to P-eutrophication	reducing externalities due to manure excess (water quality), renewable energy targets.
thermal oxidation of solid manure fractions	7.3.4.4.	thermal oxidation materials	39	90	35	16	livestock dense regions having soils showing P-excess and water bodies subject to P-eutrophication	reducing externalities due to manure excess (water quality), renewable energy targets.
P-precipitation at municipal wastewater treatment plants	7.3.6.4.	precipitated phosphate salts & derivates	29	105	30	14	regions and cities that apply more stringent treatment at water treatment plants	increased share of biological waste water treatment plants.
thermal oxidation of sewage sludge	7.3.6.5.	thermal oxidation material derivates (e.g. DAP, MAP))	98	100	98	46	regions with low acceptance for sewage sludge landspreading and mono-incineration capacity	focus on increased soil protection from metal accumulation and nutrient excess, ban on landfilling of biodegradable waste.
P-precipitation from the food processing industry	7.3.7	precipitated phosphate salts & derivates	2	105	2	1	regions with dairy and potato processing industries	focus on externalities and costs associated to industry waste water discharges.
thermal oxidation of slaughter residues	7.3.5	thermal oxidation material derivates (e.g. DAP, MAP))	indeter-minate	indeter-minate	indeter-minate	indeter-minate	livestock dense regions	synergies with energy recovery from animal by-products
pyrolysis materials of solid manure fractions	7.3.4.4.	pyrolysis & gasification materials	indeter-minate	indeter-minate	indeter-minate	indeter-minate	livestock dense regions having soils showing P-excess and water bodies subject to P-eutrophication	reducing externalities due to manure excess (water quality), soil quality improvement, increased consumer and market acceptance.
pyrolysis of slaughter by-products	7.3.5.3.	pyrolysis & gasification materials	indeter-minate	indeter-minate	indeter-minate	indeter-minate	potentially somewhat more important in livestock dense regions	increased consumer and market acceptance.
OVERALL TOTAL			**213**		**213**	**100**		

NB: Values given are best possible estimates based on the information available and are subject to a high degree of uncertainty and as a result should only be interpreted as a rough approximation of the total estimates.

Table 11: Market estimate for TRL 7-9 processes under the anticipatable 2030 legislative framework as aggregated per STRUBIAS material group and per input material



	P content (kt P yr-1)	relative contribution (% of total)
Aggregated per STRUBIAS material group		
precipitated phosphate salts & derivates	80	38
thermal oxidation materials & derivates	133	62
Pyrolysis & gasification materials	indeterminate	not considered
Aggregated per input material		
manure	78	39
municipal wastewater	128	60
food processing industry	7	1

NB: Values given are best possible estimates based on the information available and are subject to a high degree of uncertainty and as a result should only be interpreted as a rough approximation of the total estimates.

7.3.8.2 Further market stimulations

The inclusion of STRUBIAS CMCs in the EU Fertilising Products Regulation ((EU) 2019/1009) could significantly contribute to increasing the sustainability in the EU agricultural sector and reducing negative externalities of production and consumption patterns. These topics are high on the political EU agenda that includes ambitious targets for waste recycling, water quality improvements, and reducing human exposure to contaminants. Although highly challenging to quantify, it is believed that shifting policy agenda targets towards 2030 plus further developments of specific TRL 6 processes could further stimulate the potential 2030 market for P-fertilisers derived from STRUBIAS materials. **Positive feedback loops induced by legislation could possibly further increase the production volumes of STRUBIAS materials, especially in the case of acceptable implementation costs.**

There is a continued political and public focus on externalities caused by the agricultural sector as well as on concerns on newly emerging pollutants (e.g. personal care products and pharmaceuticals). The recycling of P from manure and sewage sludge in the form of high-quality STRUBIAS materials can contribute to reducing nutrient leaching to water bodies and decreasing the accumulation of contaminants in soils **compared to reference scenarios of landspreading**. Especially in regions of nutrient excess, the **nutrient use efficiency** of STRUBIAS materials can be greater than for these organic sources because the nutrient release can be better synchronised with plant needs, thus reducing the scope for its loss to deeper soil layers and surrounding water bodies (see Section 8.9.2). Moreover, benefits are associated with the improved logistics for P-material storage, transport and handling, **improving the efficient return of nutrients to P-depleted soils and regions**. Finally, STRUBIAS production pathways for precipitated phosphate salts & derivates and thermal oxidation materials & derivates could effectively reduce organic and inorganic contaminants from the input materials, and thus **reduce the abundance of contaminants in the environment** relative to some of the currently applied business-as-usual scenarios (e.g. landspreading).

The scenarios below include examples of such further progress on legislative and policy measures that could stimulate the production volumes of STRUBIAS materials. Note that the second scenario is based on the breakthrough of promising technological developments for manure fractions; this assumption is reasonable as there are various TRL 6 processes that show a significant 2030 market potential.

1. As part of an ex-ante impact assessment, the European Commission has already investigated the impact of restricting the application of sewage sludge on soil (Milieu Ltd - WRc - RPA, 2010a). Any changes in Council Directive 86/278/EEC of 12 June 1986 on the protection of the environment, and in particular of the soil, when sewage sludge is used in agriculture will largely impact the STRUBIAS market potential. An option that has been investigated is to introduce, for instance, **more stringent standards on heavy metals in landspread sewage sludge** (option 3 – stricter limits on heavy metals as described in Milieu

Ltd - WRc - RPA, 2010c). Under such possible new provisions, 53% of all sewage sludge would have to be incinerated. Under such a scenario, the total P recovered from municipal wastewaters would increase from 98 kt P yr^{-1} to 156 kt yr^{-1} **(+ 58 kt P yr^{-1})**.

2. As indicated in Section 7.3.4, EU Member States should tackle the sources of pollution by fully implementing the Water Framework Directive measures and water-related legislation, especially the Nitrates Directive, Industrial Emissions Directive and Urban Wastewater Treatment Directive. In the present assessment, a total amount of 83 kt of recovered P is recovered from the agricultural sector, a number that is in line with the P that is lost to surrounding water bodies (81 kt of P yr^{-1}; van Dijk et al., 2016). However, about 924 kt P yr^{-1} also accumulates yearly in European soils (van Dijk et al., 2016), indicating the significant potential for P-recovery from this fraction. **An interesting TRL 6 STRUBIAS process is to apply a pretreatment to manure (e.g. moderate acidification, thermal hydrolysis technology, and others) to increase the phosphate content of the liquid digestate from which the recovered phosphate salt will be precipitated.** At present, such technologies are already applied on manure in Europe, although the extent of occurrence in the year 2010 was still limited (Foget et al., 2011). In 2010, only 1 377 t manure-P was acidified, whereas 665 t manure-P was subjected to thermal hydrolysis (Foget et al., 2011) but the combination of these techniques with P-precipitation has not yet been demonstrated in an operation environment.

Acids can be added prior to the digestion process to decrease the pH and shift the phosphate/total phosphorus equilibrium, and thus the P-recovery potential (up to 80% of the total P present). The thermal hydrolysis process is a high-pressure, high-temperature steam pretreatment application for anaerobic digestion feedstocks. The feedstock is heated and pressurised by steam within a reaction tank before being rapidly depressurised (flashed). This results in the breakdown of the cell structure within the biomass. As the organic matter is introduced into the digester in a broken-down condition, the digestion process is more effective, resulting in increased gas production and improved digestate quality (Pell Frischmann Consultants Ltd, 2012). As such, the P-recovery efficiency could be increased from 13% to 50%, simultaneously optimising the N/P ratio of the solid digestate fraction. In the event that the pretreatment process could be applied on manures that will be further processed by anaerobic digestion, an increase in the P-recovery efficiency of 13% to 50% would result in an additional recovery of **131 kt P yr^{-1}** (from 48 kt P yr^{-1} to 179 kt P yr^{-1}).

Another option to improve P recycling from manure is the use of manure incineration ashes as replacement for phosphate rock by the mineral fertiliser industry. The solid digestate fraction is combusted in order to achieve destruction of organic matter. If the moisture content within the digestate is sufficiently low and the incinerator efficiency is high, the process can become autothermal (the process generates sufficient heat to allow combustion to continue without the need for an external heat source or additional fuel) and energy recovery can be achieved (Pell Frischmann Consultants Ltd, 2012). Alternatively, the solid digestate fraction can be thermochemically treated in a rotary kiln by reaction addition (e.g. Si and/or Na_2SO_4; AshDec process) or pyrolysed. The char produced by the process can be used as a soil improver or as a partial replacement for peat in growing media production.

3. **Fish residues** in Europe and Norway are still a largely unexplored P-source. As outlined in Section 13.4.2, the P content of fish residues from catches and aquaculture for the EU-28 could amount to up to 27 kt P yr^{-1}. Moreover, the fish catches and aquaculture production in Norway produce about 45% of the total fish residues in Europe. Fish excreta and undigested feed from land-based aquaculture also form a P-source that can potentially be recovered as STRUBIAS materials. The STRUBIAS subgroup indicated that these residues are already used for the production of P-fertilisers in Norway. Given that the characteristics of the protein fraction of fish residues are similar to those of terrestrial animal by-products, it is believed that fish residues could further increase the P-recovery potential in an indeterminate manner (up to 40 kt P yr^{-1}).

7.3.8.3 Substitution effect

Results from the previous section will be used to estimate **the substitution effect of mined and synthetic fertilisers by fertilising products containing precipitated phosphate salts & derivates, pyrolysis & gasification materials and thermal oxidation materials & derivates for the year 2030**. This is based on the opening of the EU market, taking into account existing feedstock, the expected recovery rate and the equivalence of the fertilising features of recovered nutrients compared to those of mined and synthetic inorganic fertilisers.

By summing the values given in Section 7.3.8.1, it is estimated that **a total of 213 kt bio-available P yr^{-1} could be recovered**. In the event that the further **market stimulations for sewage sludge and manure outlined in Section 7.3.8.2 were to materialise, an additional 189 kt P yr^{-1}** could be recovered in the EU-28 as P-fertilisers.

The 2030 apparent consumption of mined rock phosphate and processed P-fertilisers is estimated at 1 220 kt P yr^{-1} (see Section 7.2.3.2). To avoid double-counting during the calculation of the substitution effect, it is necessary to check the alternative fate and treatment scenario of the STRUBIAS input materials (manure, sewage sludge, wastes from food processing). Any bio-available P derived from organic sources that currently contributes to plant P uptake, but is projected to be used as an input material for STRUBIAS production processes, should thus be corrected for. It should be added to the estimated 1 220 kt bio-available P yr^{-1} that is applied as P-fertilisers on land. The following assessment is made for each of the pathways:

i. *Manure slurries to be used as an input material to produce K-struvite (Section 7.3.4.3)*

   These manure fractions are typically characterised by a low nutrient density and an N:P ratio that is often imbalanced relative to plant demands. Therefore, these materials are (i) subject to nutrient removal (e.g. aeration to remove N, P-removal using salts; cfr. Process *Stichting Mestverwerking Gelderland - SMG*), or applied on land in regions characterised by nutrient excess, thus leading to a large accumulation of P in agricultural soils, especially in livestock-dense regions (van Dijk et al., 2016). Under such conditions, the mineral P substitution efficiency of these manure fractions is below the relative share of P that is recovered from the manure as precipitated phosphate salts (see Section 8.6.2). Hence, the fraction recovered is only the P that is supplied in excess relative to plant P demands. Therefore, the added value of these manures as a P fertiliser and their contribution to plant P nutrition is at this moment negligible, and **the newly produced STRUBIAS material will thus not replace present-day plant nutrients.**

ii. *Poultry litter combusted to poultry litter ashes (Section 7.3.4.4)*

At present, most of the excess poultry litter is subject to hygienisation and drying in order to enable transport over relatively large distances and its application as a P-fertiliser in agricultural areas characterised by soils that are poorer in P. Therefore, a correction should be made to avoid double-counting. Assuming that poultry litter has a relative agronomic efficiency of 0.85 relative to mined rock phosphate and processed P-fertilisers (Eghball et al., 2002; see Section 9), the **total fertiliser P-demand for the year 2030 is estimated to increase by 33 kt P yr^{-1} (39 kt P yr^{-1} x 0.85).**

iii. *Effluents and sludges from municipal wastewater treatment plants as input materials for P-precipitation and thermal oxidation processes (Section 7.3.7)*

As outlined in Section 7.3.8.1, these processes will mostly take place in countries in western Europe (the Netherlands, Belgium, Germany, Austria, etc.) where the land application of sewage sludge is, anyhow, largely restricted (see Section 13.5). Moreover, the budgeted sludge fraction that is considered in the thermal oxidation pathway (39%) corresponds reasonably to the sum of the sewage sludge amounts that are currently already incinerated (27.3%) and landfilled (9.2%). Therefore, no correction for double-counting is required for the processes listed in Table 10. However, in the event that further market stimulations were to materialise, resulting in an increased amount of sewage sludge being incinerated (see Section 7.3.8.2), the supplementary P-recovery would effectively take place on sewage sludge that is currently spread on land, possibly after anaerobic digestion or composting. Therefore, a correction should be made to avoid double-counting. The agronomic efficiency of the P in sewage sludge relative to mined rock phosphate and processed P-fertilisers is estimated at 0.46 (Oenema et al., 2012). **Therefore, the total fertiliser P-demand for the year 2030 under this scenario of further market stimulations is estimated to increase by 27 kt P yr^{-1} (58 kt P yr^{-1} x 0.46).**

iv. *Wastewaters and wastes from the food processing industry to be used as an input material for precipitated phosphate salts (Section 7.3.7)*

These input materials consist of wastewater from the potato industry and dairy industry. At present, these materials are not returned to agricultural land, but rather subject to chemical precipitation processes and removal of the flocculant-rich sludges from the biogeochemical P cycle. Therefore, no correction for double-counting is required for this process pathway.

Taking into consideration the points i) to iv), it is estimated that the total demand for P-fertilisers in the year 2030 will be 1 253 kt P yr^{-1} when considering the processes listed in

Table 10. If the further STRUBIAS market stimulations outlined in Section 7.3.8.2 were to materialise, the total demand would increase to 1 280 kt P yr^{-1}.

In summary, as a best estimate, the opening of the P-fertiliser market to STRUBIAS materials will result in a substitution effect of mined and synthetic fertilisers by fertilising products containing precipitated phosphate salts & derivates and thermal oxidation materials & derivates of 17-31%. Moreover, it should be noted that pyrolysis & gasification materials have not been included in this assessment because an initial entry is considered more likely in the organic farming sector. The on-the-ground use of pyrolysis & gasification materials might enable the detailed testing of the agronomic efficiency of pyrolysis & gasification materials, possibly leading to more refined and precise estimates given in Section 6.2.4. In the event that these results lead to increased market and consumer confidence for the use of these materials in conventional agriculture, the substitution potential will further increase. It is noted that these estimates are associated with a large degree of uncertainty as many STRUBIAS materials are mostly a co-product of a process that has a different primary aim (e.g. waste management and control, biogas production, manure hygienisation). STRUBIAS materials mostly fit in transformation cascades of biogenic materials. On many occasions, STRUBIAS materials are only a co-product of these processes, and it will be the main driver for the process that determines the extent to which STRUBIAS materials are placed on the market.

7.4 Market outlook for non-fertiliser PFCs derived from STRUBIAS materials for the year 2030

STRUBIAS materials – and more specifically thermal oxidation materials & derivates and pyrolysis & gasification materials – can also be used in PFCs other than PFC 1 – fertilisers. STRUBIAS materials could be applied as liming materials, soil improvers, plant biostimulants, or as part of a fertiliser blend. Three main types of such STRUBIAS materials were indicated by the STRUBIAS subgroup: pulp biomass ashes, slags from the iron and steel industry, and C-rich char-like materials. Nonetheless, possibly more materials could be produced as the proposed STRUBIAS recovery rules enable the use of a broad range of input materials and production process conditions. Estimating a 2030 market for such fertilising materials is extremely challenging and associated with large uncertainties. Biomass ashes and slags from the iron and steel industry are by-products from production processes focusing on a different primary product (energy, paper and cardboard, and iron and steel, respectively). Therefore, the market for these materials will be determined by the economic aspects and policy decisions taken in the coming years.

Bark, wood and pulp residues from wood handling can be combusted for energy recovery. The ashes contain nutrients and have a neutralising value and can thus be suitable as a (forest) fertiliser or liming material as long as the wood-originated metals meet harmonised requirements for fertilisers. Wood and pulp biomass is used for energy production in many EU Member States, especially in northern Europe. Wood and biomass combustion in Denmark, Finland and Sweden generates > 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). The market outlook for the near future for these materials remains uncertain, amongst others, due to the decisions to be taken in the framework of renewable energy.

Slags are produced as by-products from the metallurgical industry through a variety of techniques (see Section 13.8). Due to their high Ca and sometimes relatively high P contents, slags from the iron and steel industry are currently applied on land as liming materials or fertilisers. The ratios of slag to steel output indicate that large volumes of slags are produced. Manufacturing a tonne of pig iron produces, for instance, 0.25-0.30 tonnes of blast furnace slag that can be further processed to granulated blast furnace slag (Jewell and Kimball, 2014). The output mass of basic oxygen furnace slag per tonne of crude steel is 0.10-0.15 tonnes (Jewell and Kimball, 2014). Altogether, approximately **25-40 million tonnes of** slags by the steel industry are produced annually (Branca et al., 2014; European Commission, 2016b).
Iron slag and steel slag are used primarily as aggregates in concrete, asphaltic paving, fill, and road bases. Slags can also be used as a feed for cement kilns. About 2-3% of the slags, representing > 800 000 tonnes, are used for the production of **fertilising products** (European Commission, 2016b). In Germany, slag use as fertilisers and liming materials has a long tradition; about half a million tonnes of steel industry lime are used in agriculture. Blast furnace slag, basic oxygen furnace slag and secondary metallurgical ladle slag from the production of low-alloyed steel are used as fertilising materials. **Based on preliminary evaluations and confidential data submitted by the STRUBIAS subgroup, the overall share of the blast furnace slag and secondary metallurgical slag have relatively low metals levels and would be able to meet the proposed criteria for CMC for thermal oxidation materials & derivates.** Slag can be used as a liming material or P-fertiliser production and micronutrients supply. The marketing of these products is today a reality in Germany, Austria, Finland, Sweden, France and probably other Member States. Basic slags are an effective liming material, having a high of acid-neutralising value (25-30%) and some Mg. Due to their low P contents (~ 0.01-2%; European Commission, 2017a), steel industry slags today make a low to moderate contribution to European agriculture. The steel industry is, however, currently testing processes that aim to increase the quality and the P-content of the output materials to enable their use as an added-value quality fertilising material (e.g. FEhS/Salzgitter process; see section 14.2.2).

C-rich char-like materials produced through a wet or dry pyrolysis process from biomass (e.g. plant materials, food processing residues, bio-waste) show a relatively high production cost that might limit the open-field applications at large application rates. Nonetheless, the STRUBIAS subgroup indicated their potential to be used as part of soil-less plant-growing media in horticulture, where they could possibly replace or complement other substrates like peat and lignite (Gruda, 2012). The growing media industry in the EU has a EUR 1.3 billion turnover for peat-based substrates. It is particularly important in Germany, Italy, the Netherlands, France and the UK. A smaller market segment is represented by lignite, which is largely produced in Germany and Poland, and consumed for fertiliser production, or directly as growing media in horticulture. It has been estimated that, among the 22 million m^3 of soil conditioning product consumed for professional horticulture, 86% is peat. Also, for gardening 15 million m^3 of soil improvers and growing media are consumed every year, 69% of which is peat. The global lignite consumption in the EU is around 450 Mt per year, of which 7% is used in agriculture. Most of the lignite in the agriculture sector is used to produce synthetic fertiliser (anhydrous ammonia and ammonium sulphate), but more than 200 kt per year are used directly as soil improver or growing media (EUBIA, 2015). The reasons for using pure peat or using it as a main component of growing media are to be found in its availability in northern Europe, its relatively low cost and its good chemical, biological and physical properties (Barrett et al., 2016). Increasing concern over the environmental impacts of some commonly used materials has led researchers to identify and assess more environmentally sound alternatives. There has been an understandable focus on pyrolysis & gasification materials that have shown promise at an experimental level, but at present few have been taken up on a significant scale (Barrett et al., 2016). Further market uptake will depend on the interaction between economic and environmental concerns for the sector and the performance of pyrolysis & gasification materials relative to the current materials used, including not only peat but also other alternatives such as compost, coir, softwood bark, wood fibre and wool.

7.5 Conclusions on market potential for STRUBIAS materials

Based on the preliminary market assessment, market demand and trade are expected for all three STRUBIAS material groups in different segments of the EU agricultural sector. The most important share of the STRUBIAS materials will be used as fertiliser that can be used to provide nutrient, mostly phosphorus, inputs to European agriculture. Some STRUBIAS materials could also serve other fertilising functions and include uses as liming materials, soil improvers or growing media.

8 Life cycle analyses - environmental and human health impacts and production cost

The protection of human health and the environment throughout the production and use phase of STRUBIAS materials should be ensured through (1) the implementation of **the technical requirements for STRUBIAS materials and EU fertilising products** that contain such materials, and (2) any **other relevant EU and national legislation that applies to these materials**, such as the Industrial Emissions Directive (2010/75/EC), the Water Framework Directive (2000/60/EC) or the Nitrates Directive (91/676/EEC) (see Section 5). This section will assess the impacts on the environment (e.g. eutrophication, global warming), human health (e.g. carcinogenic and other toxic effects), and the cost of production. The impacts and production costs for different feedstock-process technology combinations for P-recovery through STRUBIAS pathways is evaluated based on the information collected from front-runner operating P-recovery facilities using a standardised and comprehensive life cycle methodology.

Important note

The **results presented here are dependent on the technological design and development scale for specific STRUBIAS production pathways**. The results presented in this section should be interpreted as **preliminary estimates for a P-recovery industry that is still in the initial stages of development**. Hence, the results can by no means be interpreted as final conclusions; it is a **preliminary evaluation for an emerging and continuously developing industrial sector of the circular economy**.

The analysis is oriented towards **supporting policy development at the EU level,** and therefore **considers** – in addition to the manufacturing stage of the STRUBIAS material – **life cycle stages upstream and downstream of the P-recovery operator that fall beyond their direct control.** Moreover, **the analysis does not take into consideration situation-specific aspects** related to soil nutrient status, the (national) legal framework on agricultural management and water and soil quality, resulting impacts on the business-as-usual handling of biogenic materials and type of displaced nutrient sources, regional variations in soil and climate properties, plant nutrient demands, transport distances, energy mixes, etc. To some extent, this issue has been addressed through supplementary variant analyses where key parameters (transport, energy mix, and plant P demand) have been varied (see Section 8.11), but – even so – such an analysis is unable to take into account the EU-wide variation in key parameters that determine the final impacts.

For all these reasons, the **results presented here are <u>an assessment of specific case scenarios that may not be representative of the situation-specific boundary conditions in which specific P-recovery facilities operate</u>**. Therefore, **the results presented here can, by no means, be used to evaluate the impacts of a specific P-recovery operator**. As a matter of fact, the results presented highlight that the impacts are largely dependent on decisions that fall *beyond* the direct control of the operator. Rather than focusing on the impact of specific operators, this analysis aims at putting forward opportunities for P-recovery and mechanisms that determine the final environmental and societal impacts.

Additional to the assessment provided here, the JRC will publish **a peer-reviewed scientific manusript** that builds further upon this chapter and relies on the same foundations (e.g. choice of functional unit, accounting for the displacement effects that the STRUBIAS materials may have on current-day feedstock and management). For selected pathways, a more detailed study will be presented that includes, amongst others, supplementary impact categories, a state-of-the-art assessment of external costs for nutrient-dense regions, and a full inventory of the inventory data underlying the LCA. This study will be published as a separate paper (Tonini D., Saveyn H., and Huygens, D.; Environmental and health co-benefits for advanced phosphorus recovery; in press) because the work extended beyond the deadline for concluding this report (December 2018).

8.1 Objective of the life cycle assessment

Phosphorus is essential for life and irreplaceable. It is a key element in our DNA and all living organisms require P intake to produce energy. A reduced dependence on the finite supply of phosphate rock is a huge benefit for society. Whereas other non-renewable natural resources can be replaced by other primary materials when they deplete, phosphate rock has no substitute from primary sources. Achieving long-term food security means we must change the way we source P in global food production. A core aim of STRUBIAS technologies that repackage dissipated P into concentrated P-fertilisers is to decouple end P users from source risk. **This assessment departs from the assumption that P-recovery is a function of biomass transformation cascades, and that future management systems for biogenic materials will require sustainable P management to reduce the dependency of phosphate rock as a primary raw material.** This assumption is in line with recent legislation in specific EU Member States (e.g. mandatory P-recovery from wastewater treatment plants in Austria and Germany), and national and EU policies related to agriculture and water quality management.

The occurrence of meaningful shifts in P-fertiliser manufacturing processes thanks to the opening up of the fertiliser market to STRUBIAS materials might considerably alter the environmental and human health impacts, both ‘upstream’ and ‘downstream’ of a P-fertiliser manufacturer or supplier. Conceptual frameworks often refer to possible environmental and monetary co-benefits of the circular economy model, including avoided energy- and cost-intensive transport of manure, mitigated eutrophication, economic savings due to energy recovery, and reduced waste management costs (Elser and Bennett, 2011; Mehta et al., 2015; Macdonald et al., 2016; Mayer et al., 2016). Nonetheless, case studies assessing advanced P-recovery through STRUBIAS processes from municipal wastewaters (Bradford-Hartke et al., 2015; Amann et al., 2018) and food waste (Styles et al., 2018) in Europe identified **trade-offs between environmental impacts and suggested that any environmental and human health savings or burdens are largely dependent on the technology applied and the**

resulting fertilising material properties. Economic analyses indicate supplementary costs for most, but not all, P-recovery pathways from municipal wastewaters (Egle et al., 2016; Nattorp et al., 2017).

For this assessment, **the focus is thus on the plant-available P as a product to be used by farmers**. The life cycle assessment encompasses the whole life cycle of the final P-product, from transport, to processing, to final use on land. This also includes the effects associated with possible co-services and co-products generated throughout the manufacturing chain. By incorporating information on the capital and operation costs for the different life cycle stages, an assessment of the production and opportunity cost for STRUBIAS materials will be brought forward.

This assessment evaluates the possible impacts, related to the implementation of P-recovery pathways versus current counterfactual scenarios for the handling of biogenic materials, on phosphate rock depletion, global warming potential, freshwater P eutrophication and human toxicity. **The objective of this section is to identify development opportunities, sector challenges and mechanisms that maximise the socio-environmental benefits of emerging P-recovery pathways.** This information may help to guide nascent manufacturers in the optimisation of innovative and sustainable production processes adapted to local settings and regional priorities, and aid policymakers to assess the sustainability of particular routes to reduce dependence on phosphate rock, in order to make informed decisions. Rather than focusing on the numerical results of each of the individual STRUBIAS pathways, this section aims to (i) provide insight into the way impacts and costs for STRUBIAS production pathways are properly envisaged and conceptualised, and (ii) focus on the main drivers of the impacts and costs for STRUBIAS processes.

8.2 Standardised methodology for STRUBIAS materials

In order to make a sound judgement on the technical, economic, market, environmental and human health aspects related to the production and use of STRUBIAS materials, **a standardised and uniform scientific analysis needs to be performed across all STRUBIAS material categories**. Whereas STRUBIAS materials may have already been the subject of individual studies encountered in the scientific literature (Jossa and Remy, 2015; Egle et al., 2016; De Graaff et al., 2017; Nattorp et al., 2017; Styles et al., 2018), these studies (i) feature their own goals, scope, system boundaries, and datasets, and (ii) only make an assessment for specific input material – STRUBIAS process pathway combinations. The result is a **fragmented information landscape**, which currently complicates a comparison of STRUBIAS materials. Therefore, the JRC has decided to perform an independent assessment that relies on a standardised methodological approach (environmental assessment based on the ISO 14040:2006 *Standard*) and state-of-the-art methodology (for monetary evaluation).

8.3 Conceptualisation of life cycle systems

For the reader to understand this section of the report well, it is fundamental to outline some basic aspects and principles of the life cycle methodology applied to assess human and environmental impacts and costs of STRUBIAS materials. Fertilising products can be manufactured from either primary or secondary raw materials, with both types of processes resulting in products with a similar or identical function that will be placed on the market. From a conceptual point of view for life cycle analyses, both types of processes are, however, fundamentally different.

The starting point for phosphate-rock-derived P-fertilisers is constituted by the extraction of a primary raw material. **If phosphate rock is not extracted from nature, it will remain stored in natural bedrock layers**; it is assumed to be an inert material and adequately protected from exposure such that the environmental impact from the natural weathering of the phosphate rock is negligible. The situation is different for P-fertilisers derived from secondary raw materials as the latter materials are already part of the biogeochemical processes occurring on earth, and further processing and/or handling of these biogenic materials is required anyway. Biogenic secondary raw materials cannot be withdrawn from the biogeochemical cycles, unless they are further processed into materials that can be deposited in the long run; in reasonable present-day conditions and under existing and future EU policies and legislations, biogenic materials cannot simply be landfilled. **A direct consequence of the handling of the feedstock material through a STRUBIAS production pathway is that an alternative handling scenario of the (biogenic) feedstock material will be avoided.** This observation indicates that a proper analysis will have to consider the impacts of the ***counterfactual*** **life cycle of the secondary raw materials**, as well as the costs that are associated with the handling of secondary raw materials in a counterfactual scenario. It is thus essential that a relevant counterfactual scenario is included in the assessments as any handling scenario of the secondary raw material is associated with an environmental impact and a cost, and those impacts and costs are replaced by others when that secondary raw material enters the transformation cycle that ends up with a STRUBIAS material. Relevant examples of counterfactual life stages of secondary raw materials are, for instance, the co-incineration of sewage sludge followed by transport to a landfill, or the processing (e.g. anaerobic digestion), storage (e.g. in storage tanks at the farm), and transport (over variable distances, depending on the regional situation) of pig manure. This implies, for example, that the production cost for SSP manufactured from sewage sludge should be reduced by the cost of the pertinent alternative treatment (e.g. the co-incineration of sewage sludge), and that the environmental impacts of producing struvite from manure should be decreased by the impact of handling manure under the counterfactual handling scenario (e.g. energy recovery in anaerobic digestion, emissions from transport, and possible eutrophication resulting from the use of digested manure on land). **This observation points towards (1) the importance of selecting relevant counterfactual scenarios in the life cycle analyses, and (2) the fact that impacts and 'production costs' for STRUBIAS fertilisers are dependent on the situation-specific context** (e.g. direct land application of manure on the farm may take place in Croatia, but the same manure should be transported over long distances to enable land application in line with the legislative framework in the Netherlands for example).

Secondary raw materials of biogenic origin typically include manifold valuable substances (e.g. carbon, nitrogen, energy) and contaminants (e.g. metals). Nutrient recovery may even be part of a production process that has a different aim (e.g. biogas production, processed manure with a balanced nutrient supply for specific agricultural lands). In the circular economy, in the bio-economy and in many STRUBIAS P-fertiliser production processes, different co-products are generated that can also be placed on the market as added-value materials, and thus replace other products. These **co-products have to be taken into consideration for both the STRUBIAS scenarios as well as for the counterfactual scenarios for the handling of secondary raw materials**. This is exemplified by a STRUBIAS scenario where poultry litter ash is produced from poultry manure, and a counterfactual scenario where the manure is directly applied on land. In such a case, (i) the sales price of the energy recovered from the incineration of poultry manure will be subtracted from the gross production cost of the STRUBIAS material, and (ii) the STRUBIAS material production cost should be increased because plant-available nitrogen is returned to land in the counterfactual scenario and thus the cost of purchasing mineral N is avoided; N is, however, lost during the thermal oxidation process. Similarly, the environmental footprint of STRUBIAS materials derived from manure can be decreased when in the counterfactual scenario manure might be associated with N leaching towards water bodies, but be increased when disproportionate transportation of the manure to the STRUBIAS production facility is required relative to a counterfactual scenario of on-farm manure landspreading. Also, here the situation is thus different for STRUBIAS to mined phosphate rock and processed P-fertilisers where the manufacturing aims at manufacturing a single product, i.e. the P-fertiliser.

All these observations already indicate that reporting on the absolute environmental and human health impact from the production and use of STRUBIAS materials provides little added value. The environmental and human health impacts originate from different life cycle stages for STRUBIAS production and use on land, including the avoided emissions from the alternative use of the input material, the manufacturing stage, and the use-on-land phase. Therefore, the present STRUBIAS report will **rely on scenario modelling for the assessment of environmental and human health impacts and estimates for production costs of STRUBIAS pathways**.

It is indicated that a consequential life cycle approach is the most suitable conceptual approach in line with ISO *Standard* 14040:2006 because the **handling of the feedstock material and the system co-products is expected to change as a direct consequence of the manufacturing of STRUBIAS fertilisers**. In view of maintaining the basic principle of mass and energy conservation and economic accounting in the life cycle system, the life cycle approach therefore requires the life cycle system to be expanded to include the impacts and costs related to the handling of feedstock and system co-products.

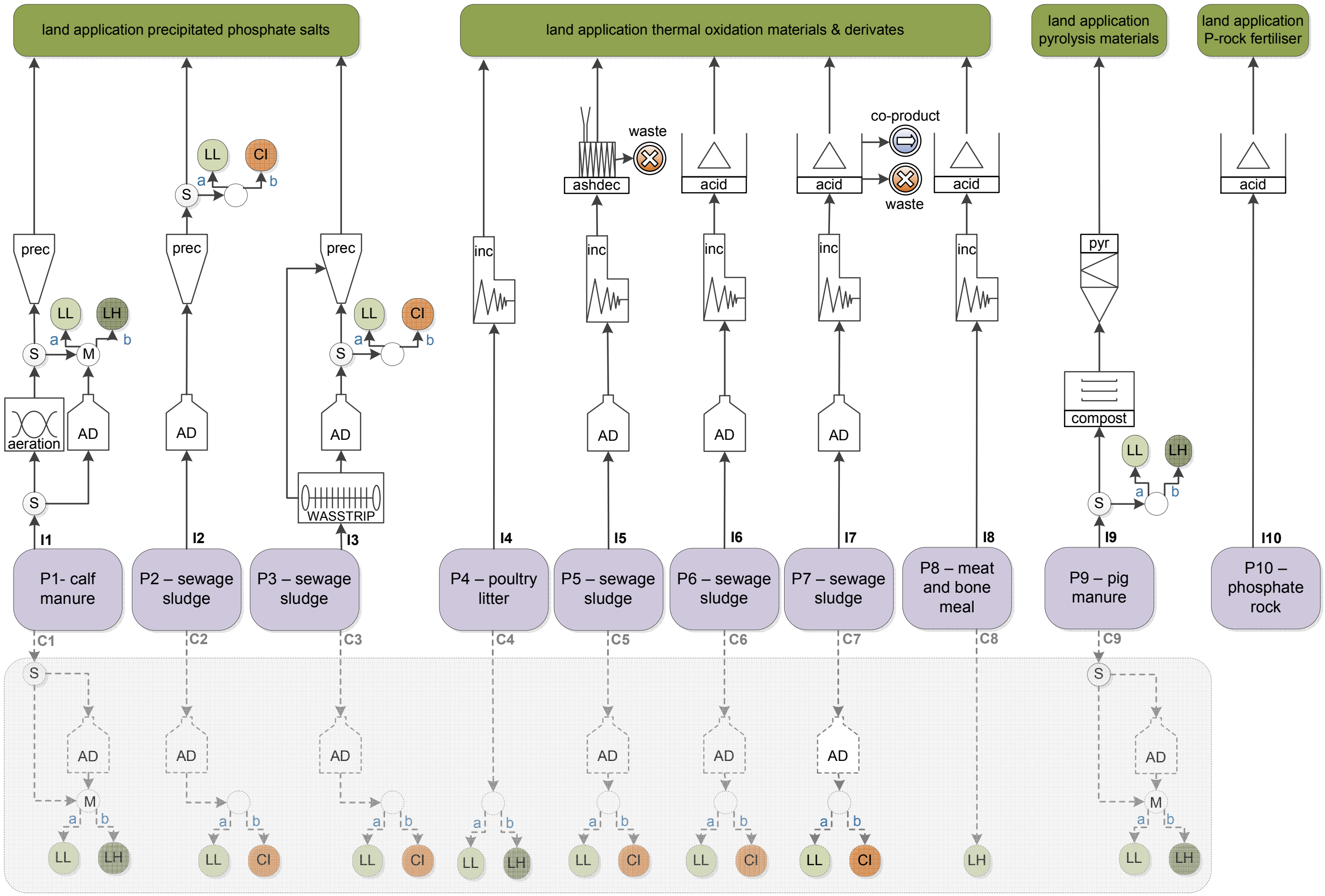
8.4 Scenario modelling

S**cenario modelling** will be applied to assess 10 different **production options** (P1-P10) for P-fertilisers. The selected production options for STRUBIAS materials include 9 different relevant P-fertilisers with a high TRL outlined in Section 7.3, which rely on manure, sewage sludge and rendered animal by-products as input materials. Additionally, the option to produce **mined rock phosphate and processed P-fertiliser** was investigated (P10, production of single superphosphate).

The different input materials for each of the **production options P1-P9** can be processed through a STRUBIAS production pathway ('**induced scenarios**', I1-I9) or through a different route under business-as-usual practices in Europe ('**counterfactual scenario**', C1-C9). The impacts and cost for each of the production shifts is then the difference between the induced scenario and the counterfactual scenario (see Section 8.5.1 for details). Additionally, the production of a **mined rock phosphate and processed P-fertiliser** was investigated (I10, production of single super phosphate). As this pathway involves the use of a primary raw material phosphate rock, no counterfactual scenario is applied (or alternatively, the impacts of the counterfactual scenario are zero; thus C10 = 0).

Nine STRUBIAS processes were investigated covering precipitated phosphate salts (three pathways, I1-I3, starting from veal calf manure and thickened sewage sludge as input materials), thermal oxidation materials & derivates (five pathways, I4-I8, starting from poultry litter, sewage sludge and meat and bone meal as input materials), and pyrolysis materials (I9, starting from pig manure as input material) (scenarios I1-I10; Figure 18). I1 involves the removal of nutrients from manure slurries through aeration (N) and K-struvite precipitation, and is performed by Stichting Mestverwerking Gelderland in the Netherlands. I2 and I3 involve the precipitation of struvites from sewage sludge prior to anaerobic digestion (I2, Airprex) and partly bypassing the anaerobic digester (I3, WASSTRIP plus Ostara – Pearl precipitation) at predominantly biological wastewater treatment plants. I4 (BMC Moerdijk) is the thermal oxidation of poultry litter and the direct application of the poultry litter ashes on the field. I5 (AshDec) involves the treatment of sewage sludge mono-incineration ashes with a sodium sulphate to partially remove heavy metals and to increase the plant availability of P contained in the sewage sludge (RAE of ~ 90%, see Section 6.2.3). I6 is the process envisaged by the mineral fertiliser industry for the production of SSP from sewage sludge ashes; it involves the acidulation of sewage sludge mono-incineration ashes with sulphuric ashes to produce SSP without significant metal/metalloid removal from the ashes. I7 is the Ecophos process in which sewage sludge mono-incineration ashes are acidulated with HCl, after which the metals/metalloids are separated from the P (H_3PO_4) and co-products ($CaCl_2$, $FeCl_3$). I8 is the same as I6 (mono-incineration plus acidulation using H_2SO_4), but uses meat and bone meal as input material. I9 involves the composting of the solid pig manure fraction obtained after solid-liquid separation, followed by the slow pyrolysis of the compost. I10 is the production of SSP through the acidulation of phosphate rock with sulphuric acid. **Total P-recovery** is low to medium for pathways that recover P as precipitated phosphate salts (I1: 13%, I2: 15%, I3: 46%) and pyrolysis materials (I9: 58%), but almost complete for the other pathways (I4-I8 and I10: > 95%). The different induced scenarios are schematically outlined in Figure 18 and described in detail in Section 14.

Counterfactual scenarios will be applied in function of the input material applied, and dependent on the regional situation (scenarios C1-C9; Figure 18). In order to account for regional variations in Europe for the counterfactual handling of the input materials '*nutrient-deficient areas (NDA)*' and '*nutrient surplus areas (NSA)*' were envisaged. The counterfactual handling of **manure** (C1, C4, C9) in NDA includes land application, possibly after anaerobic digestion, at low application rates that take full advantage of all macronutrients present in the manure. It is assumed that sufficient land is available to spread the manure in NDA. In NSA, manure is applied at an application rate in line with the maximum for manure nitrogen application of 170 kg N ha^{-1} yr^{-1}, as determined by Council Directive 91/676/EEC concerning the protection of waters against pollution caused by nitrates from agricultural sources. As a result of such application, P and K might be applied at rates that exceed crop requirements, and thus not contribute to plant nutrition but only to possible adverse effects (e.g. nutrient leaching and run-off to surrounding waters). As indicated by van Dijk et al. (2016), this situation is especially prevalent for P in various livestock-dense regions in Europe and is one of the major reasons for excessive P-accretion in European soils. It is also referred to in Section 8.6.2 for the principles governing the calculation of the nutrient fate. The fates of manure in NDA and NSA thus represent common scenarios for the handling of manure in Europe (see Section 13.3). The counterfactual handling of **sewage sludge** (C2-C3, C5-C7) in NDA includes anaerobic digestion at the wastewater treatment plant followed by land application at low application rates that take full advantage of all macronutrients present in the sewage sludge. For NSA, co-incineration together with mixed municipal waste at a co-incineration plant, followed by further use of the ashes as road base material was assumed. For P-recovery scenarios that only recover a share of the P present in the sewage sludge as precipitated phosphate salts, the fraction not recovered as a STRUBIAS material was assumed to have the same fate as the sewage sludge in the counterfactual scenario (i.e. land application and co-incineration in NDA and NSA, respectively). The fates of sewage sludge in NDA and NSA include the most common routes for the return of sewage sludge in Europe (see Section 13.5). The counterfactual handling scenario for **animal by-products other than manure** (C8 – meat and bone meal) includes their transport and direct application on land as an (organic) fertiliser in NDA and incineration and disposal in NSA. At present, about 20% of the category 2 and 3 by-product material is applied as fertiliser on land, with the remaining 80% not being used as fertiliser material (mainly used as pet food or feed, though some amounts are also incinerated; see Section 13.4.4).



NB: The results for each pathway P1-P9 are obtained from the difference between the induced use (I1-I9; upper half) and the counterfactual use (C1-C9; lower half, in grey box/dashed lines). S: solid-liquid separation, M: multiple inputs, AD: anaerobic digestion, prec: precipitation of phosphate salts, inc: mono-incineration, pyr: slow pyrolysis, acid: acidulation process; the lowercase blue letters indicate the use routes for biogenic materials within nutrient-deficient areas (NDA, a) and nutrient surplus areas (NSA, b) as follows - LL: land application of co-products at low application rates; LH: land application of co-products at high application rates; CI: co-incineration of co-products.

Figure 18: Schematic representation of the 10 production options for P-fertilisers through STRUBIAS production routes (P1-P9) and through the acidulation of phosphate rock (P10)

8.5 Study boundaries

8.5.1 Life cycle stages

The **boundaries of the life cycle system start at the extraction or inflow of the primary or secondary raw material to the P-fertiliser production plant and end with the use-on-land phase of the manufactured P-fertiliser**. The impacts of supplying such feedstock to STRUBIAS production processes are accounted for by applying the same input material both for the induced as for the counterfactual scenario. The life cycle stages included in the impact and cost assessment include **transport** from the production site of the input material to the processing plant, **material transformation and manufacturing stages** as depicted in Figure 18, **granulation**, **transport** to use on land, **storage** at the farm, **land application**, and **use on land**. The cost assessment does not include externalities (e.g. additional cost for drinking water production from surface waters) due to the uncertainty associated with their monetary values; these will only be presented as separate environmental impacts. The approach to calculate the STRUBIAS production cost is also known as life cycle costing and aims to assess the conventional budget costs, without monetising environmental impacts (e.g. Martinez-Sanchez et al., 2015).

Under the default scenario modelling approach, a **standard transport distance of 25 km** was assumed for all transport stages (e.g. from site of collection to processing plant, from wastewater treatment plant to incineration facility, from P-fertiliser manufacturing site to use on land). The only exception was the transport from the place of phosphate rock extraction to the manufacturing site. As part of the **variant analyses** (see Section 8.11), transport distances for manure were varied to simulate a scenario where the manure is exported from one EU Member State to another, with an assumed transport distance of 500 km. **Storage** was assumed for all materials that will be applied on agricultural land in the induced and counterfactual scenarios, and was material-specific (e.g. liquid manure in a covered lagoon, solid manure in a (covered) field heap with an impermeable concrete floor, P-fertilisers in an indoor silo). Nutrient losses during storage were considered (e.g. ammonia volatilisation, methane and nitrous oxide emissions). It was assumed that storage only occurred after the material transformation cycle, and thus prior to use on land by the farmer. The assessment includes the **granulation/pelletising** of slurries or powders formed within a manufacturing process for all P-fertilisers. Granulation is not included for manures and processed manures that are applied on land.

The cost assessment includes **capital costs** with an investment amortisation time of 20 years, and a net interest rate of 5%. **Operational costs** include costs of intermediates (e.g. sulphuric acid, hydrochloric acid, lime; purchased as primary raw materials on the market) and labour. Revenues from co-products (e.g. energy, bio-available nutrients present in processed manure fractions, $CaCl_2$/$FeCl_3$ in I7) were included. For P-recovery as precipitated phosphate salts at biological wastewater treatment plants, the benefits of reduced polymer needs and sludge volumes for dewatering were taken into consideration as well as the reduced energy needs for the plant (e.g. due to the precipitation of N in struvites) and chemical coagulant needs relative a default biological wastewater treatment plant. Operational benefits due to reduced plant maintenance needs and avoided tube clogging were, however, not considered due to the lack of available data. In all scenarios, it was assumed that heat was not recovered as a marketable product.

8.5.2 Energy and mass balances

The impacts associated with the following elements were assessed: **carbon**, the **macronutrients N, P and K**, and the **metals and metalloids** arsenic (As), cadmium (Cd), chromium (Cr), copper (Cu), mercury (Hg), nickel (Ni), lead (Pb) and zinc (Zn). The impact of other micronutrients and/or possible contaminants was disregarded in this assessment because of their limited impact on the overall results when compliant with the STRUBIAS recovery rules. Energy and mass were traced throughout the system by fully closing the mass and energy system balances. The output material of the system was in agreement with the product quality (nutrient content, metal/metalloid content) required for the P-fertilisers.

8.6 Calculations

8.6.1 Functional unit and basic calculation approach

The **functional unit** of the assessment is **1 kg of bio-available P recovered <u>in the P-fertiliser material</u>**, either being a STRUBIAS material (P1-P9) or SSP as a selected mined phosphate rock and processed P-fertiliser (P10). **Choosing this functional unit enables a comparison between the different pathways that produce P-fertilisers from both primary and secondary raw materials, including sewage sludge, manure, and others**. This choice is unconventional but **spot on** because (i) we assume that **P-recovery is a function or co-function** of (future) biomass transformation cascades, (ii) we evaluate the system from a **product perspective** since all pathways result in the production of a marketable P-fertiliser product, and (iii) possible other co-functions and co-services of the different pathways are fully **credited** through system expansion. In contrast to other studies that compare management scenarios for the handling of biogenic materials from a waste management perspective (research question ‘what are the impacts related to the full share of those activities that are expected to change when implementing P-recovery units as part of the waste or wastewater treatment plant’? – note: in this case a comparison with phosphate-rock-derived P-fertiliser production processes is impossible due to the conflicting functional unit), we thus evaluate the system from an alternative product perspective, and fully account for the waste management function as an integral part of the biomass transformation cascade. The research question of our study would be ‘what are the impacts related to the full share of those activities that are expected to change when using available biogenic feedstocks for innovative processes that produce a marketable P-fertiliser in a circular economy, and how do those impacts compare to the production of phosphate-rock-derived P-fertilisers in a linear economy?’. **This approach is in line with the key principles of a circular economy that focuses on the recovery of dissipated materials as value-added materials for society.**

Co-products and co-services other than bio-available P in the P-fertiliser (e.g. energy, bioavailable N and P in manure) were diverted and credited, both from an impact perspective (e.g. possible N leaching from manure fractions) and from a monetary point of view (e.g. avoided purchase cost of mineral N fertiliser for farmer; see also Section 8.6.3) (see Figure 19).

The choice of this functional unit implies that, for each individual scenario Px, different quantities of input materials are processed, depending on the P content of the input material, the P-recovery efficiency and the agronomic efficiency of STRUBIAS material resulting from pathway Ix relative to mined and synthetic P-fertilisers (Table 12). Based on these data, the amount of input material that is applied to obtain 1 kg of bio-available P recovered in the P-fertiliser material in the induced pathways (I1-I10) can be calculated. **The results for the counterfactual pathways are then obtained by processing the same amount of input material through pathways C1-C9.**

Table 12: Conversion factor between functional unit for the LCA, and input and output materials applied for the different production options for P-fertilisers outlined in Figure 18.

pathway	Functional unit	Total P in STRUBIAS material (kg)	Total weight of STRUBIAS material (kg)	composition STRUBIAS material	Total weight of input material applied (kg)	Type of input material applied
P1	1 kg bio-available P in STRUBIAS material	1.00	16.7	K-struvite	12410	raw calf manure (slurry, 4% DM)
P2		1.00	11.0	struvite	5731	undigested sewage sludge (5% DM)
P3		1.00	7.9	struvite	1772	undigested sewage sludge (5% DM)
P4		1.08	17.6	poultry litter ash	139	raw poultry litter (48% DM)
P5		1.11	15.0	AshDec fertiliser	925	undigested sewage sludge (5% DM)
P6		1.00	18.0	SSP-like material	817	undigested sewage sludge (5% DM)
P7		1.00	5.0	TSP	824	undigested sewage sludge (5% DM)
P8		1.00	11.5	SSP-like material	31	meat-and-bone meal (97% DM)
P9		1.18	28.3	pyrolysis material	3565	raw pig manure (slurry, 5% DM)
P10	1 kg bio-available P in SSP	1.00	12.7	SSP	7.5	phosphote rock (100% DM)

NB: SSP: single superphosphate; TSP: single superphosphate.

The environmental and human health impacts for fertilising products containing precipitated phosphate salts & derivates, thermal oxidation materials & derivates, and pyrolysis & gasification materials are thus calculated by attributing the impacts and costs of the material transformation process to the P-fertiliser based on the calculated impacts/costs and the bio-available P recovery efficiency for each pathway, as follows:

$$IM\ (Ix/Cx) = \sum_{m=1,Ix/Cx}^{M,Ix/Cx} IM_{man} + \sum_{t=1,Ix/Cx}^{T,Ix/Cx} IM_{TS} + \sum_{k=1,Ix/Cx}^{K,Ix/Cx} AIM_{co} + \sum_{o=1,Ix/Cx}^{O,Ix/Cx} IM_{UOL-OFM} + IM_{UOL-PF,Ix}$$

Where IM (Ix/Cx): results for a specific impact category for pathways Ix or Cx; $\sum_{m=1,Ix/Cx}^{M,Ix/Cx} IM_{man}$: the sum of the impacts resulting from manufacturing operations for the different life cycle stages in the manufacturing chain (1→M) of scenarios Ix or Cx; $\sum_{t=1,Ix/Cx}^{T,Ix/Cx} IM_{TS}$: the sum of the impact resulting from the different transport and storage life cycle stages (1→T); $\sum_{k=1,Ix/Cx}^{K,Ix/Cx} AIM_{co}$: the sum of avoided impacts resulting from the different co-products and co-services (1→K) generated for scenarios Ix or Cx (e.g. avoided energy production, avoided fertiliser production); $\sum_{o=1,Ix/Cx}^{O,Ix/Cx} IM_{UOL-OFM}$: the sum of the impacts resulting from the use on land (1→O) of fertilising materials other than the P-fertiliser (e.g. processed manure); $IM_{UOL-PF,Ix}$: the impact resulting from the use on land of the P-fertiliser, thus the STRUBIAS material for I1-I9 or SSP for I10.

$$COST(Ix/Cx) = \sum_{l=1,Ix/Cx}^{L,Ix/Cx} CAPEX_{man} + \sum_{l=1,Ix/Cx}^{L,Ix/Cx} OPEX_{man} + \sum_{t=1,Ix/Cx}^{T,Ix/Cx} COST_{TS} + \sum_{c=1,Ix/Cx}^{C,Ix/Cx} savings_{co}$$

Where $COST(Ix/Cx)$: production cost for pathways Ix or Cx; $\sum_{l=1,Ix}^{L,Ix/Cx} CAPEX_{man}$: the sum of the capital cost of the equipment for the different life cycle stages (1→L) of the manufacturing chain; $\sum_{l=1,Ix}^{L,Ix/Cx} OPEX_{man}$: the sum of the operational cost for the different life cycle stages (1→L) in the manufacturing chain; $\sum_{t=1,Ix/Cx}^{T,Ix/Cx} COST_{TS}$: capital and operational costs associated with transport and storage; and $\sum_{c=1,Ix}^{C,Ix/Cx} savings_{co}$: monetary savings due to the avoided production of conventional market products due to co-products and co-services (1→C) (e.g. avoided energy production, avoided mineral N fertiliser production).

Based on these results, the impacts and costs can then be calculated for each entire production option as follows:

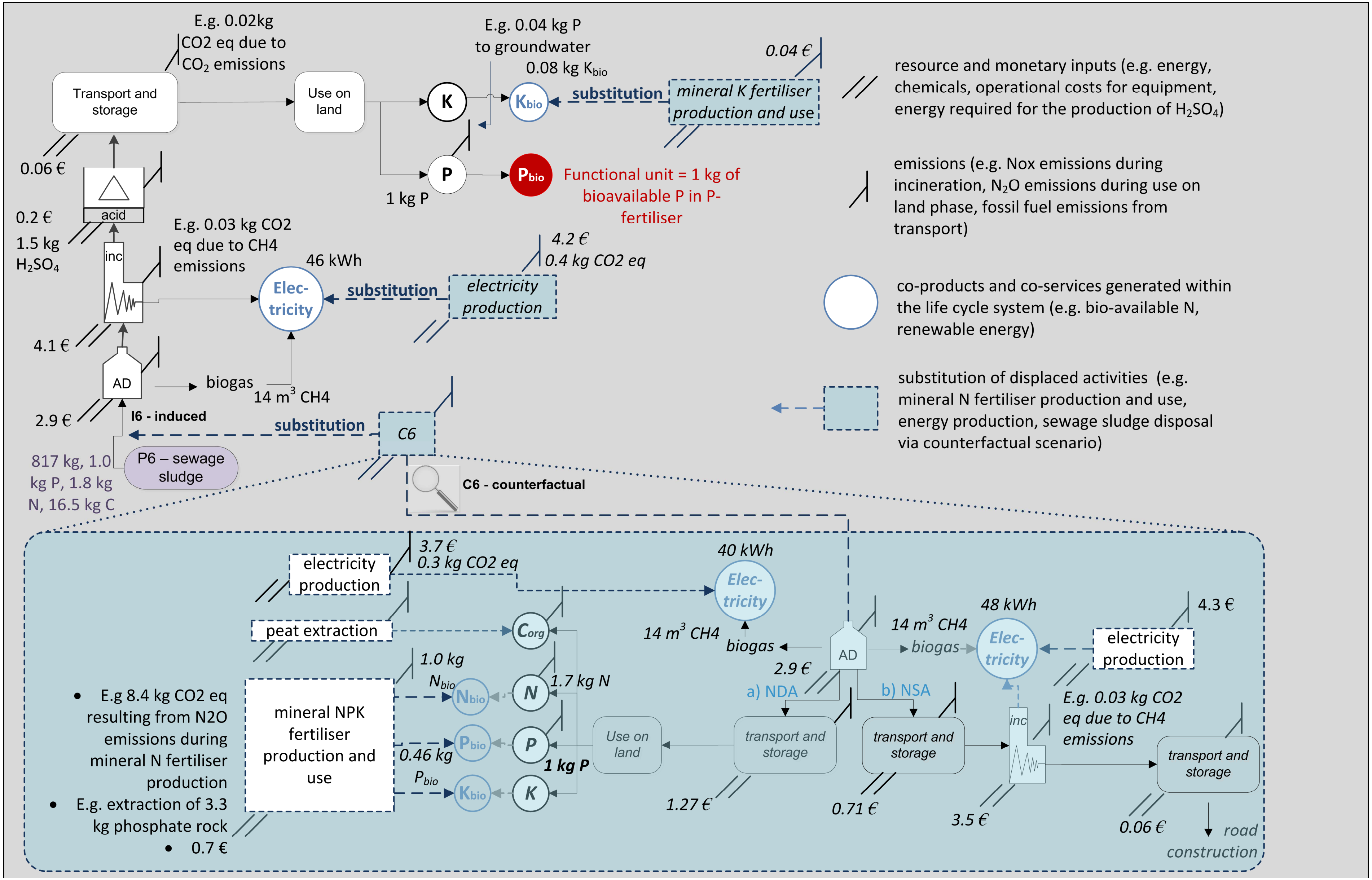
$$IM(Px) = IM(Ix) - IM\ (Cx)$$

Where $IM(Px)$: results for a specific impact category for STRUBIAS production option Px; $IM(Ix)$: results for the specific impact category for pathway Ix; and $IM\ (Cx)$: results for the specific impact category for pathway Cx.

$$COST(Px) = COST(Ix) - COST\ (Cx)$$

Where $COST(Px)$: estimated production cost for STRUBIAS production option Px, $COST(Ix)$: production cost for pathway Ix, and $COST\ (Cx)$: production cost for pathway Cx.

The conceptualisation of the life cycle system applied as well as the calculation principles are outlined schematically in Figure 19.


E.g. 0.02kg CO2 eq due to CO_2 emissions
Transport and storage
Use on land
0.06 €
acid
0.2 €
1.5 kg H_2SO_4
inc
4.1 €
AD
2.9 €
I6 - induced
biogas
14 m³ CH4
E.g. 0.03 kg CO2 eq due to CH4 emissions
46 kWh
Elec-tricity
substitution
electricity production
4.2 €
0.4 kg CO2 eq
E.g. 0.04 kg P to groundwater
0.08 kg K_{bio}
K
K_{bio}
substitution
mineral K fertiliser production and use
0.04 €
P
P_{bio}
1 kg P
Functional unit = 1 kg of bioavailable P in P-fertiliser
substitution
C6
P6 – sewage sludge
817 kg, 1.0 kg P, 1.8 kg N, 16.5 kg C
C6 - counterfactual
resource and monetary inputs (e.g. energy, chemicals, operational costs for equipment, energy required for the production of H_2SO_4)
emissions (e.g. Nox emissions during incineration, N_2O emissions during use on land phase, fossil fuel emissions from transport)
co-products and co-services generated within the life cycle system (e.g. bio-available N, renewable energy)
substitution of displaced activities (e.g. mineral N fertiliser production and use, energy production, sewage sludge disposal via counterfactual scenario)
electricity production
3.7 €
0.3 kg CO2 eq
40 kWh
Elec-tricity
14 m³ CH4
biogas
peat extraction
C_{org}
1.0 kg N_{bio}
N_{bio}
N
1.7 kg N
mineral NPK fertiliser production and use
P_{bio}
P
0.46 kg P_{bio}
1 kg P
K_{bio}
K
Use on land
• E.g 8.4 kg CO2 eq resulting from N2O emissions during mineral N fertiliser production
• E.g. extraction of 3.3 kg phosphate rock
• 0.7 €
AD
2.9 €
a) NDA
b) NSA
transport and storage
1.27 €
transport and storage
0.71 €
14 m³ CH4
48 kWh
biogas
Elec-tricity
electricity production
4.3 €
inc
3.5 €
E.g. 0.03 kg CO2 eq due to CH4 emissions
transport and storage
0.06 €
road construction

A consequential product system has been used to answer the research question: 'what are the impacts related to the full share of those activities that are expected to change when using available biogenic feedstocks for innovative processes that produce a marketable P-fertiliser in a circular economy, and how do those impacts compare to the production of phosphate-rock-derived P-fertilisers in a linear economy?' Therefore, the life cycle system is approached from a product perspective and *1 kg of bio-available P in the P-fertiliser* is used as a functional unit for this life cycle assessment (red box P_{bio}; note that the quantity of the input material applied has been back-calculated in order to end up with the functional unit at the end of the production and use chain). Impacts can then be compared for all production options (P1-P10), regardless of source material used.

Consequential models reflect physical and monetary causalities, and the consequences associated with the substitution of displaced activities are considered. This includes, amongst others, the avoided handling and use of the biogenic material that is used as feedstock to produce STRUBIAS materials. The impacts associated with the alternative feedstock handling are the net result of the sum of different processes (possibly also involving processing, transport and use on land as life cycle stages), and may ultimately determine the overall system result. In order to better understand the impacts from the avoided counterfactual feedstock handling (C), these results are presented in a disaggregated manner in addition to the gross impacts from the induced pathway (I). The net results for the pathways P1-P9 are then calculated as $P_{1-9} = I_{1-9} – C_{1-9}$. The source material for mined rock phosphate and processed P-fertilisers is obtained through the extraction of a primary raw material. In the event that phosphate rock is not extracted, it will remain stored in natural bedrock layers adequately protected from exposure so the environmental impact due natural weathering can be ignored. Therefore, the impacts of the counterfactual feedstock handling scenario for P10 are considered zero and P10 = I10.

Now, the impacts and costs of the induced (I6) and counterfactual (C6) scenario are calculated individually. This implies that we have to take into consideration all impacts, costs and possible savings from the scenario. For instance, if the STRUBIAS manufacturing process involves the mono-incineration of the sewage sludge then this process is associated with a cost (e.g. costs of building and operating the incinerator plants), an environmental impact from input (e.g. mono-incineration requires limestone; the impacts of extracting the limestone from nature are taken into consideration) and output materials (e.g. NOx emissions to air leaving the incinerator), and a saving (some energy can be recovered from the sewage sludge; as this energy is not part of the functional unit applied, it is assumed that the energy is diverted and credited both from an impact perspective as well as from a monetary point of view; the placing on the market of the renewable energy implies for example that fossil fuel emissions to air from coal combustion are avoided, and that the STRUBIAS manufacturer will receive monetary compensation for the avoided energy production he substitutes on the market).

By opting for the STRUBIAS pathway instead of the counterfactual scenario, all impacts and costs that are associated with the counterfactual scenario have to be deducted, whether they are positive or negative. For instance, the bio-available nutrients (N_{bio}, P_{bio}, K_{bio}) present in the digested sludge in the counterfactual scenario (NDA) are not part of our functional unit and thus assumed to be a co-product or co-service in the C scenario. This implies that the bio-available nutrients are diverted and credited in the C scenario; the placing on the market of the bio-available nutrients in the NDA counterfactual scenario implies that the environmental and cost impacts from the production and use of mineral N, P and K fertilisers are displaced or avoided. Note that for the overall production option results, the substitution of displaced activities in the counterfactual scenario are ultimately system burdens ($P = I – C = I - (\text{impacts}_{\text{counterfactual}} – \text{savings}_{\text{counterfactual}}) = I - \text{impacts}_{\text{counterfactual}} + \text{savings}_{\text{counterfactual}}$; represented by boxes that again lose their blue colour and italic text in the counterfactual scenario).

The possible diversion and crediting of bio-available P derived from biogenic feedstocks in the counterfactual scenario may result in an altered demand for phosphate rock. This is exemplified by comparing NDA to NSA. In the counterfactual scenario C for NDA, where P is brought back to agricultural land, the crediting will result in the substitution by phosphate rock, but not in the counterfactual scenario for NSA, where P is lost in road construction materials. In the counterfactual scenario for NSA, the biogenic material is fully removed from the biogeochemical cycle, and any P-recovery from the dissipated P will effectively contribute to a reduced phosphate rock dependence relative to the production and use of mined rock phosphate fertilisers. In NDA, the biogenic material is already a plant-available P-source in the counterfactual scenario; turning it into a STRUBIAS material will therefore only reduce the depletion of phosphate rock in the event that the effective plant P uptake from the biogenic material is increased during the STRUBIAS manufacturing process.

These conceptual and calculation approaches are exemplified as follows for fictitious monetary costs, but the same principles apply to the environmental impacts.

The cost of producing 1 kg of bio-available P through the induced pathway is:

COST(I6) = 2.9 € (anaerobic digestion) + 4.1 € (mono-incineration) + 0.2 € (acidulation unit) + 0.06 € (transport and storage of SSP-like material) – 4.2 € (avoided energy production through substitution) - 0.04€ (avoided K-fertiliser production through substitution) = 3.0 €

The cost of processing the same amount of input material through the counterfactual pathway in NDA is:

COST (C6 for NDA) = 2.9 € (anaerobic digestion) + 1.3 € (transport and storage) – 3.7 € (avoided energy production through substitution) - 0.7 € (avoided NPK fertiliser production through substitution) = - 0.2 €

The cost of processing the same amount of input material through the counterfactual pathway in NSA is:

COST (C6 for NSA) = 2.9 € (anaerobic digestion) + 0.7 € (transport and storage) + 3.5 € (co-incineration) + 0.06 € (transport and storage) – 4.3 € (avoided energy production through substitution) = 2.9 €

Hence, the net cost for P6 is (COST(P6) = COST (I6) – COST (C6)):

COST (P6 for NDA) = 3.0 € (COST I6) – (-0.2 € (COST C6 for NDA)) = 3.2 €

COST (P6 for NSA) = 3.0 € (COST I6) – 2.9 € (COST C6 for NSA)) = 0.1 €

The cost relative to pathway 10 (SSP from mined rock phosphate) can be calculated by comparing COST(P6) to COST(P10).

NB: The processes and numbers given above are fictitious and incomplete, therefore the results do not correspond to the actual result of pathway P6.

Figure 19: Conceptualisation of the life cycle system applied in this study as exemplified by pathway P6, with landspreading ((a) on the lower left-hand side), and co-incineration ((b) on the lower right-hand side) as selected counterfactual scenarios for nutrient-deficient areas (NDA) and nutrient surplus areas (NSA), respectively

8.6.2 Fate and credits for fertilising materials other than STRUBIAS

It was assumed that farmers require plant-available NPK and use organic C as part of good farming practices. When plant-available nutrients and organic C were returned to land in the scenarios, the **avoided cost for the purchase of mineral NPK fertilisers and peat as a substitute for the organic C was thus taken into consideration** as outlined above.

In the event that fertilising substances were returned to land, the fate of the included nutrients was modelled as follows for N, P, K and organic C in **nutrient-deficient areas (NDA)**:

- For **inorganic N fractions**, the nitrogen use efficiency was assumed to be 80%, the mean value of the targeted ranges for good management practices as outlined by Brentrup and Pallière (2010). It is thus assumed that plants take up 80% of the inorganic N supplied under good management practices. It is acknowledged that wide variation in N budgets across Europe depends, amongst others, on agricultural intensity, soil types, and farm management practices (de Vries et al., 2011). For the loss of N by emission of N_2O via denitrification, we used the Intergovernmental Panel on Climate Change (IPCC) linear Tier 1 N_2O default emission factor of a 1% loss of applied N as N_2O-N (Good and Beatty, 2011). Minimal default losses of 1% were also assumed for NH_3 emissions under good management and storage practices. The remaining 18% was then assumed to be equally distributed between denitrification (9%) and N leaching losses to groundwater (9%). For the **organic N fractions (manure, sewage sludge)**, it was assumed that the stable organic N fraction was not available to the crops in the long term. The latter fraction was assumed to be 11% of the total organic N, in line with Bruun et al. (2006). Hence, it was assumed that 89% of the organic fraction was available in the long term to plants. Because of the long-term N release from the organic fraction, the nitrogen use efficiency of the plant-available organic N fraction was assumed to be 60% in line with the average values given for conditions that involve a risk for (over-winter) N leaching losses (Brentrup and Pallière, 2010). The supplementary N losses relative to inorganic N fractions were fully attributed to N leaching losses, resulting in a value of 29% for leaching losses from organic manure N sources. Other losses and emissions from the plant-available N fraction were assumed to be equal to mineral N fertiliser: NH_3 losses (1%), N_2O losses (1%), N_2 losses (8%), and N surface run-off (1%). Plant N uptake was then calculated as the sum of the inorganic N plus the plant-available organic N being taken up, as follows: $((N_{inorganic} \times 0.8) + (N_{organic} \times 0.89 \times 0.6)) / (N_{inorganic} + N_{organic})$.
  The nitrogen agronomic efficiency of STRUBIAS material resulting from pathways relative to synthetic mineral N-fertilisers were assumed to be 100% for struvites, and 10% for the pyrolysis materials derived from pig manure. Thermal oxidation materials & derivates do not contain nitrogen as the N is lost during the manufacturing process.
- For **mineral P fractions**, it was considered that crops take up 90% of the applied P (Syers et al., 2008). P losses to water were estimated at 8% of the P applied, equally distributed over losses to surface waters through run-off (4%) and groundwater through leaching (4%) (van Dijk et al., 2016). The remaining 2% was estimated to be stored in the soil. The mineral P fertiliser substitution efficiency of **manure and processed manure P fractions** was assumed to be the average for P-deficient areas (70%) and areas with adequate P-

background soil concentrations for plant growth (100%), based on Eghball et al. (2002). Hence, plant P uptake from organic P fractions was estimated at 76.5% (0.85% x 90%). Losses to surface (4%) and ground (4%) water for organic P fractions are assumed to be similar for mineral P-fertilisers, and the fraction of P stored in soils was slightly increased relative to mineral P fertilisers (15.5%). The mineral P fertiliser substitution efficiency of anaerobically digested sewage sludge and meat and bone meal was assumed to be 46% and 40%, respectively (Oenema et al., 2012). When the plant P demand was lower than the bio-available P fraction in the organic P fractions, it was assumed that the bio-available P fraction that was not taken up by the plant was stored in the soil in NDA and potentially lost to ground- and/or surface water due to soil P saturation in NSA (Schoumans, 2015). The agronomic efficiency of STRUBIAS materials resulting from pathways relative to mined and synthetic P-fertilisers was assumed to be 100% for all pathways, with the exception of poultry litter ashes, AshDec fertiliser and pig manure biochar for which relative agronomic efficiencies of 93%, 90% and 85% were assumed, respectively (see Chapter 6).

- The mineral K fertiliser substitution efficiency of **organic K fractions** was assumed to be 73%, based on Eghball et al. (2002). Potassium losses from the soil ecosystem were not further analysed as K has no influence on any of the impact categories considered.

It was assumed that the fate of the applied nutrients might not only be constrained by the intrinsic properties of the fertilising material, but also by inappropriate fertilising management practices that lead to the supply of nutrients in excess of plant demand. Manure and organic fertilising materials are sometimes characterised by imbalanced stoichiometric ratios, especially N/P ratios. As a result, the application of high amounts of manure or processed manure can result in the inefficient uptake of certain nutrients that are applied in excess of plant needs. With N often being the limiting element for plants, the application of manure and processed manure at the maximal application rates of 170 kg ha^{-1} yr^{-1} as laid down in the EU Nitrates Directive, can lead to a P and K surplus in the soils that is then stored in the soil matrix or leached towards ground waters. This situation is not uncommon as 39% of all P entering the EU as P-fertilisers and animal feed is accumulated in agricultural soils as a result of ineffective nutrient management practices (van Dijk et al., 2016).

For NSA, it was assumed that manure is applied at an application rate in line with the maximum for manure nitrogen application of 170 kg N ha^{-1} yr^{-1} from the Nitrates Directive. The recommended P and K application rates for productive cropland ecosystems were estimated at 17.5 kg P ha^{-1} yr^{-1} and 76.8 kg K ha^{-1} yr^{-1}, respectively, based on typical application rates and FAO fertiliser guidelines for major crops (Johnston and Steen, 1999; Roy et al., 2006). The recommended P application rates are at the higher end of the recommended P application rate for most EU Member States (0-22 kg P ha-1 yr-1 for most countries in western and northern Europe; Tóth et al., 2014). **Any bioavailable P applied in excess of the recommended application rates was considered not to contribute to plant nutrition.**

Across all pathways, it was assumed that **STRUBIAS fertilisers and mineral fertilisers were applied according to best management practices**. This implies that no constraints apply for plant nutrient uptake due to possible fertiliser applications in excess of plant nutrient demands.

8.6.3 Other assumptions

With respect to electricity production, the **standard electricity mix for the Netherlands**, as a representative country for P-recovery processes, was used in the assessment. With respect to conventional nitrogen fertiliser, the average EU mix was assumed (27% AN, 33.3% CAN, 15.3% urea-AN and 24.3% urea).

In line with the IPCC guidelines, **zero carbon emissions from biogenic wastes** with a short turnover time were assumed. It was, however, assumed that the **stable C fraction of processed manures and pyrolysis materials contributes to the mitigation of greenhouse gas emissions**. Therefore, the fraction of C remaining in the soil after the applied life cycle time period of 100 years was accounted for as sequestered carbon for the impact category 'global warming potential, GWP'. This fraction was assumed to be 11% and 90% of the organic C applied, for processed manure (Bruun et al., 2012) and pyrolysis materials (Lehmann and Joseph, 2015), respectively.

8.7 Data sources for life cycle inventories

A part of the primary data of the life cycle inventories for the different induced and counterfactual scenarios, data were collected by a third party (RDC Environnement) on behalf of the JRC from STRUBIAS material producers and literature information (Wiechmann et al., 2013b; Janik et al., 2015a; Jossa and Remy, 2015; Egle et al., 2016; Eurostat, 2016; STOWA, 2016a; Centre, 2017; De Graaff et al., 2017; ECN, 2017; Nattorp et al., 2017).

Primary data from life cycle inventories as provided by the manufacturers were, in some cases, modified by the JRC for standardisation purposes (e.g. transport distances, liquid-solid separation efficiencies; energy and biomass recovery efficiency). Therefore, **the results presented can by no means be generalised and a significant modification of the results can be expected in the event that the actual implementation differs from the assumptions applied in this study**. The results presented here are thus not fully representative for specific recovery pathway or operators, and actual results may be dissimilar. It is important to recall that this section of the report relies on the use of hypothetical scenario modelling. As a matter of fact, it may be most informative to focus on the results provided for the different life cycle stages in the tables as these results enable a better visualisation of the driving mechanisms of the net results for the different environmental impact categories and production costs.

8.8 Phosphate rock depletion

STRUBIAS P-fertilisers are not derived from phosphate rock, but from secondary raw materials. Nonetheless, the counterfactual handling scenario of the input materials applied for

STRUBIAS manufacturing scenarios may involve land application (e.g. manure, sewage sludge in NDA), and may thus serve as a P-source for a farmer. Therefore, the net effect on phosphate rock depletion is dependent on the efficacy of the P-return in the counterfactual scenario; the more efficient the P return in the counterfactual scenario, the less interesting the STRUBIAS pathway becomes to address source risks.

The **net impact on phosphate rock depletion is highest for pathway P10** (**Figure 20**). For every kg of bio-available P produced, 7.2 kg of phosphate rock is extracted. The production of 1 kg of bio-available P in the STRUBIAS scenario results in net phosphate rock extraction values that range from 0 kg to 6.5 kg of phosphate rock (**Figure 20**).

The phosphate rock depletion values are typically **higher in NDA** because the counterfactual use of the biogenic materials involves landspreading; in the event that the counterfactual scenario is avoided, the bio-available P contained in the biogenic material is assumed to be replaced by mined rock phosphate P-fertilisers. Because the plant P bio-availability is lower for sewage sludge and meat and bone meal than for manure, greater savings can be obtained for the former than for the latter. In **NSA**, the landspreading of manure at high application rates results in the accumulation of P in the soil; the P applied exceeds the plant P demand. Co-incineration was assumed as the fate of the sewage sludge and no bio-available P is thus applied on land in the counterfactual scenario. Hence, the production of 1 kg of bio-available P exclusively occurs from dissipated P sources and the net result is a zero phosphate rock extraction rate (**Figure 20**).

Note that the **overall potential of the different STRUBIAS pathways to address source risks** is not only dependent on the net phosphate rock depletion per kg of bio-available P produced, but also the total amount of bio-available P that can be produced from the available feedstock via each pathway (thus dependent on feedstock availability and P-recovery efficiency with the latter typically being higher for thermal oxidation materials & derivates and pyrolysis & gasification materials than for precipitated phosphate salts & derivates.)

8.9 Environmental and human health impacts

The results for the induced and counterfactual scenarios for the different production options P1-P10 are presented in a disaggregated manner in Table 13 and Table 14 for nutrient-deficient areas (NDA) and for nutrient surplus areas (NSA). The aggregated net results (P = I – C) are presented in Figure 20. Specific impact categories have been presented, but results and trends for other impact categories are in line with the presented results as follows: (i) global warming potential ~ photochemical ozone formation, particulate matter, fossil resource depletion; (ii) marine eutrophication – N ~ terrestrial acidification, freshwater eutrophication; and (iii) human toxicity - cancer ~ human toxicity – non-cancer, ecotoxicity.

Positive values indicate a burden to the environment, while **negative values indicate a saving**. The sum of burdens and savings provides the overall net contribution; this sum may also be negative. This may be a result of credits from co-products.

Table 13: Results of the life cycle analyses presented in disaggregated form for the different life cycle stages for the impact categories global warming potential, marine eutrophication – nitrogen, and human toxicity - cancer for the pathways Ix and Cx in <u>nutrient-deficient areas (NDAs)</u> as graphically presented in Figure 18 (see Section 8.6.1 for explanations on the abbreviations).

		P1		P2		P3		P4		P5		P6		P7		P8		P9		P10
category	stage	I1	C1	I2	C2	I3	C3	I4	C4	I5	C5	I6	C6	I7	C7	I8	C8	I9	C9	I10
global warming potential (kg CO2-eq kg^{-1} P bioavailable)	IM_{man}	29.9	21.3	52.4	45.6	13.6	14.1	1.5	0.0	9.8	7.4	8.3	6.5	9.9	6.6	1.0	0.0	17.4	15.5	3.5
	IM_{TS}	72.4	86.3	8.0	8.4	1.2	2.6	0.8	16.4	0.5	1.4	0.4	1.2	0.4	1.2	0.2	0.1	26.0	25.6	0.0
	AIM_{co}	-202	-572	-138	-139	-42.0	-43.0	-7.8	-35.4	-3.7	-22.5	-3.3	-19.8	-4.4	-20.0	-1.7	-25.8	-78.2	-113.6	0.0
	$IM_{UOL-OFM}$	46.7	163.6	49.0	50.8	13.9	15.7	0.0	-0.4	0.0	8.2	0.0	7.2	0.0	7.3	0.0	7.3	25.0	34.1	0.0
	IM_{UOL-PF}	0.0	0.0	1.4	0.0	1.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	-30.2	0.0	0.0
	Total	**-52.5**	**-301**	**-26.8**	**-34.3**	**-12.2**	**-10.6**	**-5.6**	**-19.3**	**6.5**	**-5.5**	**5.5**	**-4.9**	**6.0**	**-4.9**	**-0.4**	**-18.3**	**-40.0**	**-38.3**	**3.5**
freshwater eutrophication - phosphorus (kg P-eq kg^{-1} P bioavailable)	IM_{man}	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
	IM_{TS}	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
	AIM_{co}	-0.44	-0.54	-0.41	-0.48	-0.08	-0.15	0.00	-0.08	0.00	-0.08	0.00	-0.07	0.00	-0.07	0.00	-0.07	-0.05	-0.14	0.00
	$IM_{UOL-OFM}$	0.43	0.50	0.41	0.47	0.08	0.15	0.00	0.07	0.00	0.08	0.00	0.07	0.00	0.07	0.00	0.07	0.05	0.13	0.00
	IM_{UOL-PF}	0.07	0.00	0.07	0.00	0.07	0.00	0.07	0.00	0.07	0.00	0.07	0.00	0.07	0.00	0.07	0.00	0.08	0.00	0.07
	Total	**0.06**	**-0.03**	**0.06**	**-0.01**	**0.07**	**0.00**	**0.07**	**0.00**	**0.07**	**0.00**	**0.07**	**0.00**	**0.07**	**0.00**	**0.07**	**0.00**	**0.08**	**-0.01**	**0.07**
human toxicity - cancer (kg CTUh kg^{-1} P bioavailable x 1E-07)	IM_{man}	3.62	3.03	7.47	7.32	1.51	2.26	1.03	0.00	1.95	1.18	1.77	1.04	1.95	1.05	0.34	0.00	3.09	1.21	0.36
	IM_{TS}	2.18	2.50	0.08	0.08	0.02	0.02	0.03	0.02	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.00	0.74	0.73	0.00
	AIM_{co}	-12.46	-22.97	-8.13	-8.94	-1.97	-2.76	-0.95	-2.17	-0.43	-1.44	-0.38	-1.27	-0.55	-1.29	-0.23	-1.07	-2.51	-4.82	0.00
	$IM_{UOL-OFM}$	5.33	5.36	50.77	51.08	15.79	15.80	0.00	4.08	0.00	8.25	0.00	7.28	0.00	7.35	0.00	0.24	1.62	1.96	0.00
	IM_{UOL-PF}	0.02	0.00	0.31	0.00	0.00	0.00	4.08	0.00	3.73	0.00	3.66	0.00	0.13	0.00	0.21	0.00	0.35	0.00	1.30
	Total	**-1.29**	**-12.09**	**50.49**	**49.54**	**15.35**	**15.32**	**4.18**	**1.93**	**5.27**	**8.00**	**5.07**	**7.06**	**1.55**	**7.13**	**0.32**	**-0.82**	**3.28**	**-0.91**	**1.66**

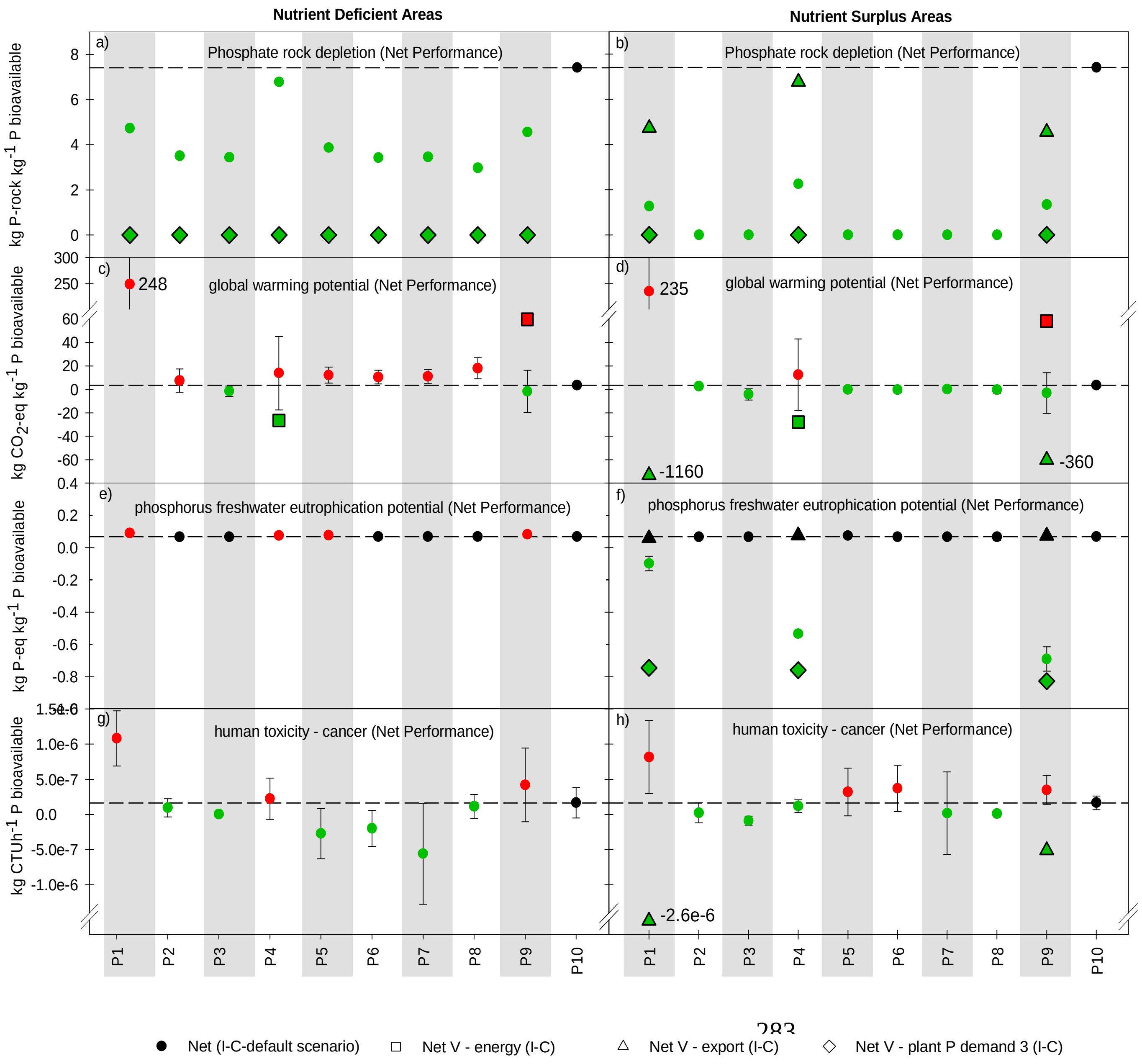
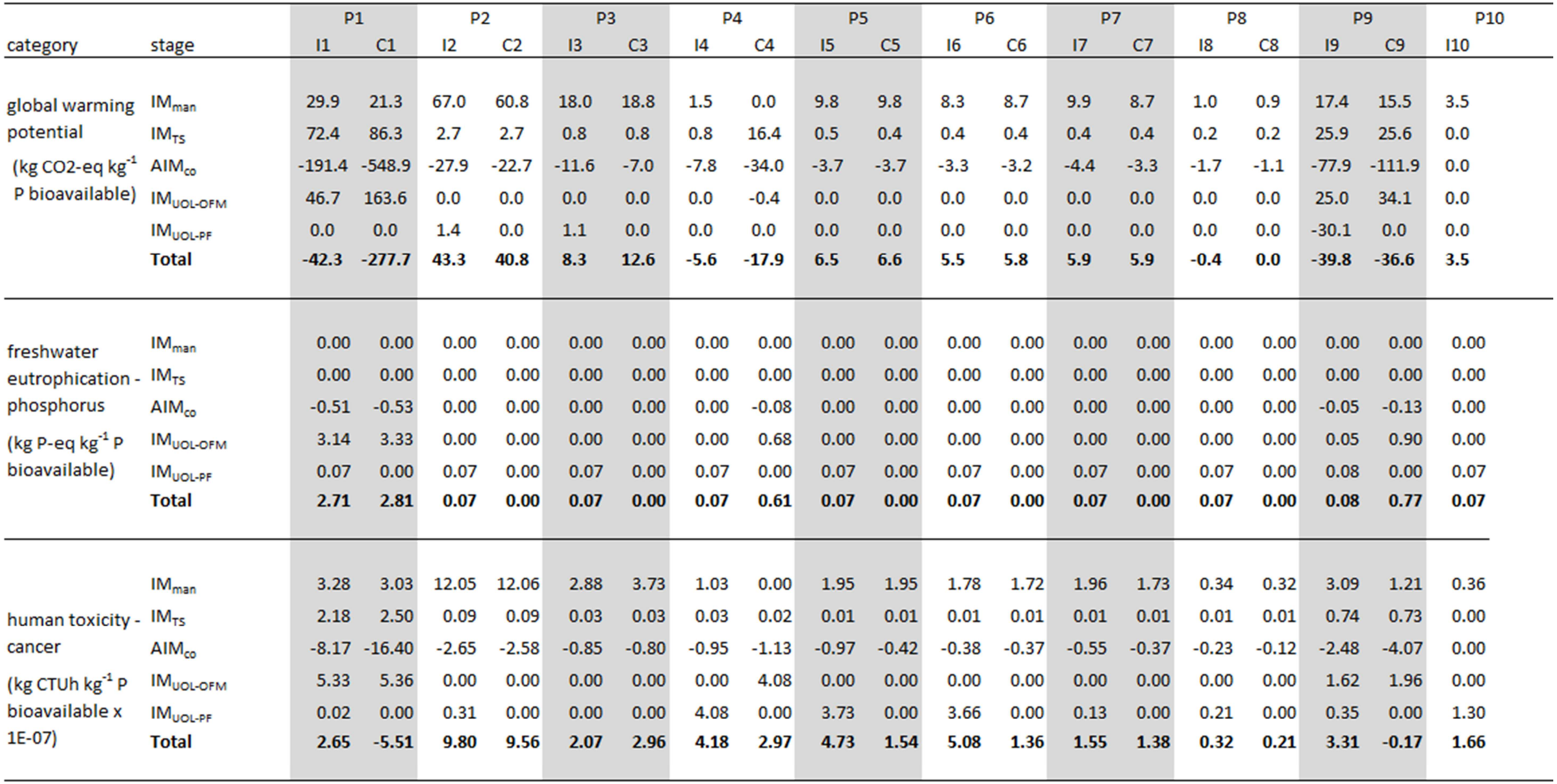


Figure 20: Aggregated net results for the impact categories phosphate rock depletion, global warming potential, freshwater eutrophication – P, and human toxicity cancer for the production options Px in nutrient-deficient areas (graphics on the left) and nutrient surplus areas (graphics on the right) as graphically presented in Figure 18.

NB: Green dots indicate benefits from the STRUBIAS production option compared to the production of the mined rock phosphate and processed P-fertiliser SSP (P10).
Red dots indicate burdens from the STRUBIAS production option compared to the production of SSP (P10).
Black dots indicate no net effects from the STRUBIAS production option compared to the production of the mined rock phosphate and processed P-fertiliser SSP (P10).
Squares, triangles and diamonds represent the outcome of the variant analysis for alternative assumptions on energy, manure exports, and plant P demands, respectively, as outlined in Section 8.11; only shown when result falls beyond the confidence interval for the default scenario as indicated by the error bars.

Table 14: Results of the life cycle analyses presented in disaggregated form for the different life cycle stages for the impact categories global warming potential, marine eutrophication – nitrogen, and human toxicity - cancer for the pathways Ix and Cx in <u>nutrient surplus areas (NSAs)</u> as graphically presented in Figure 18 (see Section 8.6.1 for explanations on the abbreviations)



		P1		P2		P3		P4		P5		P6		P7		P8		P9		P10
category	stage	I1	C1	I2	C2	I3	C3	I4	C4	I5	C5	I6	C6	I7	C7	I8	C8	I9	C9	I10
global warming potential (kg CO2-eq kg^{-1} P bioavailable)	IM_{man}	29.9	21.3	67.0	60.8	18.0	18.8	1.5	0.0	9.8	9.8	8.3	8.7	9.9	8.7	1.0	0.9	17.4	15.5	3.5
	IM_{TS}	72.4	86.3	2.7	2.7	0.8	0.8	0.8	16.4	0.5	0.4	0.4	0.4	0.4	0.4	0.2	0.2	25.9	25.6	0.0
	AIM_{co}	-191.4	-548.9	-27.9	-22.7	-11.6	-7.0	-7.8	-34.0	-3.7	-3.7	-3.3	-3.2	-4.4	-3.3	-1.7	-1.1	-77.9	-111.9	0.0
	$IM_{UOL\text{-}OFM}$	46.7	163.6	0.0	0.0	0.0	0.0	0.0	-0.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	25.0	34.1	0.0
	$IM_{UOL\text{-}PF}$	0.0	0.0	1.4	0.0	1.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	-30.1	0.0	0.0
	Total	**-42.3**	**-277.7**	**43.3**	**40.8**	**8.3**	**12.6**	**-5.6**	**-17.9**	**6.5**	**6.6**	**5.5**	**5.8**	**5.9**	**5.9**	**-0.4**	**0.0**	**-39.8**	**-36.6**	**3.5**
freshwater eutrophication - phosphorus (kg P-eq kg^{-1} P bioavailable)	IM_{man}	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
	IM_{TS}	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
	AIM_{co}	-0.51	-0.53	0.00	0.00	0.00	0.00	0.00	-0.08	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	-0.05	-0.13	0.00
	$IM_{UOL\text{-}OFM}$	3.14	3.33	0.00	0.00	0.00	0.00	0.00	0.68	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.05	0.90	0.00
	$IM_{UOL\text{-}PF}$	0.07	0.00	0.07	0.00	0.07	0.00	0.07	0.00	0.07	0.00	0.07	0.00	0.07	0.00	0.07	0.00	0.08	0.00	0.07
	Total	**2.71**	**2.81**	**0.07**	**0.00**	**0.07**	**0.00**	**0.07**	**0.61**	**0.07**	**0.00**	**0.07**	**0.00**	**0.07**	**0.00**	**0.07**	**0.00**	**0.08**	**0.77**	**0.07**
human toxicity - cancer (kg CTUh kg^{-1} P bioavailable x 1E-07)	IM_{man}	3.28	3.03	12.05	12.06	2.88	3.73	1.03	0.00	1.95	1.95	1.78	1.72	1.96	1.73	0.34	0.32	3.09	1.21	0.36
	IM_{TS}	2.18	2.50	0.09	0.09	0.03	0.03	0.03	0.02	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.74	0.73	0.00
	AIM_{co}	-8.17	-16.40	-2.65	-2.58	-0.85	-0.80	-0.95	-1.13	-0.97	-0.42	-0.38	-0.37	-0.55	-0.37	-0.23	-0.12	-2.48	-4.07	0.00
	$IM_{UOL\text{-}OFM}$	5.33	5.36	0.00	0.00	0.00	0.00	0.00	4.08	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	1.62	1.96	0.00
	$IM_{UOL\text{-}PF}$	0.02	0.00	0.31	0.00	0.00	0.00	4.08	0.00	3.73	0.00	3.66	0.00	0.13	0.00	0.21	0.00	0.35	0.00	1.30
	Total	**2.65**	**-5.51**	**9.80**	**9.56**	**2.07**	**2.96**	**4.18**	**2.97**	**4.73**	**1.54**	**5.08**	**1.36**	**1.55**	**1.38**	**0.32**	**0.21**	**3.31**	**-0.17**	**1.66**

8.9.1 Global warming potential

The impact on the global warming potential is largely **dependent on the regional situation where the P-recovery and nutrient return to agricultural land takes place**. For *NDA*, producing and using STRUBIAS materials is generally associated with a net positive contribution to the global warming potential, except for P3 (WASSTRIP followed by struvite precipitation) and P9 (pyrolysis of pig manure). The contribution to the global warming potential is often larger than for mined phosphate rock and processed P fertilisers. The main reason for this observation is that STRUBIAS production processes, with the exception of struvite precipitation at biological wastewater treatment plants, involve a reduction or removal of the bio-available nitrogen (and to a lesser extent organic C) which is returned to agricultural land. This is not the case in the counterfactual scenarios. Therefore, the **avoided mineral fertiliser production (through the Haber-Bosch process) and peat extraction (necessary to account for C losses)** is generally much higher (i.e. incurring relatively more savings on global warming potential) in the counterfactual scenarios. These observations indicate that reducing and removing bio-available N from the input materials that are returned to land in the STRUBIAS production scenarios has a significant impact on the global warming potential. With the exception of P3, STRUBIAS manufacturing processes in NDA also generally contribute to increasing the global warming potential, often due to energy requirements for the additional operations at the manufacturing site (e.g. incineration). For **NSA**, the counterfactual scenarios for the different input materials involve the co-incineration of sewage sludges. For this regional situation, nitrogen and organic C are thus not returned to land in either the induced or the counterfactual scenario for thermal oxidation materials & derivates. Minor amounts of nitrogen are even returned to agricultural land in the induced scenario for precipitated phosphate salts & derivates, but not in the counterfactual scenario of incineration. Hence, **benefits from N** returned to the field are small overall in induced and counterfactual scenarios for NSA, and do not influence the net results for the production options. For this reason, the net impact on the global warming potential is generally lower compared to NDA; many scenarios show an overall impact close to zero. On some occasions, a lower contribution to global warming relative to mined phosphate rock and processed P fertilisers was observed. This shows the crucial importance of considering local context and nutrient management to capture the environmental consequences associated with these technological pathways.

For most options, **the use-on-land phase of the STRUBIAS material does not significantly affect the overall impacts for global warming due to their low N content, and thus N_2O emissions**. In the event that the feedstock is not thermally processed in full, **the use on land of the rest fraction** (e.g. rest sludge after struvite separation), however, contributes significantly to the global warming potential, mostly due to **N_2O emissions**. For P9, the **pyrolysis of pig manure**, the return of the STRUBIAS material to land is associated with a negative contribution to global warming due to the assumed stability of the C contained in the material incurring long-term C sequestration. According to the scenario modelling applied here, this effect is, however, partly neutralised because of avoided energy and nitrogen fertiliser production in the counterfactual scenario. Nonetheless, on the assumption that 90%

of the C contained in the pyrolysis material remains in the soil after 100 years, a net saving is still observed for global warming potential for P9.

Relative to the **counterfactual scenarios of the direct landspreading of unprocessed and digested biogenic materials, STRUBIAS production options generally provide a net positive contribution** to global warming, expect for the Wasstrip option (due to some avoided nitrogen fertiliser production) and the pig manure pyrolysis option (due to the increased stability of the C contained in the material after soil application). This is indicated by the negative net values for pathways where (co-)incineration is not applied in the counterfactual scenario.

8.9.2 Phosphorus eutrophication

The use-on-land phase is the main process determining the impacts for potential freshwater P eutrophication. Phosphorus that is not taken up by the plants in mostly stored in soils that are not P-saturated. Hence, although the P in some STRUBIAS materials may be more bio-available than the P in the feedstock applied (e.g. sewage sludge), the implementation of STRUBIAS pathways will have **no major influence on the freshwater P eutrophication in NDA**. In NSA, the impacts on P eutrophication are dependent on the feedstock material applied. For sewage sludge and meat and bone meal, it is assumed that these materials are incinerated and disposed of in the counterfactual scenario and thus do not contribute to P eutrophication of freshwater bodies. Hence, the production and use of STRUBIAS materials from these feedstocks will not impact upon P eutrophication. For manure-derived pathways, the counterfactual scenario effectively contributes to P eutrophication as the excess P in manure is typically leached or lost through run-off. Hence, **using manure for the production of P-fertilisers that can be shipped and applied where the P loss potential is minor will effectively contribute to the mitigation of P eutrophication**.

8.9.3 Human health

The impact on human health is largely **dependent on the STRUBIAS production option applied and the regional situation where the P-recovery and nutrient return to agricultural land take place**. Relative to the production and use of mined phosphate rock and processed P fertiliser, the impacts of STRUBIAS materials are scattered, with better human health impacts for some options and worse human health impacts for others. Similarly, the comparative safety assessment relative to mined phosphate rock and processed P fertilisers is dependent on the regional situation; for P6, for instance, improvements for human health are observed in NDA, but mined phosphate rock and processed P fertilisers perform better for human health protection in NSA.

For **precipitated phosphate salts**, no net impacts were observed for the use-on-land phase under NDA and for the salts derived from manure (P1) in NSA. This implies that for these scenarios the bio-available P that is used on land has a significantly lower impact than mined phosphate rock and processed P fertilisers. Yet, one should bear in mind that the precipitation process is a separation process that produces a P-fertiliser that is low in contaminants, but that the contaminants end up in the rest fraction. When the rest fraction is incinerated (i.e. NSA with sewage sludge as input material), a negligible impact from the use-on-land phase for the options P2 and P3 is observed, with an overall lower net contribution than for P10 (SSP production and use on land) due to the decreased metal (especially Cd) concentrations in the struvites. This makes the avoided fertiliser production the life cycle stage that contributes the most for precipitated phosphate salts. The substitution effect of NK fertilisers is typically higher in the counterfactual scenario where landspreading of other fertilising materials generates more savings in this respect. However, the net difference I-C is always smaller than the total human health impact of SSP in production option P10. For P1, the impact on human health is higher for the induced scenario than for the counterfactual one, resulting in a high positive contribution of the option to human health impacts. The main reason is related to the avoided emissions from mineral N fertiliser application in the counterfactual N scenario.

For thermochemical conversion processes resulting in the formation of **thermal oxidation materials & derivates and pyrolysis & gasification materials** (P4-P9), the results are dependent on the levels of contaminants in the STRUBIAS materials. Some impacts are also associated with the manufacturing stage, with effects being more pronounced for the production options when no thermochemical conversion takes place in the counterfactual scenario. Specific production options that involve a partial (P5) or almost complete (P7) removal of the metals present in the input material, followed by their disposal in landfills (i.e. final sink), as expected perform better than others (P4, P6, P8 and P9). For sewage sludges as input materials, the best environmental benefits are achieved for NDA as the ratio of metals to bio-available P is always lower in the induced scenario than in the counterfactual one. For NSA, the human health performance depends even more so on the metal content in the STRUBIAS materials. In general, the relative concentrations of metals in the STRUBIAS materials and the mined phosphate rock and processed P fertilisers are mirrored in the overall impacts for human health. Production options that involve a removal of the metals present in the input material to levels below the metal contents in mined phosphate rock and processed P fertilisers perform better for human health, and vice versa. Benefits for human health are achieved at all times when the STRUBIAS manufacturing production option involves metal removal (P5 and especially P7), whereas impacts for STRUBIAS materials that rely on the mixing of sewage sludge ashes with acids to improve their P-bio-availability (P6) are situation-specific, i.e. dependent on the counterfactual treatment.

Relative to the counterfactual scenarios of **the direct landspreading of unprocessed and digested biogenic materials**, STRUBIAS production options can also improve **the protection of human health by effectively removing biological pathogens, pharmaceutical and personal care products, and other persistent and emerging organic pollutants**.

8.10 Production cost

The production costs for the induced and counterfactual pathways are presented in disaggregated form in Table 15 and Table 16 for nutrient-deficient areas and nutrient surplus areas, respectively.

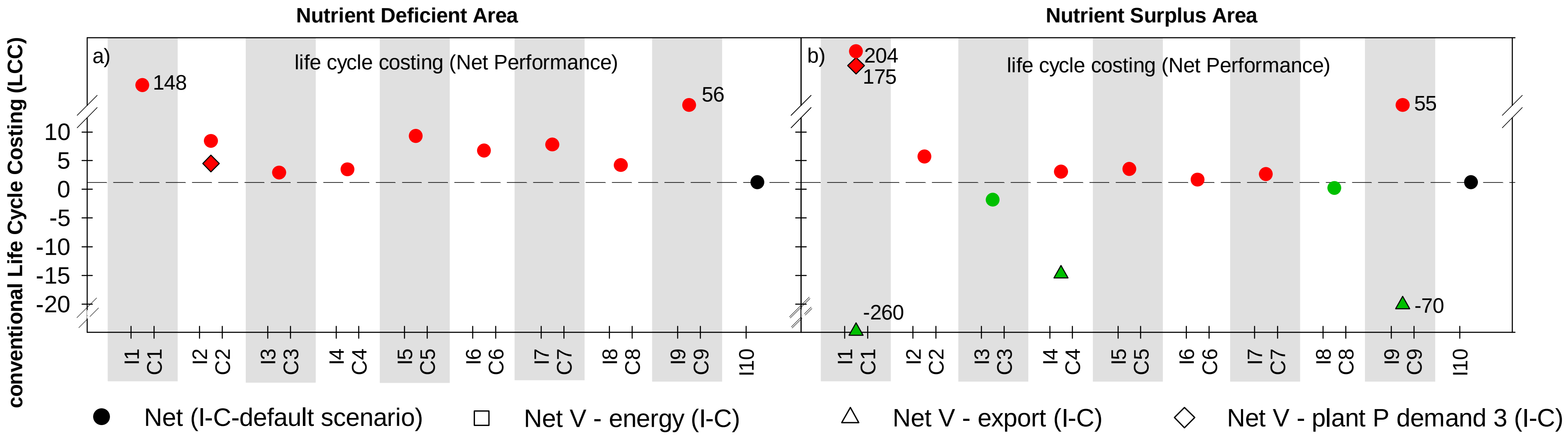
Table 15: Results of the life cycle cost analyses presented in disaggregated form for the different life cycle stages for the pathways Ix and Cx in <u>nutrient-deficient areas (NDAs)</u> as graphically presented in Figure 18 (see Section 8.6.1 for explanations on the abbreviations)

		P1		P2		P3		P4		P5		P6		P7		P8		P9		P10
category	stage	I1	C1	I2	C2	I3	C3	I4	C4	I5	C5	I6	C6	I7	C7	I8	C8	I9	C9	I10
production cost (EURO kg^{-1} P bioavailable)	CAPEX$_{man}$	158.2	2.9	24.0	20.5	8.3	6.3	5.5	0.0	8.8	3.3	7.2	2.9	7.2	2.9	2.9	0.0	35.1	3.4	0.1
	OPEX$_{man}$	41.8	21.0	14.1	10.7	3.0	3.3	2.4	0.0	4.6	1.7	3.2	1.5	4.4	1.5	1.1	0.0	13.7	1.4	1.0
	COST$_{TS}$	152.5	214.6	6.9	7.0	2.2	2.2	2.0	1.8	1.1	1.1	1.0	1.0	1.0	1.0	0.5	0.4	59.2	58.4	0.1
	savings$_{co}$	-14.1	-52.2	-28.7	-30.4	-8.2	-9.4	-7.9	-3.3	-4.0	-4.9	-3.5	-4.3	-3.7	-4.4	-1.5	-1.9	-5.2	-16.1	0.0
	Total	**338.4**	**186.2**	**16.2**	**7.9**	**5.3**	**2.4**	**2.0**	**-1.5**	**10.5**	**1.3**	**7.8**	**1.1**	**8.9**	**1.1**	**3.1**	**-1.5**	**102.8**	**47.1**	**1.2**

Table 16: Results of the life cycle cost analyses presented in disaggregated form for the different life cycle stages for the pathways Ix and Cx in <u>nutrient surplus areas (NSAs)</u> as graphically presented in Figure 18 (see Section 8.6.1 for explanations on the abbreviations).

		P1		P2		P3		P4		P5		P6		P7		P8		P9		P10
category	stage	I1	C1	I2	C2	I3	C3	I4	C4	I5	C5	I6	C6	I7	C7	I8	C8	I9	C9	I10
production cost (EURO kg^{-1} P bioavailable)	CAPEX$_{man}$	179.7	2.9	48.0	45.3	12.6	14.0	5.5	0.0	8.8	7.3	7.2	6.5	7.2	6.5	0.5	0.8	35.1	3.4	0.1
	OPEX$_{man}$	46.9	21.0	19.7	16.5	4.6	5.1	2.5	0.0	4.6	2.7	3.2	2.4	4.4	2.4	1.1	0.3	13.7	1.4	1.0
	COST$_{TS}$	152.3	179.1	6.2	6.4	2.0	2.0	2.0	1.8	1.1	1.0	1.0	0.9	1.0	0.9	0.5	0.4	59.1	58.4	0.1
	savings$_{co}$	-12.5	-40.6	-24.8	-24.6	-7.6	-7.6	-7.9	-2.7	-4.0	-4.0	-3.5	-3.5	-3.7	-3.5	-1.5	-1.1	-5.2	-15.9	0.0
	Total	**366.3**	**162.3**	**49.2**	**43.5**	**11.6**	**13.5**	**2.2**	**-0.8**	**10.5**	**7.0**	**7.8**	**6.2**	**8.9**	**6.3**	**0.6**	**0.4**	**102.7**	**47.3**	**1.2**

The aggregated net results (I-C) for production options P1-P10 are presented in Figure 21.



NB: The open diamonds represent case scenarios where manure is exported from the country of production. Here, transport distances of 500 km are assumed between the site of production of the raw input materials and the sites where the processed materials (including both STRUBIAS materials and other fertilising materials) are applied on land.

Green dots indicate savings from the STRUBIAS production option compared to the production of the mined rock phosphate and processed P-fertiliser SSP (P10).

Red dots indicate additional costs from the STRUBIAS production option compared to the production of the mined rock phosphate and processed P-fertiliser SSP (P10).

Squares, triangles and diamonds represent the outcome of the variant analyses for alternative assumptions on energy, manure exports, and plant P demands as outlined in Section 8.11; only shown when result falls beyond the confidence interval for the default scenario as indicated by the error bars.

Figure 21**: Aggregated net results for the production cost in <u>nutrient-deficient areas</u> (left) and <u>nutrient surplus areas</u> (right) as graphically presented in Figure 18 (default scenarios with low transport distances of 25 km, purple dots).**

Overall, the results indicate that **the production of STRUBIAS materials through one of the production options is more costly than the production of SSP derived from mined phosphate rock**, except for P3 in NSA.

For precipitated phosphate salts, **P3 results in net cost savings in NSA, even if the return of bio-available P to land is not considered** (net results < 0). This is because reduced sludge volumes will have to be co-incinerated. The net effect on CAPEX favours the induced scenario because the increase in capital costs due to the installation of additional units (e.g. struvite reactor, WASstrip unit) are lower than the savings gained in reduced capital investments for other units (incineration, anaerobic digestion, dewatering). The OPEX is, however, somewhat increased for the wastewater treatment operators due to the additional labour required to operate the supplementary units. For scenario P2, the increased investment and operational costs are, however, not offset as the impact on sludge volumes and polymer needs is relatively small based on our data. For NDA, the production cost for these options (P2 and P3) is higher because the additional investment and operational cost required in the induced scenario is not compensated by any economic returns from the low-cost material handling in the counterfactual scenario (i.e. landspreading of digested sludge). Note, however, that in P2 and P3, the cost savings due to reduced maintenance (e.g. pipe clogging) at the wastewater treatment plant have not been taken into consideration in this assessment. For P1, the high production cost is mostly caused by the high capital cost of the aerobic reactor and struvite precipitation costs that are very high relative to the volumes of materials that are currently being processed.

For **thermal oxidation materials & derivates**, the determining factor for the production costs is the implementation of additional processing steps (incineration, acidulation). For P4, CAPEX and OPEX from the incineration process mainly determine the cost of the production option as these manufacturing stages are not in place in the counterfactual scenario of landspreading of the dried poultry litter. The supplementary costs are partly offset by the increased savings from the energy recovery in this process, but the production cost for P4 is still higher than for mined phosphate rock and processed P fertilisers. The supplementary incineration plus acidulation steps in the induced scenarios relative to the counterfactual scenarios (landspreading of digested sludge) also imply significant additional costs for options P5, P6 and P7 in NDAs. The production cost for options P5, P6 and P7 are reduced in NSA as here the counterfactual scenarios involve co-incineration; supplementary costs are due to the shift from co-incineration to mono-incineration and the supplementary re-burning (P5) and acidulation process (P6, P7) for the induced scenario compared to the counterfactual scenario. In any case, these supplementary costs are higher than the production cost for P10 (SSP; EUR 1.2 kg^{-1} bio-available P; mostly originating from OPEX including the source materials). The differences between options P5, P6 and P7 are mainly due to the market price of the chemicals used (e.g. sodium sulphate, lime, sulphuric acid, hydrochloric acid). Note that for this assessment we have assumed the use of primary raw materials, whereas the use of industrial by-products instead of ‘primary’ chemicals is a common practice in the fertilising sector. For P8 (meat and bone meal mono-incineration, followed by acidulation), significant CAPEX and OPEX are implied. Moreover, savings from the co-products (avoided energy and fertiliser production) for the induced scenario relative to the counterfactual scenario are small. The cost of producing N-fertilisers that are avoided in the counterfactual scenario is only slightly lower than any revenues obtained from the energy recovery potential of the meat and bone meal.

The production cost for option P9 – the production of **pyrolysis materials** from pig manure – are high. The main reason is the supplementary investment to build and operate the pyrolysis facility. In the developing sector, the CAPEX and OPEX are very high relative to the amount of material processed. It remains unknown to what extent these costs can be reduced when the market develops, and potentially the unit cost will decline as the production volume increases (scale effects). Also, the loss of bioavailable N from the manure and the absence of recovered energy as a marketable product during the pyrolysis process puts this option at a disadvantage compared to the counterfactual scenario of anaerobic digestion.

8.11 Uncertainty analysis and variant analysis

A Monte Carlo analysis was performed to determine the uncertainty related to the environmental and human health impacts for the different pathways by taking into consideration the variation in input material composition, uncertainty in process manufacturing steps, and emissions during storage and use on land. For the impact category **global warming potential**, the uncertainty analyses revealed that the potential impacts of the pathways generally lie within the same range, with the exception of pathway P1 for which significantly greater burdens were observed. The feedstock composition (e.g. methane production potential) and the use on land phase (N_2O emissions) are the main contributing processes to the overall uncertainty (data not shown). The greater confidence interval observed for the pathways that rely on manure as feedstock (P1, P4, P9) are mainly explained by the greater N content of the feedstock applied on land in the counterfactual scenario (resulting in a greater uncertainty on the N_2O emissions) and the more variable carbon and nutrient content of the input material. For **freshwater P eutrophication potential**, the analysis indicated that transforming manure into P-fertiliser through STRUBIAS production processes in NSA results in an effective and significant reduction in the eutrophication potential, especially for pathways P4 and P9 which show a high P-recovery potential. For **human health – cancer**, the relatively large uncertainty observed for the different pathways is for most pathways almost exclusively explained by the variation in metal content in the feedstocks including the secondary raw materials as well as in the phosphate rock. Pathway P3 performs significantly better than pathway P10, whereas pathway P1 has the most negative impact upon human health, mainly due to the loss of bio-available N in the induced scenario.

The uncertainty of the individual results for the different impact categories in the default modelling approach for NDA and NSA are based on default assumptions for energy (Dutch energy mix), transport from site of collection to factory gate and land application site (25 km), and plant P demand (40 kg P_2O_5 ha^{-1} yr^{-1}). For the **variant analyses**, **these default assumptions were modified as follows:**

- natural gas as energy source (V-energy - represented by a square in the Figures);
- 500 km transport for liquid and solid manure fractions in NSA (V-export; simulating a situation where manure excess is evaporated to a material with 20-23% dry matter and transported over a long distance to a nutrient-deficient area - represented by a triangle in the Figures); and
- no plant P-demand (V-plant P demand; simulating a local situation of zero plant P requirements due to excessive long-term build-up of P in soils, see Tóth et al. (2014) -- represented by a diamond in the Figures).

Firstly, the variant analysis indicated **that applying an energy mix that is less green** and includes more fossil fuels (here represented by natural gas) may alter the impacts on global warming for specific pathways (i.e. P4 – poultry litter combustion, and P9 – pyrolysis of pig manure). Pathway P4 produces substantial amounts of renewable energy, and the benefits of this process in terms of greenhouse gas emissions become even more pronounced when this energy substitutes fossil fuels such as natural gas. In contrast, pathway P9 requires a substantial amount of energy during the production process (composting followed by pyrolysis), and if this energy requirement is met through the use of an increased amount of fossil fuels, the overall performance of this pathway for global warming is worsened.

Secondly, the variant analyses focused on **specific regional settings of excessive animal density that result in the need to export nutrients** due to the local surplus of nutrients in biogenic materials such as manure, and possible stimuli or legislative requirements that result in positive impacts of such exports on local ground- and surface water quality. This situation is, for instance, applicable in the Netherlands where strict P-application quotas apply for farmers. In such cases, (liquid and/or slurry) manure fractions may be concentrated (e.g. through evaporation) to enable their long-distance export. The latter counterfactual scenario is assumed in the variant analyses for ‘export’. The results indicate that, in such conditions, the implementation of STRUBIAS for the local processing of these materials has a clear positive impact on global warming and human health, mainly for manure fractions with a high moisture content (calf and pig manure, P1 and P9). Also, the implementation of the STRUBIAS process is much more cost-effective than the counterfactual scenario of evaporation and long-distance transport to a nutrient-deficient region. Local processing of poultry litter to ashes is also a more profitable solution than the drying and exporting of the original feedstock. This analysis thus confirms that STRUBIAS production processes can be an effective solution to avoid the long-distance transport of biogenic materials with a high moisture content.

Thirdly, the variant analysis confirms that greater benefits on phosphate rock depletion and freshwater P eutrophication potential can be achieved in the event that the biogenic feedstock is not applied in an effective manner in the counterfactual scenario. **Applying biogenic feedstocks on agricultural land that is P-saturated and to plants that do not have a need for P** to develop and mature (e.g. a situation characteristic for some regions in Belgium, the Netherlands and Germany; see Tóth et al. (2014)) leads to reduced P use efficiency, and P leaching and P run-off. Therefore, transforming these biogenic materials into concentrated STRUBIAS P-fertilisers effectively contributes to reduced phosphate rock depletion and freshwater P eutrophication.

8.12 Life cycle and life cycle cost conclusions

The analysis confirms that the development of STRUBIAS production pathways reduces the dependence on the finite primary raw materials, but that the net impacts are highly dependent on the current counterfactual use of the biogenic input material applied. Given that biogenic materials already contribute to a variable extent to the return of bio-available P in NDA, these regions provide the least opportunities to mitigate risks associated with phosphate rock depletion through STRUBIAS pathways. **Nutrient surplus areas (NSA) provide greater opportunities for the development of STRUBIAS pathways due to the increased savings on phosphate rock depletion**.

Phosphorus nutrient recovery is often claimed to provide co-benefits for the environment and human health, but insightful and all-encompassing studies that assess these effects are still limited. This analysis indicates that the potential of STRUBIAS materials to contribute to these co-benefits relative to mined phosphate rock and processed P-fertilisers is dependent on the production option applied and the regional situation. The main mechanisms that **contribute positively** to the environmental and human health impacts relate to increasing the bio-availability of the nutrients contained in the biogenic input materials and reducing the metal therein contained. There are also **risks of adverse and unintended negative effects**, however, in processes involving the removal of other valuable materials from the biogenic input materials (e.g. nitrogen, organic C), the lack of removal of contaminants, and related to additional manufacturing steps that are associated with high chemical or energetic demands. This indicates that recovery processes in a circular economy should aim at maintaining the highest value of materials and products by isolating the target materials for recovery from the other valuable substances and contaminants during the manufacturing process, if possible.

A review of the P-recovery initiatives in the Europe indicates that most of the countries identified as leaders in this field are EU Member States, particularly those in western and northern Europe (see Chapter 14 and Section 7.1). From an economic perspective, this makes sense as **the supplementary cost of P-recycling is lowest in regions characterised by nutrient surpluses**. Based on the momentary evaluation, STRUBIAS production costs are for most options more expensive compared to mined phosphate rock and processed P-fertilisers. Nonetheless, options for a transition towards more circular economy approaches may include a mix of complementary instruments and approaches, including regulatory measures, economic incentives, education and awareness-raising, and targeted funding for innovation and research (WHO, 2018). Hence, the materialisation of STRUBIAS production options is not only cost-dependent, but is also subject to other decisions that impact on the need to process eligible input materials for STRUBIAS materials in a different manner relative to the current business-as-usual requirements, and further market stimulations. For example, stricter targets on water quality that restrict inputs of unprocessed manure in some vulnerable areas, stricter legislation on metal contents in sludges to be used as fertilising materials, stricter requirements to hygienise manure prior to landspreading, etc. Economic incentives to stimulate, for instance, the compliance with green energy targets or to disincentive the inefficient use of biological materials (e.g. landfilling) may further stimulate the materialisation of STRUBIAS pathways. Finally, nutrient recovery through STRUBIAS production processes is currently emerging in Europe and it is likely that scale effects will further reduce production costs once the industry develops further.

From a STRUBIAS **input material perspective**, **sewage sludge** is a likely candidate to be used as an input material due the lower production costs. Several production options have been developed that could provide combined improvements for the mitigation of global warming and eutrophication, and human health protection (e.g. P2, P3, P7). For **manure** as an input material for STRUBIAS production processes, the issue is that the raw feedstocks and their digestates are already a good fertiliser, with N and P being to a large extent bio-available and contributing to plant nutrition as long as the material is applied in a sustainable manner. The processing into STRUBIAS materials does not necessarily increase the bio-availability of the nutrients contained in the material, and might even result in losses of bio-available N from the material. As a result, the environmental costs for manure processing into STRUBIAS materials relative to the counterfactual management pathways are high in nutrient-deficient regions where manure can best be applied directly on land, if possible; it is the most direct return of nutrients and organic matter with the best impacts on the global warming and human health impacts. Therefore, the markets for STRUBIAS materials that apply manure as a feedstock are most likely to develop in regions characterised by a high livestock density and/or regions where manure exports are binding in legislation (e.g. the Netherlands). Nonetheless, the removal of specific contaminants in manure (e.g. biological pathogens, veterinary antibiotics) and/or the production of other primary products (e.g. renewable energy) through STRUBIAS production processes could also be a driver for the market growth of STRUBIAS derived from manure. As a matter of fact, it is believed that this mechanism will be the dominant driver for STRUBIAS production options from manure (see Section 7.3.4). Hence, **especially for manure, it is believed that STRUBIAS production pathways could form part of material cascades in a developing bio-economy sector in the EU**.

Overall, this analysis indicates that **the implementation of STRUBIAS pathways to transform biogenic materials into concentrated P-fertilisers can provide opportunities to maximise benefits on food security, food safety and environmental protection**. The extent of the impacts is, however, dependent on the regional situation and the STRUBIAS production pathway applied. The development of some pathways (P2, P3 and P7 in nutrient surplus areas) will simultaneously result in a reduction of the dependence on phosphate rock, reduced global warming impacts on, and increased environmental protection relative to the production and use of mined rock phosphate and processed P-fertilisers. The development of STRUBIAS pathways in nutrient-deficient areas and the realisation of specific pathways (P1, P4, P5, P6, P8 and P9) in nutrient surplus areas is associated with **trade-offs**, with potential benefits occurring at the expense of burdens for global warming potential or human health.

This implies that the implementation of STRUBIAS pathways is dependent on the needs and priorities of local stakeholders, and thus the nexus of costs and impacts upon human health, agronomic yields, water quality, energy balances, resource depletion, climate regulation and long-term food security. The exploitation and prioritisation of the corresponding services by stakeholders are largely context-specific, but STRUBIAS production processes can also provide opportunities here for addressing critical issues related to phosphate rock depletion, global warming, eutrophication, and potentially human health, or a combination of those.

9 Economic aspects

9.1 Sales prices of P-fertilisers

9.1.1 DAP free on board as a benchmark for P-fertiliser prices

Fertilisers, in the simplest of terms, have a very specific purpose: increasing crop yields per hectare. They achieve this purpose by compensating for any nutrient deficiencies in the soil, which could be due to a variety of reasons, from the geological nature of specific cropland areas, to weather events that remove topsoil nutrients, or again by bumper harvests that result in significant uptake of nutrients by plants themselves.

This simplified reasoning suggests that, in an ideal market with perfect information, the expenditure of the farmer on fertiliser application will be equal to the marginal yield gain expressed in monetary terms:

$$Expenditure^{i}_{fert\ application} = Price^{i}_{crop} * Area^{i}_{crop} * (Yield^{i}_{\mathrm{fert,crop}} - Yield^{i}_{nofert,crop}\ )$$

where Expenditure$_{\mathrm{fert\ application}}$: the expenditure of the farmer on fertiliser application, as determined by the cost of fertiliser purchase, fuel, time and machinery; $Price_{crop}$: the sales price for the crop as received by the farmer (EUR per tonne yield); $Area_{crop}$: the area of the cultivated crop (ha); $Yield_{fert,crop}$: the crop yield on fertilised areas (tonne yield per ha); and $Yield_{nofert,crop}$: the crop yield on unfertilised areas (tonne yield per ha).

This formula is very simple in its structure; however, it already suggests a significant complication, which is that the true value of fertilisers is specific to individual patches of land, their specific nutrient balances, the crops planted, and their reactivity to fertiliser application. Also, as fertilisers consist of various nutrients (primary nutrients being nitrogen (N), P and potassium (K)), the yield gain for each individual fertiliser will depend on the nutrient content of the actual fertiliser mix applied, and again the actual yield gain will depend not only on the nutrient considered, but on the balance for other nutrients and the form in which each is applied. Such a pricing model, albeit undoubtedly interesting, is far from the scope of this study and can only be replicated on a very site-specific scale with in-depth agronomic analysis.

A more meaningful and robust approach looks a few steps back in the **value chain**, as described in Figure 22. It is important to notice that sometimes all the agents involved are the same company, for example the Norwegian company Yara would act as a raw material supplier to its own manufacturing sites, and would sell its own products via its own distribution system, including providing services to farmers like soil sampling, agronomic analysis, and in some cases even direct application to the field. Individual companies will have various degrees of integration along the value chain.

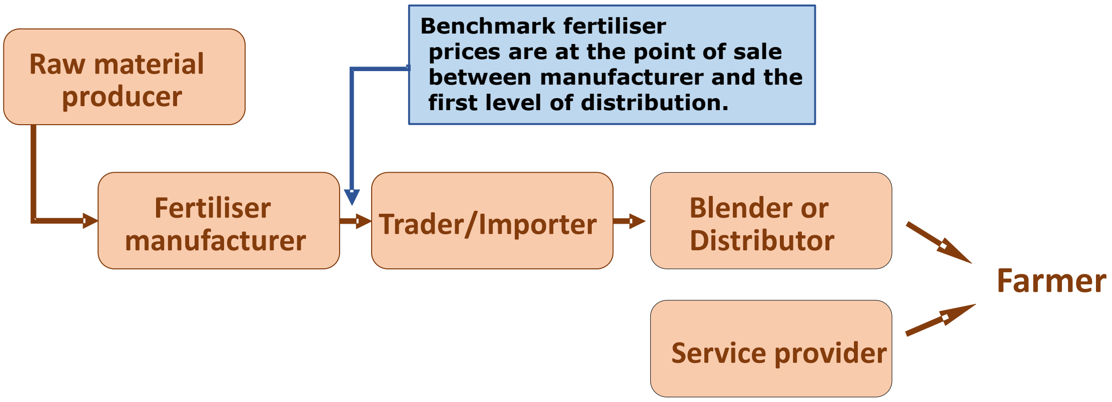


Figure 22: Schematic overview of the fertiliser value chain

When looking at the European market specifically, the most common structure sees a separation between fertiliser manufacturers and traders/importers, which in turn are in many cases distribution companies or blenders themselves. **For this reason, this work uses trade-based prices**: this allows prices to be identified more accurately, as these are observed at a consistent point. Moreover, focusing on trade values allows for the identification of product-specific prices: farmers would receive tailored formulations often bundled with various additives (e.g. micronutrients) and services, which makes farm-gate prices less transparent. In contrast to farmers, traders and importers tend to focus on more ‘commoditised’ varieties, such as diammonium phosphate (DAP), which is a sufficiently standardised product globally. Prices are given for **free on board (FOB) agreements**. This implies that the seller is obligated to deliver the goods to a destination for transfer to a carrier designated by the buyer. The location designation in the FOB trade agreement is the point at which ownership is transferred from the seller to the buyer; it is most often the port located in the country of phosphate rock extraction.

Generally speaking, traded materials at this benchmark **would then be considered a raw material for further processing**, for example in the form of bulk blends (for blenders) or physical NPK compounds (for fertiliser manufacturers). Straight application of commoditised products is also practised, but blends and compounds account for the majority of actual application to the soil.

For the price setting of P-fertilisers, it is important to determine which product acts as the ‘P_2O_5 benchmark’ against which other products are priced. **On a global basis, DAP is usually acknowledged as the main price-setting product, due to its larger traded volumes, which makes prices relatively more transparent.** Also, DAP ranks amongst the fertilisers with the highest values of ‘total nutrient content’ ($N+P_2O_5+K_2O$) across commoditised products, meaning that it minimises freight costs per tonne of nutrient compared to most alternatives. Generally speaking, markets in western Europe tend to feature DAP as the main price-setting product, while eastern Europe tends to show a preference for MAP instead. Fertecon's market assessment for different EU Member States (France, Germany, Italy, Poland and Romania) indicated that, expressed on a P-basis, **trade-based price differences between DAP and TSP are relatively minor (~ < 10%), whereas the**

sales price for SSP is slightly higher (~ + 20%). This difference can be explained by the increased logistical cost associated with the transport of SSP (low P-content; 16-22% P_2O_5 versus 46% P_2O_5 in TSP/DAP). Trade volumes for SSP are relatively low due to the inefficient transport logistics; therefore, local producers face less competition, and therefore can push prices to the upper end of any reasonable range.

9.1.2 Historical and forecast overview on P-fertiliser prices

All prices are given for the conventional farming sector in the EU-28. P-fertilisers sold to specific niche sectors (e.g. fertiliser use in horticultural application, home gardening and growing media) might be associated with higher sales prices, but such price settings are not covered in this document.

Through the **1990s and into the early 2000s, prices** for P-fertilisers, using DAP as a proxy, could be categorised as being low. Margins in the industry were generally poor – in most years a range of 5-10% would be typical. This enabled producers to trade **moderately profitably**, but critically there was very **little incentive for existing producers to invest in new capacity**, and no incentive at all for new producers to enter the market. As a consequence, there was only limited investment in P-fertiliser capacity, mostly by state-owned enterprises, and closures in other parts of the world ensured that there was no major increase in overall productive capacity.

Following the Asian economic crash in 1997, economic growth around the world was strong over the 1997-2007 time period. Improving standards of living means that people generally consume more food (especially in developing nations) and can also afford more high-quality food especially proteins such as meat. Demand for cereals and other staple crops increases, both as food for humans and for animals which are then consumed by humans. **Increased demand for crops increased the use of fertiliser, which meant that fertiliser and P-fertiliser utilisation rates increased, and prices increased**.

The increase in margins during the 2002-2007 period began to encourage producers to consider projects and commence the building of P-fertiliser manufacturing construction sites. To build new phosphate fertiliser capacity will generally take between 36 and 48 months once construction begins, and clearly there will normally be a few years of planning required in advance, especially if it is for a greenfield development (as opposed to expanding an existing site). Therefore, **during the super-heating of the global economy in 2007 and 2008, demand exceeded the industry's ability to supply, and consequently prices increased significantly.** In 2004, the average price of DAP on the basis of FOB Morocco was USD 235/t (EUR 189/tonne); in 2007, it was USD 425/t (EUR 310/tonne) and in 2009 it was USD 1 090/t (EUR 782/tonne). With the economic crash in late 2008 reality reasserted itself, and in 2009 price levels declined. Producers tried to hold on to gains for a period, but over 2009 and 2010 it was very clear that if they were to do so it would be at the expense of production volumes.

From 2011 onward there have also been the commissioning of projects which were planned in the mid-years (and later) of the 2000s. The other key (and related) structural change in the market has been the build-up of production capability in China, which has meant that China has gone from a net P-importer in the late 1990s to being a significant P-fertiliser exporter. Although capacity has been decommissioned in less cost-efficient locations, P-fertiliser production has returned to a position of net surplus, with a corresponding reduction in prices. Figure 23 shows the price evolution for DAP and rock on the basis of FOB Morocco / North Africa. The indicated breaks in the prices due to the absence of a market; producers wanted prices buyers were unprepared to pay.

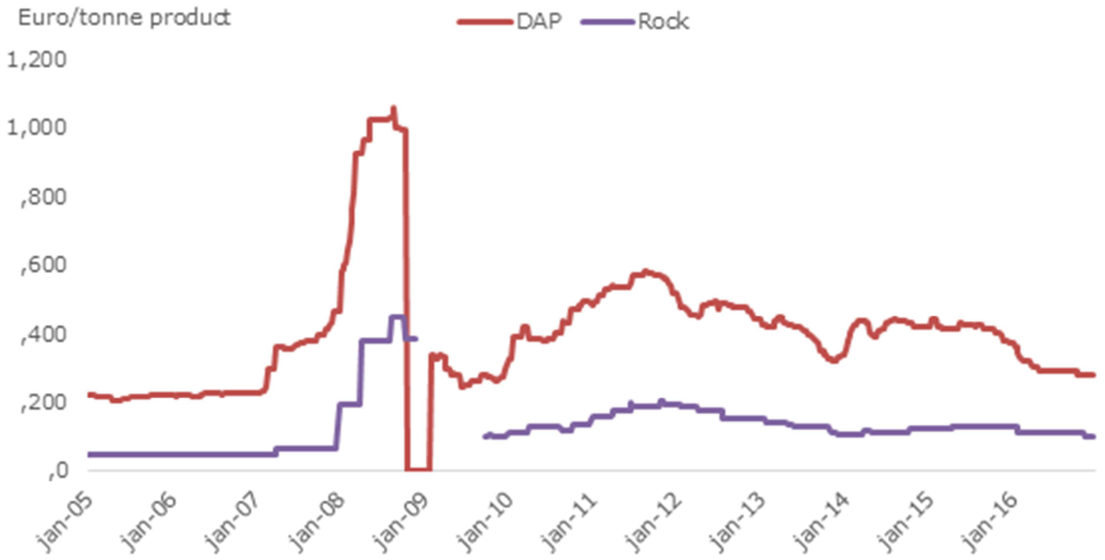


Figure 23: Price evolution for diammonium phosphate (DAP) and phosphate rock (EUR/tonne product; assuming an average exchange rate of EUR 1.2/USD; prices given for free on board agreements) (Source: Fertecon)

Looking forward, Fertecon is expecting the market to maintain the current (2017) level of balance through to 2025, i.e. that growth in demand will be offset by increased capacity to supply, and therefore there will not be any significant improvement on utilisation rates in the industry. This means that price levels will be maintained at levels which provide a margin for the major producers, but will also progressively eliminate some of the marginal capacity. Therefore, DAP prices are estimated to increase moderately from 2017 onwards with an average price increase of 1-2% per year (green line, Figure 24).

The price per tonne P in mined rock phosphate and processed P-fertilisers is evaluated based on the DAP prices. For the calculations, the value of N present in DAP is subtracted from the DAP price, after which the N-devaluated DAP is expressed on a P-basis. The N in DAP is evaluated based on the ammonia price (USD 381 per tonne ammonia-N, or USD 69 for the 18% N in DAP). Also, a constant EUR 1.2/USD exchange rate is used in these calculations. **A price of EUR 988 per tonne of P FOB is indicated for the year 2017.**

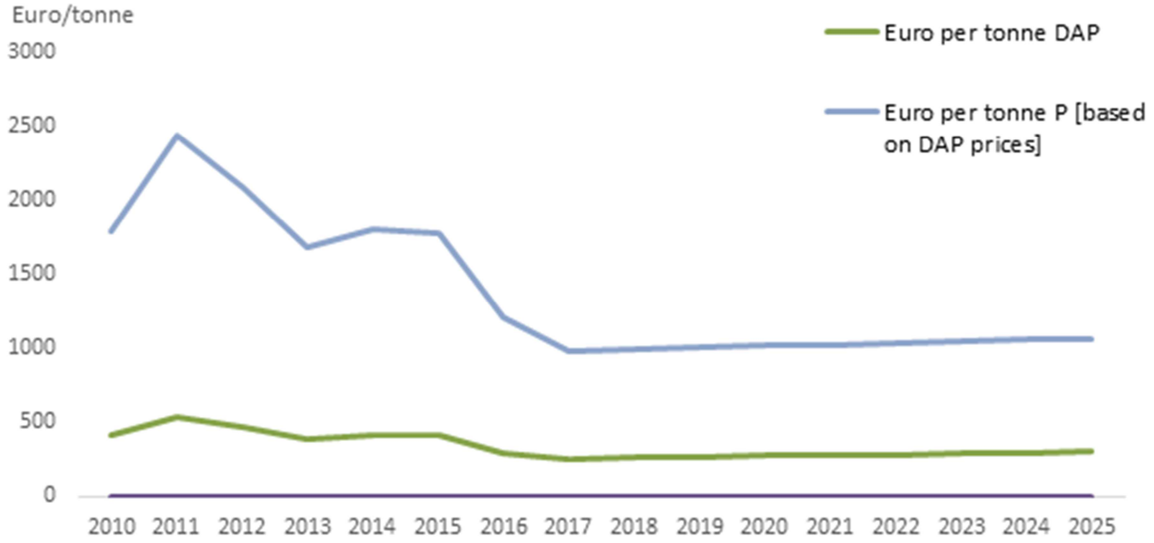


Figure 24: Fertecon DAP Price Forecast to 2025 expressed per tonne of DAP and per tonne of P (FOB Morocco; exchange rate of EUR 1.2/USD applied by JRC)

The 2025 FOB forecast is also within the range of prices experienced prior to the 2007-2009 super-cycle, when adjusted for inflation. This is a relevant comment: there are many bodies that describe current price levels as low. This is factually accurate in comparison to recent years, but not in the historical context. In comparison to the 1990-2005 period, **the current price level might be better described as 'normal'.**

9.1.3 Price setting for STRUBIAS materials on the market

Given the lack of legal outlets for many STRUBIAS materials in many EU Member States, **these materials are at present not placed on the market under a stable reference framework.** At present, the volumes of STRUBIAS materials on the internal fertiliser market are very low, and mostly delivered to specific niches of the internal agricultural market. Depending on the legal framework, market access is only granted to specific STRUBIAS materials, mostly under national legislation. Therefore, it is impossible to provide a meaningful evolution of the sales prices for a given fertilising product containing recovered materials in the EU observed in order to determine the main factors affecting prices of STRUBIAS materials and their relative importance. Moreover, the answers received in response to the JRC questionnaire indicated that **in view of confidentiality, it is not appropriate to communicate sales prices in publically available reports and that they do not provide added value to this report**.

Rather than giving an overview of the expected sales prices, this section provides **a general benchmark** for the economic valuation of STRUBIAS materials and incineration ashes that can be used as raw materials by the fertiliser industry. Sales prices generally vary broadly across the different market sectors targeted by a producer. This section **focuses on the**

drivers that can be expected to affect sales prices of materials for use in conventional agriculture.

Values are presented for Cost and Freight (CFR) **agreements** with downstream users, and thus include the costs of shipment to **the downstream processor or user (**Table 17**)**. For the producer of STRUBIAS materials the potential revenues will be lower, as any distribution costs will need to be deducted. Particularly with regard to materials of low economic value (e.g. incineration ashes as an intermediate raw material within the P-fertiliser production chain), it is important to recognise that the indicated values are CFR.

In this assessment, an important distinction is made in estimated pricing **between products sold as raw materials for the fertiliser industry and products sold directly as fertilisers**. Importantly, it is generally not possible to isolate prices for each specific end use when the same product can be used in various ways.

- **Raw materials for chemical processing**: This case mainly refers to mono-incineration ashes derived from P-rich input materials such as sewage sludge, meat and bone meal, and possibly manure fractions. Sewage sludge ashes are not useable as fertilisers directly due to the metals/metalloids contained, meaning that a lower price per tonne of P_2O_5 can be expected than for finished fertiliser products as various additional costs are incurred before reaching the end point in the manufacturing chain. For ashes derived from meat and bone meal and manure, possible post-processing could further improve the plant availability of the P contained. As such manufacturing steps involve large-scale industrial manufacturing, the specific chemical composition of the product sold is crucial, as relatively small variations in material quality (e.g. P content, Al/Fe content) can result in significant increases in operating costs for the consumers of these materials. The price setting for such materials is therefore largely **determined by the price setting of CFR phosphate rock** which is consumed as a source material by downstream manufacturers of complex P-fertilisers.
- **Fertilisers ready for sale**: This is the case of products such as **DAP, MAP or TSP**. In most cases, no further processing is required, meaning that higher prices per tonne of P_2O_5 can be expected compared to raw material prices; the expected sales price includes the manufacturing costs already incurred throughout the value chain, plus some profit margin of all the actors involved in the value chain. Physical qualities like particle size or hardness, or the content of impurities, can play a role in pricing. The chemical composition of such products is, generally speaking, homogenised and standardised. Here, the **price setting occurs relative to finished fertiliser products already on the market**, and the value of P is likely to be linked to the value of P in a multinutrient fertiliser such as DAP.
- **P-materials for secondary compounds**: This is **somewhere in between the previous two categories**. Some producers purchase products (e.g. DAP), grind them into smaller particles, and re-granulate a mix of various particles to obtain a product containing various nutrient grades. Contrary to bulk blends (which are just a physical mix of

individual fertilisers), these re-granulated products have a uniform nutrient content in each granule. Since the raw materials for this process need to be ground, physical characteristics play almost no role in price-setting, and **the nutrient content is far and away the main determinant of traded value**. A condition is, however, that the materials have characteristics that make them suitable for blending. **Buyers of such products are often able to obtain a relatively lower price per tonne of P_2O_5 compared to products sold as finished fertilisers, mainly by targeting material with poorer physical characteristics**. A relevant example for such a case could be powdered struvites. Hence, for STRUBIAS materials that will be used as raw materials for secondary compounders, the price for the P contained in the STRUBIAS materials is likely to be set relative to P-fertiliser products used by blending companies. After the application of a discount, the price for such materials could mirror the price of finished fertilisers on the market.

The value of STRUBIAS materials that will be brought on the market as P-fertilisers is likely to be determined to a significant extent by the P content contained in the material**.** The benchmark for the valuation of incineration ashes that will be used as intermediates in the P-fertiliser production chain is **phosphate rock** ($P_{intermediate}$ in Table 18), whereas end materials of STRUBIAS production processes (i.e. the CMC material) will be benchmarked relative to materials for blending or finished fertilisers, or **DAP** (P_{final} in Table 18). The current CFR market prices for DAP and phosphate rock, expressed as Euro per tonne P contained in the material, are given in for different EU Member States.

Table 17: Value of P present in DAP and phosphate rock from Morocco (32% P_2O_5, EUR per tonne P, CFR Morocco) assessed for different EU Member States for the year 2017 based on import prices (Source: Fertecon; exchange rate of EUR 1.2/USD applied by JRC)

	France	Germany	Italy	UK	Poland	Romania	Average
DAP	1213	1397	1213	1234	1198	1256	1252
phosphate rock	558	685	594	852	539	665	649

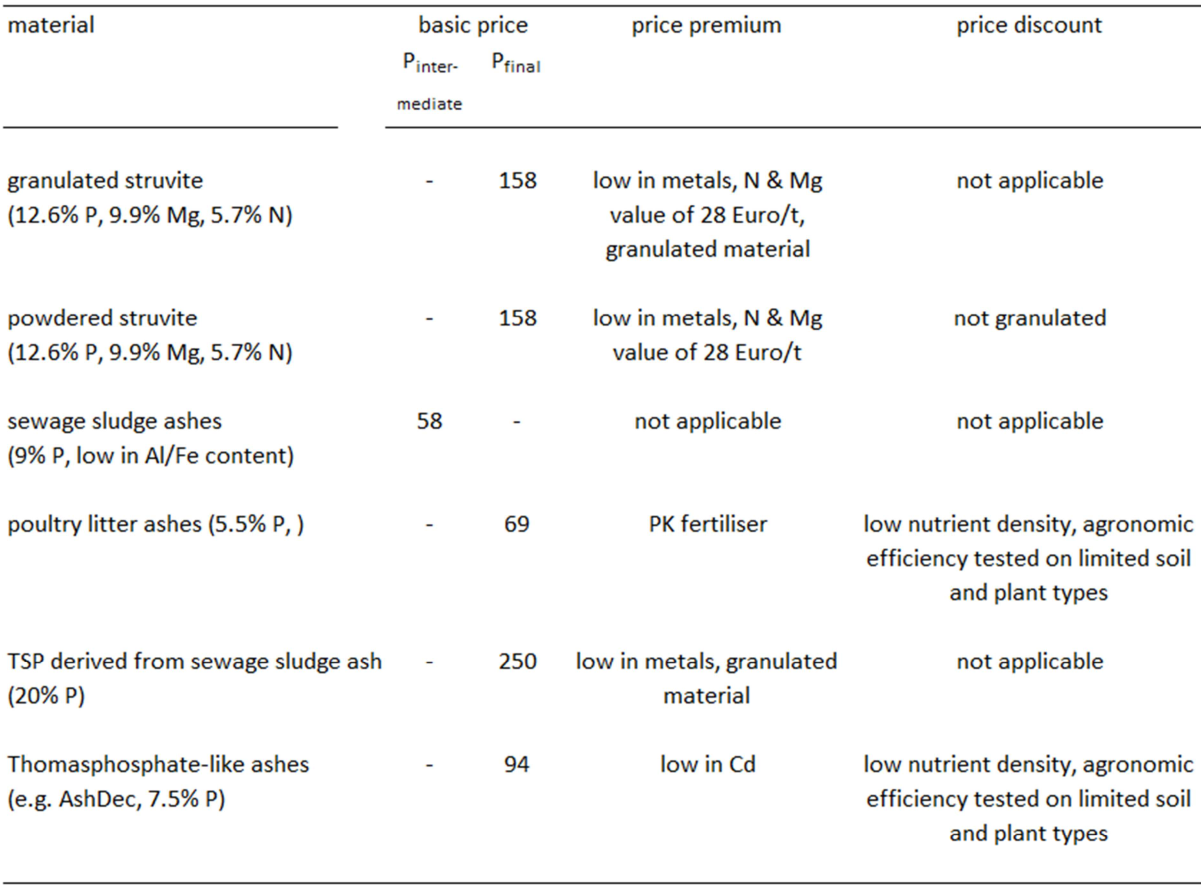
The price as estimated based on the P content is further expected to be modified by price premiums and price discounts.

- **Price premiums** could possibly apply to materials that are low in metals/metalloids, mainly precipitated phosphate salts & derivates. The use of precipitated phosphate salts & derivates in fertiliser blends could effectively reduce the Cd content in P-fertilisers, and as such avoid the need for decadmiation (an estimated cost of about EUR 27-68 per tonne P) or the reliance on possibly more expensive phosphate rocks, low in Cd content. Supplementary nutrients in STRUBIAS fertilisers are valued for fertilisers ready for sale and raw materials for secondary compounds used by compounders or blenders who would be able to fully monetise the nutrient values in the formulation. **Magnesium** is valued at EUR 130 per tonne Mg. An additional Mg value for STRUBIAS materials containing Mg (e.g. struvite) will be added. **Nitrogen** is valued at around EUR 320 per tonne across EU Member States. The value of N is based on the price of ammonia as a source of N. An additional N value for STRUBIAS materials containing N (e.g. struvite) will be added. For struvites, the combined monetary value for Mg and N will, however, be relatively lower than the monetary value for P (< 20%).

- **Price discounts** are expected to apply for STRUBIAS materials that i) have a reduced or unknown relative agronomic efficiency compared to mined rock phosphate and processed P-fertilisers, ii) are non-granulated, iii) have a lower nutrient and P-density that increases fertiliser distribution and application costs, iv) show a reduced suitability for their use in fertiliser blends, and v) are generally associated with a reduced degree of consumer confidence.

Examples of possible benchmark CFR prices based on their P content for selected STRUBIAS materials in conventional agriculture are given in Table 18.

Table 18: Benchmark CFR prices for the valuation of incineration ashes and STRUBIAS materials based on the P contained and other relevant price-setting factors (EUR per tonne material)



material	basic price		price premium	price discount
	$P_{inter-mediate}$	P_{final}		
granulated struvite (12.6% P, 9.9% Mg, 5.7% N)	-	158	low in metals, N & Mg value of 28 Euro/t, granulated material	not applicable
powdered struvite (12.6% P, 9.9% Mg, 5.7% N)	-	158	low in metals, N & Mg value of 28 Euro/t	not granulated
sewage sludge ashes (9% P, low in Al/Fe content)	58	-	not applicable	not applicable
poultry litter ashes (5.5% P,)	-	69	PK fertiliser	low nutrient density, agronomic efficiency tested on limited soil and plant types
TSP derived from sewage sludge ash (20% P)	-	250	low in metals, granulated material	not applicable
Thomasphosphate-like ashes (e.g. AshDec, 7.5% P)	-	94	low in Cd	low nutrient density, agronomic efficiency tested on limited soil and plant types

For some STRUBIAS materials, the target market for pyrolysis & gasification materials may not be the conventional agricultural market. Specific materials of all STRUBIAS material groups could possibly make a market entry in niche markets where fertilising products are traded at higher prices (e.g. organic farming, horticulture, and growing media). Also, the marketing of STRUBIAS materials as part of the circular economy could drive sales prices upwards in some markets. In organic farming, the absence of competition from mined rock phosphate and processed P-fertilisers could positively impact upon the sales prices of the STRUBIAS materials. Hence, in niche markets, **the specificity of the source materials and the possible environmental impacts of STRUBIAS production processes on natural resources can be expected to impact upon the sales prices of the STRUBIAS materials**.

It is expected that the future evolution in sales prices will closely reflect the expected price evolution for phosphate rock (intermediate raw materials) and P value in finished fertilisers (DAP) as given in Section 9.1.2. For these materials, a negligible increase for the 2016-2025 period is expected. The future **prices for STRUBIAS materials are therefore expected to remain stable**. Nonetheless, consumer confidence in STRUBIAS materials and the field verification of the presumed agronomic efficiency in the relevant agricultural sectors could further impact the future prices of STRUBIAS materials.

Based on this assessment, it is clear that **STRUBIAS materials can vary largely in pricing, depending on the STRUBIAS material group, the characteristics of the output material, and the target markets.**

9.2 Compliance costs for STRUBIAS materials

The compliance cost of producing fertilising products containing precipitated phosphate salts & derivates, thermal oxidation materials & derivates, and pyrolysis & gasification materials is dependent on the identity of the contaminant proposed in the STRUBIAS recovery rules. The cost data are given for testing as performed according to the *International Standards* indicated in Section 5.7.3.

The assessment on the compliance cost indicates the costs associated with the testing of the parameters included in the recovery rules for the corresponding CMC. Some of the parameters included in the compliance scheme should be measured anyway because they are directly included as part of the testing requirements at PFC level (e.g. organic C), or are measured concomitantly with PFC-level parameters (e.g. metals like Cr, Th and V, which are measured in the same sample run as other metals and metalloids such as As, Cd and Pb). The cost assessment will take into consideration the testing requirements that are specific to the CMC (as indicated in red). **This cost corresponds to the supplementary testing costs relative to equivalent fertilising products produced from primary raw materials for which no CMC testing requirements other than REACH registration have been proposed (i.e. CMC 1 in the EU Fertilising Products Regulation ((EU) 2019/1009)).**

Table 19: Price estimates for the different parameters included in the proposals for the STRUBIAS recovery rules for the three candidate material groups

STRUBIAS material group		Price estimate in EUR mean (range)	Comment
Precipitated phosphate salts & derivates			
	Nutrients, including total P, total Al, total Fe	76 (38-112)	PFC testing requirement for testing on nutrients, metals and metalloids – no additional cost
	Organic carbon	36 (17-80)	PFC testing requirement
	Macroscopic impurities	69 (50-90)	
	Salmonella spp., *Escherichia coli*, *Enterococcaceae*	74 (36-170)	
	PAHs (only required for some input materials)	65 (45-100)	
	Clostridium perfringens	35 (20-50)	
	Ascaris sp. eggs	60	
	Dry matter content	10 (5-30)	
Total cost		244	
Thermal oxidation materials & derivates			
	Organic carbon	36 (17-80)	PFC testing requirement
	PAHs	65 (45-100)	
	PCDD/F	404 (200-660)	
	Cr, Tl, V (individual measurement)	30 (25-37), per element	Usually measured together with other metals and metalloids
	Cr, Tl, V (at times also including other elements not included in CMC testing scheme, e.g Ca, K, Mg, As, Cd, Cu, Hg, Pb, Ni, Zn, Cu)	200 (85-525)	PFC testing requirement for testing on metals and metalloids – no additional cost
	Cl-	43 (20-69)	
	pH	9 (2 – 30)	
Total cost		521	

Pyrolysis & gasification materials			
	Organic carbon	36 (17-80)	PFC testing requirement
	H	31 (20-45)	
	PAHs	65 (45-100)	
	PCDD/F	404 (200-660)	
	dl-PCB	81 (60-120)	
	Cl-	43 (20-69)	
	pH	9 (2 – 30)	
Total cost		633	
Labelling requirements			
	Neutralising value	30 (20-40)	

NB: The total compliance cost is the sum of the different analyses that are exclusively required for testing at CMC level (parameters depicted in black are included in the measurements to test compliance with parameters at PFC level; parameters depicted in red are exclusively required to test compliance with the proposed STRUBIAS recovery rules).

Additionally, EU fertilising products, regardless of whether these are produced from primary or secondary raw materials, should comply with the REACH Regulation ((EC) No 1907/2006). The REACH Regulation addresses the manufacture, use and placing on the market of chemical substances and mixture, and their potential impacts on both human health and the environment. For full guidelines on the links of recovered substances to the REACH Regulation, reference is made to the documents 'ECHA – Guidance on waste and recovered substances' (ECHA, 2010), 'Guidance for identification and naming of substances under REACH and CLP' (ECHA, 2016) and 'Guidance on the interpretation of key provisions of Directive 2008/98/EC on waste' (European Commission, 2012). For **STRUBIAS materials that will be placed on the market**, Article 2(7)(d) of REACH could possibly provide an **exemption for the registration of STRUBIAS substances that are already REACH-registered**. Once the type (substance on its own or in a mixture) and composition of the recovered STRUBIAS material have been established, identified and documented, the recovery operator is therefore in a position to examine whether the **exemption criteria** under Article 2(7)(d) of REACH are fulfilled.

9.3 Possible economic benefits and drawbacks of producing fertilising products containing STRUBIAS materials

9.3.1 Effects on costs for producers of eligible input materials for STRUBIAS pathways

The incorporation of STRUBIAS materials in the EU Fertilising Products Regulation ((EU) 2019/1009) might **prevent additional costs** for producers of eligible input materials for STRUBIAS pathways, through reducing costs for the management and/or disposal of the specific materials, and reduced compliance costs (sensu Wijnands and Linders, 2013)**.** Benefits might, for instance, ensue for operators of incineration facilities that sell sewage sludge ashes to a P-recovery operator instead of sending them to a landfill, or for farming cooperatives that locally process their excess manure fractions to nutrient-concentrated STRUBIAS materials instead of exporting them to other EU Member States. As observed in Chapter 14, the STRUBIAS pathways might become an additional chain of a material recycling cascade. Therefore, they might contribute to the economic exploitation of entire recycling cascades by further making use of currently unused residues of such cascades.

9.3.2 Effects on the competitiveness of the European agricultural sector

The average expenditures of fertilisers at EU level range between 1% and 12% of total farm costs (Wijnands and Linders, 2013). The costs are relatively high for crop-producing farms: for specialised field crops almost 12%. The addition of STRUBIAS materials as possible ingredients for EU fertiliser products will **stimulate innovation to possibly develop new fertilising products**, for example with different nutrient-release dynamics to conventional water-soluble P-fertilisers or through combining STRUBIAS materials with other CMCs in a single product (e.g. pyrolysis materials as additives to compost). Those new products can be **placed on the market as EU fertilising products without administrative burdens** due to lengthy procedures. Hence, creating a level playing field in the EU will promote enhanced market integration and **improved competition** between fertilising manufacturers and fertiliser blending companies with **possible effects on farmers' purchase prices for fertilising materials**. Finally, producing P-fertilisers from secondary raw materials produced locally in Europe will **reduce the susceptibility of the European agronomic sector to P-fertiliser price volatility** due to possible geopolitical tensions and the depletion of easily available high-quality phosphate rock.

9.3.3 Effects on the transparency and competitiveness of the European fertiliser sector

At present, some actors in the internal fertilising product market are sceptical about the mutual recognition Regulation (Regulation (EC) No 764/2008), as it has created disruption on the market resulting in unfair competition because viewpoints on safety regulation between National Competent Authorities differ. This is especially an issue of concern for new fertilising products that are derived from secondary raw materials, such as STRUBIAS. If already existent, the different national regulations related to these fertilising materials create market fragmentation and generate substantial administrative burdens for companies and national competent authorities. **Harmonisation of the legislation concerning all fertilising materials at EU level will increase the transparency within the EU and thus decrease transaction costs** (Wijnands and Linders, 2013). Actors producing, trading or using these fertilising materials can rely on the same framework, and make use of the principle of free movement of goods in the EU. Information on the composition, tolerance levels and maximum level of contamination is available in the same legislation, **ensuring transparency to downstream users**, including retailers, fertilising blending companies, and farmers.

The Regulation will offer the possibility to render the fertilisers sector **less dependent on imports of critical raw materials**, such as phosphate rock. The Regulation may thus help to reduce the sector's vulnerability to imports of a finite raw material for which price settings may possibly depend on long-term material availability and the geopolitical situation in supplying countries.

STRUBIAS production pathways mainly depart from biogenic organic materials that are currently spread on land as an input material, especially manure and to a lesser extent sewage sludge. Some of the STRUBIAS production processes, particularly the thermal oxidation pathway, result in the destruction of other valuable agronomic resources, such as nitrogen or organic matter. In such cases, these elements will have to **be replaced by alternative materials, such as N fertilisers, or compost**. If those specific pathways develop substantially, increased demands for those alternative materials can equally be expected.

9.3.4 Restructuring of fertiliser production and distribution systems

STRUBIAS materials are often part of a material transformation cascade that generates a set of co-products (e.g. clean water, renewable energy, STRUBIAS materials). Different operators are active in this cascade, and STRUBIAS material manufacturers will have to collect eligible input materials for their production process at facilities of upstream operators in the material transformation chain. Hence, **actors belonging to different sectors are active in the material transformation cascade (e.g. agriculture, wastewater treatment, waste management, food processing industry) and will have to establish agreements in order to synchronise material streams, considering quantitative and qualitative aspects, amongst the different sectors involved**. Efficient business models are needed to turn the various benefits of P-recovery into a market success. Accordingly, new multi-stakeholder business models that create synergies between waste management actors and the fertilising industry should emerge to harness economic opportunities in value creation from the recovery and reuse of resources that would otherwise be irretrievably lost (and cost money to dispose of).

Manufacturers that place on the market EU fertilising products containing STRUBIAS materials shall perform the **quality assessment procedures** as described in Section 5.6. Companies that have an internal production or product quality control system in place will have lower costs than companies that do not yet have such a system. In the first case, a new regulation might incur some cost for amendments (Wijnands and Linders, 2013). In the second case, a control system has to be put in place and that will involve investment costs. These procedures and inspections will create a certain administrative burden, and adverse impacts will be relatively higher for small and medium enterprises than for large companies.

9.3.5 Technical adaptations for downstream users of STRUBIAS materials

Fertiliser **blending companies** will be given the opportunity to manufacture innovative fertilising products with a tailored chemical and physical composition that include STRUBIAS materials (see Section 9.3.2). This may, however, **require additional testing and possibly the implementation of novel techniques to ensure the physical and chemical compatibility of the different input materials**. The extent to which blending companies will undertake process modifications will be dependent on the market potential of these investments in blending procedures and the market acceptance and confidence in STRUBIAS materials. It should, however, be noted that further efforts related to blending are not required as STRUBIAS materials can be placed on the market without industrial processing.

It is expected that STRUBIAS materials will be incorporated into EU fertilising products that have the same physical form as those currently available on the market. Therefore, **no additional costs for farmers owing to equipment adaptation requirements for fertiliser application on land are expected**.

9.3.6 Externalities and soil degradation

Phosphorus and nitrogen fertilisers applied to the soil in quantities that exceed the uptake capacity of plants can result in P accretion in soils, N and P leaching and run-off, and may result in environmental impacts. Such externalities constitute the increasing costs to be borne by society (Vollaro et al., 2016). A persistent surplus of N and P in the soil leads to environmental impacts on water (nitrate pollution, eutrophication, acidification, etc.), air (air quality and acid rain), climate (GHG emissions contributing to global warming), soil (acidification and accumulation of heavy metals) and biodiversity (loss of species) (Sarteel et al., 2016). A consequence is that nutrient surpluses impact upon externalities: **human health costs**, **additional costs for drinking water treatment**, etc. All impacts depend on what practices are implemented and on local conditions in terms of climate and soil but also on the possible magnitude of nutrient overload in environmental compartments. For the EU-27 in 2008, the total social cost of eutrophication was estimated to be between EUR 75 million and EUR 485 billion per year (Van Grinsven et al., 2013). Although the willingness-to-pay approach used in that study recognises considerable uncertainties and conceptual challenges in such a monetised valuation of non-commensurable issues, the cost estimate provides a general idea of the externalities and the need to address those from an economic perspective. As outlined in Section 8.9.2, STRUBIAS production processes can, on some occasions, provide solutions to improve nutrient efficiency. **STRUBIAS processes can thus reduce externalities** by promoting processing techniques to reduce the dissipation of nutrients, and providing options for the fine-tuning of fertilisation management by providing a more balanced nutrient ratio in fertilising materials.

Land degradation costs an estimated USD 40 billion annually worldwide (The Ellen MacArthur Foundation, 2014). Soil degradation might contribute to food shortages, higher commodity prices, desertification and ecosystem destruction in the EU (Jones et al., 2010).

Society has a duty to ensure that the soil resources within their territories are managed appropriately and sustainably. There is plentiful information demonstrating the benefits of maintaining and increasing soil organic C stocks to mitigate soil degradation across a range of different soils, agro-ecosystems and climatic zones (Chabbi et al., 2017). On average, soils in Europe are most likely to be accumulating carbon. Soils under grassland and forests are a carbon sink (estimated up to 80 million tonnes of carbon per year) whereas soils under arable land are a smaller carbon source (estimated to be from 10 million tonnes to 40 million tonnes of carbon per year) (European Environment Agency, 2012). STRUBIAS production processes might impact soil organic carbon through different mechanisms. The first mechanism applies to all STRUBIAS processes that help to reduce excess N in the soil, as the latter may cause an increase in mineralisation of organic C which, in turn, leads to an increased loss of organic C from soils. The second group of mechanisms is specific to the STRUBIAS material groups that might increase or decrease the return of soil organic matter to soils:

- The effect for precipitated phosphate salts & derivates is **neutral** as this technology involves only a separation of the predominantly inorganic fraction (the precipitate) from an organic-rich stream that can be further applied on land as a soil improver or N fertilising material.
- The thermal oxidation process effectively destroys the organic matter present in the feedstock. The net effect for thermal oxidation materials & derivates is thus **dependent on the counterfactual use of this material**, with **negative** effects on soil organic C expected for materials that are otherwise returned to land (e.g. poultry litter) and **neutral** effects for materials that are otherwise (co-)incinerated (e.g. sewage sludge in some EU regions).
- Pyrolysis & gasification materials with a low $H:C_{org}$ ratio are often C-rich materials with a claimed long turnover time in soils. Lehmann et al. (2015) indicated that more than 90% of the initial C present in the biochar will remain in the soil after 100 years. Given that on average 50% of the C from the feedstock material is lost during the pyrolysis process (Enders et al., 2012), this would imply a potential long-term sequestration of 45% of the C present in the feedstock material. That value is higher than the typical long-term C sequestration from raw, digested or composted feedstocks (on average 11%; Bruun et al., 2006). The **effect of pyrolysis & gasification materials on the soil organic C balance is thus evaluated as positive** (Lal, 2009; Beesley et al., 2011; Wu et al., 2017).

GENERAL CONCLUSIONS

10 Concluding assessment for STRUBIAS materials

As outlined in Section 3.1, the STRUBIAS material groups are assessed against three different criteria. Based on the techno-scientific data collection and analyses, the following conclusions could be derived:

I. *The use of the materials will not lead to overall adverse environmental or human health impacts*.

The potential negative health impacts identified relate to risks in the recycling of contaminants contained in the eligible STRUBIAS input materials and the *de novo* production of specific contaminants. Therefore, the eligible input material list was restricted for all three STRUBIAS material groups, excluding for instance mixed municipal waste as starting material for STRUBIAS production processes. Moreover, specific requirements were proposed on, for example, conditions for thermal oxidation. Still, particular contaminants of concern were identified that could be transferred to the STRUBIAS materials. This refers specifically to biological pathogens and some organo-chemical compounds (pharmaceutical compounds, polycyclic aromatic hydrocarbons, etc.) for precipitated phosphate salts & derivates, metals for thermal oxidation materials & derivates, and volatile organic carbon for pyrolysis & gasification materials. Therefore, direct and indirect product quality requirements to restrict those contaminants in the STRUBIAS materials were proposed. Altogether, the technical requirements for the different Component Material Categories should ensure that the use of the STRUBIAS materials does not lead to overall adverse environmental or human health impacts, thus ensuring food safety. This was confirmed using life cycle analyses that indicated that the STRUBIAS production options assessed provide savings for one or more of the impact categories assessed (i.e. global warming potential, eutrophication potential, or human health cancer toxicity) relative to mined phosphate rock and processed P-fertilisers.

II. *The material shall provide plants with nutrients or improve their nutrition efficiency, either on its own or mixed with another material [following the definition of fertilising products in the EU Fertilising Products Regulation ((EU) 2019/1009)].*

A detailed analysis of this criterion using meta-analyses approaches indicated that precipitated phosphate salts & derivates and thermal oxidation materials & derivates that comply with the proposed STRUBIAS recovery rules generally provide plants with nutrients, especially P, with a similar agronomic efficiency to mined phosphate rock and processed P-fertilisers. For pyrolysis & gasification materials, it was indicated that plant responses for P-rich pyrolysis & gasification materials can vary widely depending on the feedstock and production conditions of the pyrolysis & gasification materials, with some materials showing similar efficiencies to mined phosphate rock and processed P-fertilisers. It is observed that not all these organic P-fertilisers are effective within every agronomic setting. However, responsible use practices for those fertilisers can also contribute to reducing the reliance on primary raw materials under specific situations, in line with the principles of precision and targeted agriculture to increase nutrient use efficiency in the EU. STRUBIAS materials that show a low phosphorus content can serve other fertilising functions (e.g. N, K and/or micronutrient plant nutrition, soil improver, liming material, growing media, plant biostimulant), thus contributing to improving plant nutrient uptake efficiency in the short or long term in specific situations. It is concluded that all three STRUBIAS material groups provide plants with nutrients or improve their nutrition efficiency, although the latter may only hold true under specific European settings for some pyrolysis & gasification materials.

III. *Trade on the internal market can be expected for such a fertilising material, based on the current market and the future market and trade forecasts.*

The internal market for STRUBIAS materials is currently emerging, for which at present no EU-wide legal framework exists. Therefore, assessing the expected trade of STRUBIAS materials on the internal market is challenging. Further market developments, technological progress and economic benefits due to scale effects might develop. The materialisation of STRUBIAS production pathways not only depends on the feedstock availability and production cost, but is also subject to other decisions that impact on the need to process eligible input materials for STRUBIAS materials in a different manner relative to the current business-as-usual practices. With the intention to promote a more circular and resource-efficient economy, policy targets, financial incentives or financial disincentives for the handling of biogenic materials will thus impact upon the STRUBIAS market outlook. Market demand and trade are expected for all three STRUBIAS material groups in different segments of the EU agricultural sector. The most important share of the STRUBIAS materials will be used as fertiliser that can be used to provide nutrient, mostly phosphorus, inputs to European agriculture. As a best estimate, the opening of the P-fertiliser market to STRUBIAS materials is expected to result in a substitution effect of mined phosphate rock and processed P-fertilisers by fertilising products containing precipitated phosphate salts & derivates and thermal oxidation materials & derivates of 17% to 31%. Market demand for STRUBIAS materials is also expected for those used in liming materials, soil improvers or growing media.

Overall, it is concluded that precipitated phosphate salts & derivates, thermal oxidation materials & derivates, and pyrolysis & gasification materials meet all three criteria against which they were assessed. STRUBIAS materials offer an avenue to enhanced food security and sustainable fertilisation, while conserving the environment and its natural resources in Europe and elsewhere. A stable legal framework for all three STRUBIAS material groups is therefore desirable to promote the trade and use of these materials in the EU agricultural sector and to provide a material quality benchmark for producers and consumers of P-fertilising materials containing STRUBIAS materials.

APPENDIX

11 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 wastewater treatment configuration
CMC	Component Material Category in the EU Fertilising Products Regulation ((EU) 2019/1009)[31]. An EU 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 EU Fertilising Products Regulation ((EU) 2019/1009).
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 EU 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 wastewater 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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[31] 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 contain 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 12 for chemical conversion factor to phosphorus pentoxide)
PAHs	Polycyclic aromatic hydrocarbons (also polyaromatic hydrocarbons or polynuclear aromatic hydrocarbons)
PAP	Processed animal protein
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 EU fertilising products shall belong in the EU Fertilising Products Regulation ((EU) 2019/1009)[32] 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 superphosphate - a water-soluble mineral phosphorus fertiliser that contains calcium dihydrogen phosphate and gypsum

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

STRUBIAS materials	STRUvite, BIochar and ASh-based materials. The acronym STRUBIAS, was chosen as working title and does not necessarily reflect the final scope of any possible proposals for CMC categories
STRUBIAS subgroup	A technical working group that constitutes a subgroup of the Commission expert group on Fertilisers. The STRUBIAS subgroup 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 superphosphate - 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.

12 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

13 Potential input materials for fertilising products containing STRUBIAS materials

13.1 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 25). 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 25). Animal production and non-food production are associated with lower losses (Figure 25) (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 wastewater 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 wastewater treatment plants. Untreated and uncollected wastewaters 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}).

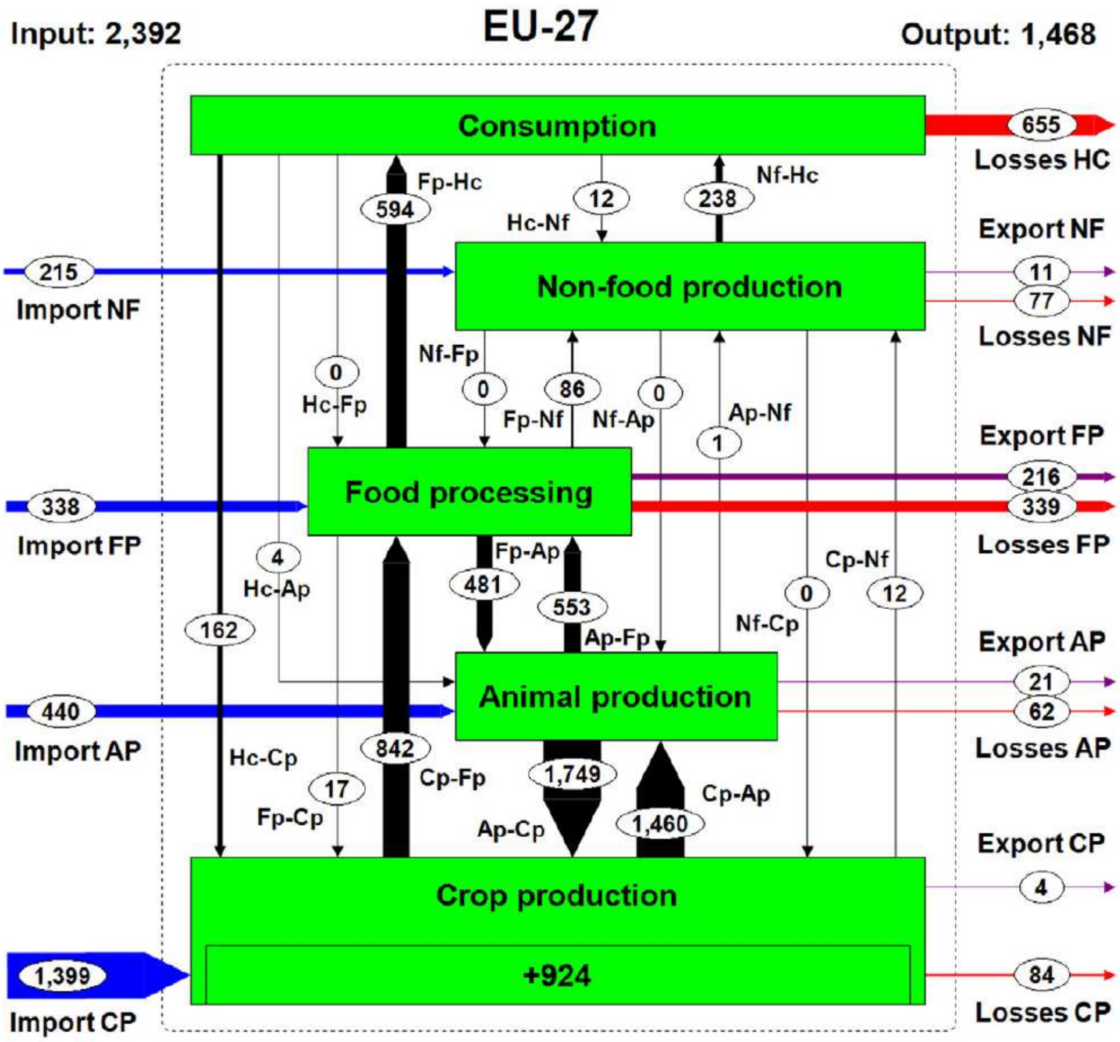


Figure 25. Phosphorus (P) use for the EU-27 in 2005 [kt P yr^{-1}] aggregated at the food and non-food production–consumption–waste chain; showing the imports (blue), exports (purple), losses (red) and internal upward/downward flows (black) for crop production (CP), animal production (AP), food processing (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 - © Elsevier Ltd., 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.

13.2 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).

13.3 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 with 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 quantities 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 with 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.

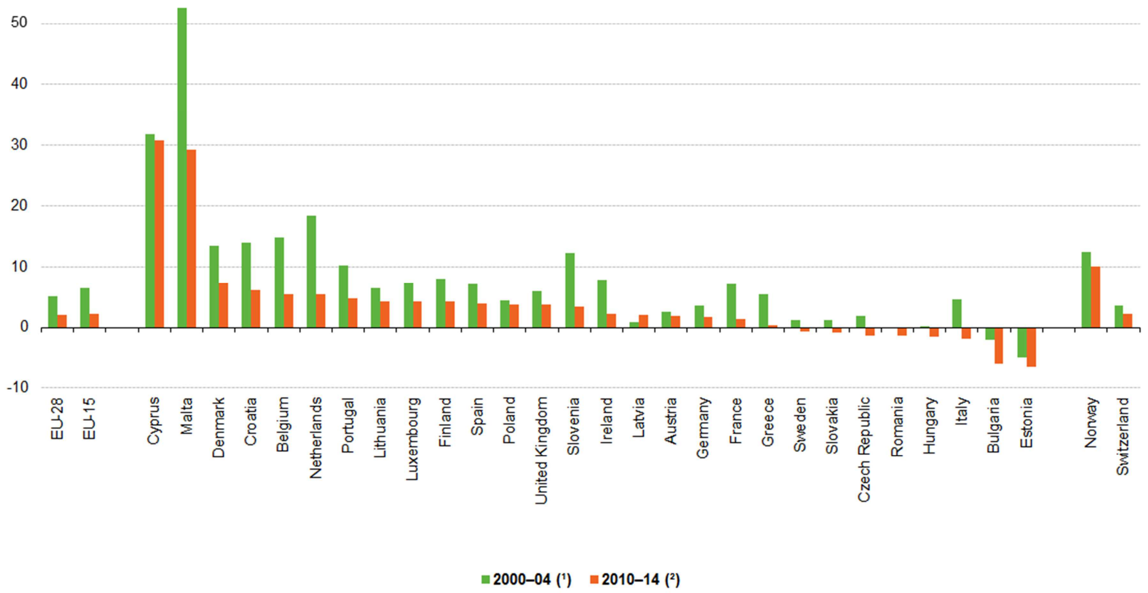
The outputs of the gross phosphorus balance are:

- 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 26). 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 26: 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, aluminium and iron sulphate, aluminium 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.

13.4 Animal by-products other than manure

13.4.1 Food waste from households, retail and food services

Food waste in the household sector involves **waste associated with 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 13.4.4).

13.4.2 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, 2006a). 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 undigested 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 13.4.3 and 13.4.4).

13.4.3 Materials from the meat industries

The cattle, pig and poultry industries are the largest source of animal food industry waste (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 27. 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 some of their respective main treatment routes (adopted from Moller, 2015)

Category 1	Category 2	Category 3
By-products of animals suspected of being infected with transmissible spongiform encephalopathy (TSE) and specified risk materials, including bovine brain and spinal cord, fallen ruminants	By-products of animals presenting a risk other than TSE, non-ruminants that have died in ways other than being slaughtered, animals killed to eradicate an epizootic disease, and manure	By-products arising from the production of goods intended for human consumption using slaughtered animals. These may include leftovers from cantinas and food processing industry
Material shall 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 P from the nutrient cycle	Material can be a. Treated as category 1 material b. Used for the manufacturing of organic fertilisers following processing by pressure sterlisation and permanent marking of the resulting material c. Composted or transformed into biogas following processing by pressure sterilisation d. In case of manure, applied on land in unprocessed form.	Material shall be a. Treated as category 1 or category 2 material b. Used for manufacturing of animal feed after pasteurisation c. Used for the manufacturing of organic fertilisers following processing by pressure sterlisation

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 **wastewaters** are produced at the slaughterhouse. This category contains dung and urine from animal holding areas, washings from distribution vehicles, and wastewater generated during the process of meat and bone meal production (European Commission, 2005). These wastewaters are either treated on-site or discharged to municipal wastewater treatment (European Commission, 2005).

13.4.4 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 by-product 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 27.

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 27). category 2 material is mainly used for the production of biodiesel (fat fraction), and fertilisers (protein fraction) (Dobbelare, 2017) (Figure 27). About 180 kt of category 2 protein material is used for the production of (organic) fertilisers in the year 2016 (Dobbelare, 2017). The category 3 animal by-products are typically processed to produced fat and Processes Animal Proteins (PAP), that can be used for pet food, animal feed including and fish food, 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 food, human food (gelatin) and fur feed (EFPRA, 2017) (Figure 27). 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.

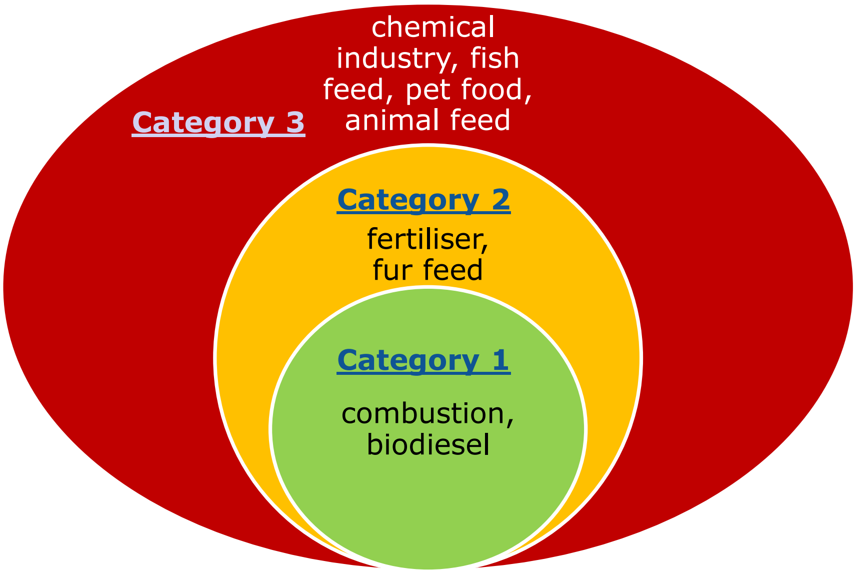


Figure 27: Potential application of processed animal by-products of category 1, 2 and 3 (adopted from EFPRA (2017))

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., 2005b; 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., 2005b; Wopenka and Pasteris, 2005; Czaja and Hermann, 2011). The combustion induces a wide range of structural modifications, such as crystallisation 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**.

13.5 Effluents and residues from municipal wastewater treatment plants

Wastewater treatment plants process grey/black water from households and industrial wastewater produced. The dominant share of P from **industrial wastewaters** 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.

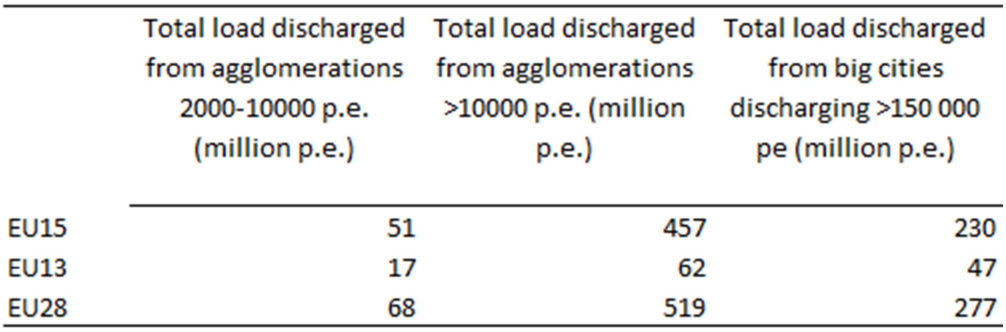
Municipal sewage is used water mainly coming from domestic activities and sometimes combined with used water from industrial activities and/or with surface run-off. The Urban Wastewater Treatment Directive (91/271/EEC) defines an **agglomeration** as an area where the population and/or economic activities are sufficiently concentrated for urban wastewater to be collected and conducted to an urban wastewater 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 Wastewater Treatment Directive** (91/271/EEC) defines in which conditions the wastewater 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 wastewater 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, 2013a). 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, 2013a).

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

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



	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

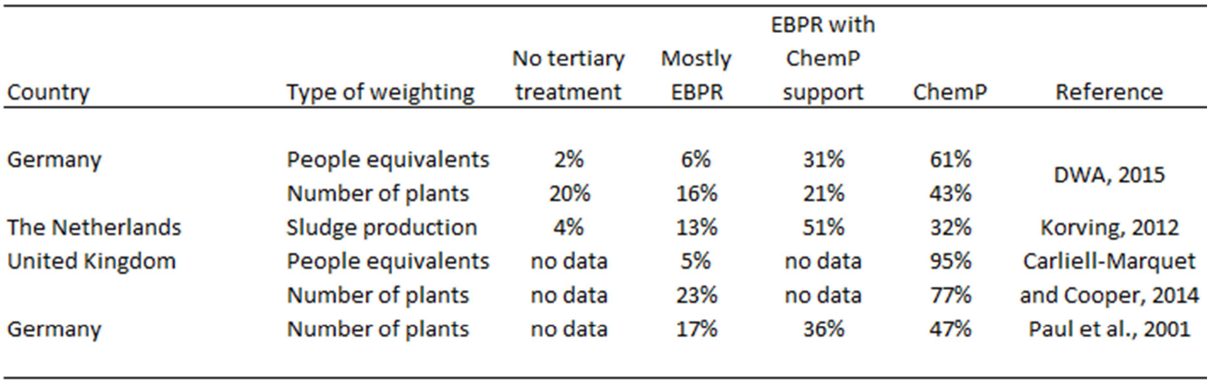
Different configurations exist for P removal in wastewater treatment plants, with the most popular techniques being **chemical phosphorus removal (Chem-P)** using Fe or Al salts and **enhanced biological phosphorus removal (EBPR)** .

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 (aluminium sulphate), sodium aluminate, polyaluminiumchloride, ferric chloride, ferric sulphate, ferrous sulphate, 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 sulphide 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 wastewater 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 wastewater is a requirement under Article 5 of the Urban Wastewater Treatment Directive (91/271/EEC), **this directive does not require nor incite to P recovery or reuse, nor to recovery or reuse of other nutrients, water, etc.**

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 28). 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**.

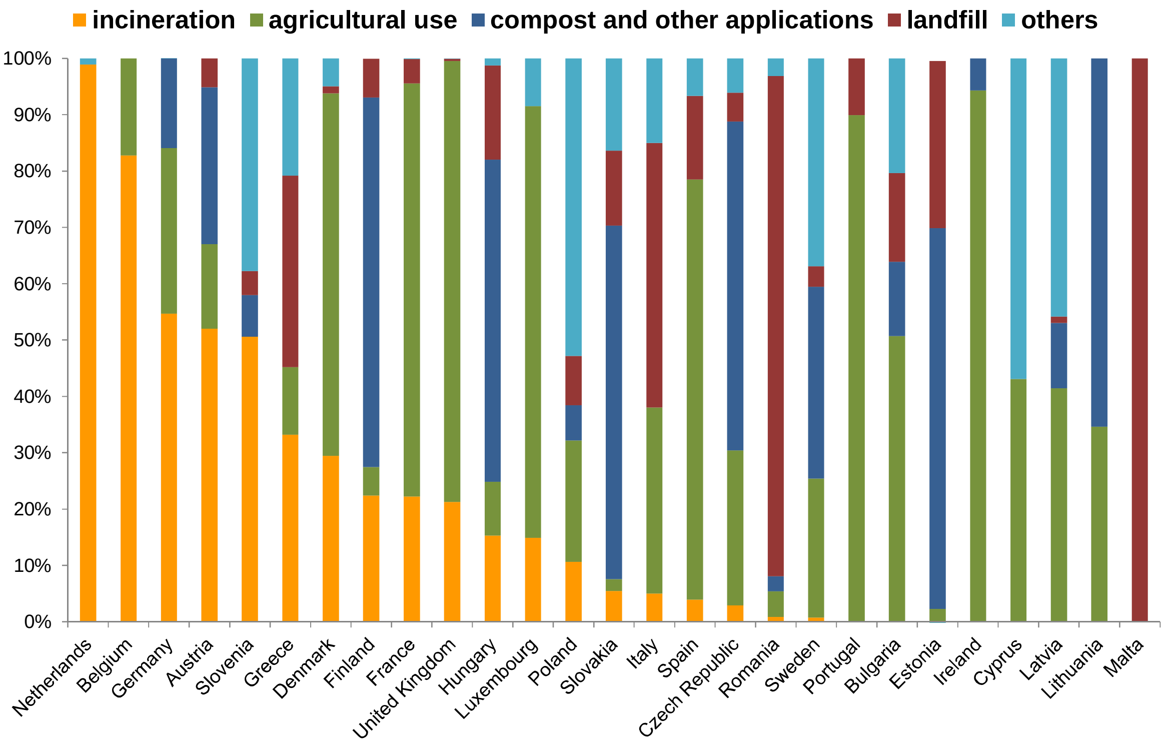


Figure 28: Routes for sewage sludge disposal in the year 2012 in EU-27 (year 2010 data for Italy) (data available from Eurostat).

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 28). 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 28). 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 28).

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 vaporise 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., 2005b; 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 wastewaters 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 wastewater treatment plants. Untreated and uncollected wastewaters 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 wastewater treatments plants, as well from the incinerated sewage sludges (see section 14).

13.6 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.

13.6.1 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 greyish-colored substance that causes the after-cooking grey discoloration (Rossell, 2001). This is a very significant P-source in the wastewater from potato processing installations. The wastewater have a typical $PO_4^{3—}$P concentration of about 200 mg L^{-1} (European Commission, 2017c). **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, 2017c). 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).

13.6.2 Wastewaters 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, by-products 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 wastewaters 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 wastewaters.

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

- 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 wastewater varies between 20 and 200 mg L^{-1} (European commission, 2006c). 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 wastewater: 60 – 65 mg P L^{-1}; P-recovery efficiency of about 75%).

In the **brewery**, wastewater from the anaerobic reactor is driven to a reservoir where it is mixed with aerobic sludge (from the second sedimentation) and with untreated neutralised wastewater (taken before anaerobic digestion). The $PO_4^{3—}P$ concentration in the wastewater 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.**

13.7 Chemical industry wastewaters

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 wastewaters**. 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).

13.8 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 burnt products, such as lime and/or dolomitic lime, 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, aluminium, 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.

Slags that are the result of **a thermal oxidation at high temperatures** could be considered **as thermal oxidation materials & derivates in the STRUBIAS project** (i.e. steel slags including basic oxygen furnace slag and electric arc furnace slag, but not ‘ladle furnace slags’ or secondary metallurgical slags). 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.**

13.9 Forest-based industry residues and green waste

13.9.1 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 actual total production of **forestry residues in the EU was 67.6 million tonnes** dry matter 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 **33% removal of forestry residues may be sustainable** if combined with good management practices. Assuming these values, a total of about 22 million tonnes dry matter 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, resulting in values of 900 – 1350 kt N and 54-81 kt P yr^{-1} (assuming an estimated moisture content of 25-50%). 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).

13.9.2 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, 2015b).

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, 2015b).

13.9.3 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 wastewater treatment plants to feed the biomass**.

13.9.4 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.**

13.10 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).

13.11 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 13.2 - 13.9.**

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.

13.12 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 wastewaters, food processing residues, residues from gelatin production process;
- to **remove nutrients for improved functioning of biological wastewater 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.

14 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.

14.1 Precipitated phosphate salts & derivates

14.1.1 Wastewater treatment plants

In **conventional wastewater treatment** plants, **P** is mainly **eliminated** by **chemical precipitation with metal salts (ChemP)** or **enhanced biological phosphorus removal (EBPR)** or by 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 aluminium (Al(III)), ferric (Fe(III)), ferrous (Fe(II)), polyaluminiumchloride, and calcium (Ca(II)) ions. The direct use of P-rich sludge as a fertiliser is associated with 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).

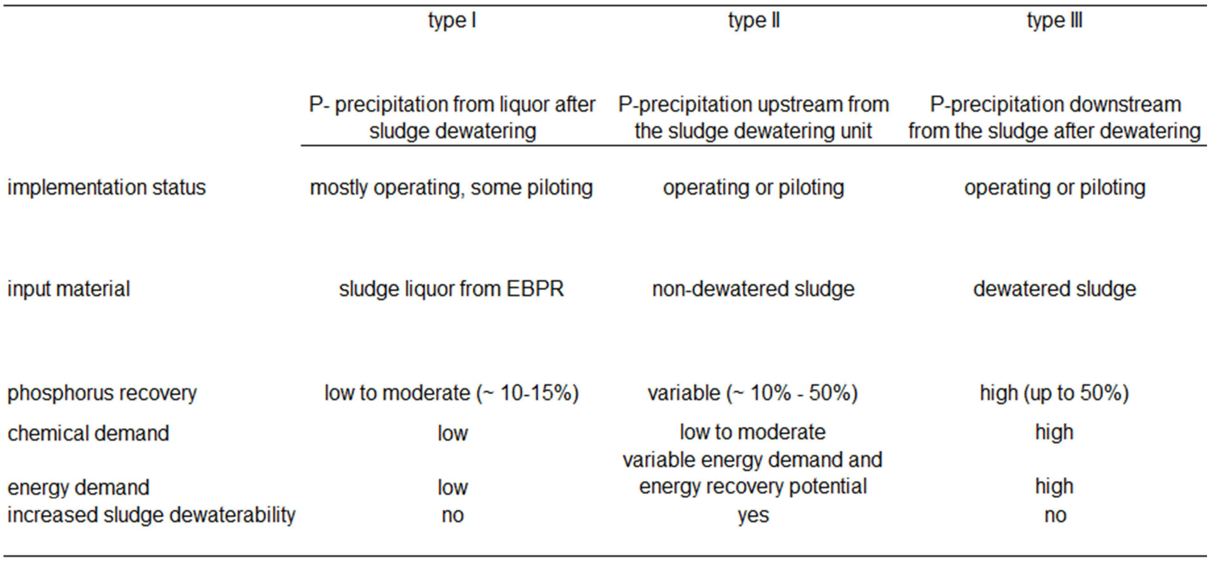
Phosphate salts can be recovered from **sludge liquor** and from **digested sludge** when the PO_4^{3-} precipitates together with Mg^{2+} or Ca^{2+}, possibly also trapping NH_4^+ and/or K^+ in the molecular structure. Struvite, the most commonly recovered phosphate salt, forms from equimolar quantities 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 wastewater 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.

Four types of recovered phosphate forming processes are considered: (I) from the sludge liquor after the sludge dewatering unit, (II) upstream from the sludge dewatering unit, (III) downstream from the digested sludge. 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 options for P recovery. The implementation of a P-recovery system **before the anaerobic digester** (type II) 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 wastewater 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 microcrystals 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 microorganisms 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 wastewater treatment plants.** These processes can use sludges produced in biological (EBPR) and chemical (precipitation with metal salts) wastewater 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 hydrothermal carbonisation treatment on downstream EBPR or ChemP sludges in order to break up the cellular structures of the sludge followed by an acid leaching to transfer phosphorous from the solid into the liquid phase. The solid phase, a P-depleted substance, is then separated from the P-rich filtrate. Finally the phosphorous in the filtrate is precipitated using Ca ions. The process is at present in a pilot stage.
- The P-RoC process is based on the precipitation reaction by means after the addition of calcium silicate hydrates. The latter acts as a reagent due to the release of calcium- and hydroxide-ions. Additionally, the mineral material provides a surface at which phosphate minerals can be generated, like highly disordered hydroxyapatite, struvite and brushite. P-RoC-technology precipitates dissolved phosphate in sludge liquor – with or without previous sludge disintegration.

There are specific processes (PASCH®, P-bac (INOCRE)®) that produce **struvite from ashes as input material**. These products will be described in section 14.2 (thermal oxidation materials & derivates).

14.1.2 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 wastewater (e.g. potato industry, dairy industry, type I processes) and anaerobically digested bio-waste 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).

14.1.3 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 wastewater 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 wastewater 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 with 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.
- Wastewater 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 wastewater 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.).

14.2 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.

14.2.1 Raw ashes and melting/sintering materials

14.2.1.1 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. Fluidising velocity is higher than in a BFB and can be between 4.5 m/s and 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 14.2.1).

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 14.2.2).

14.2.1.2 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, 2006b). 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, 2006b). The FGT technology impacts upon the quality of the combustion residues.

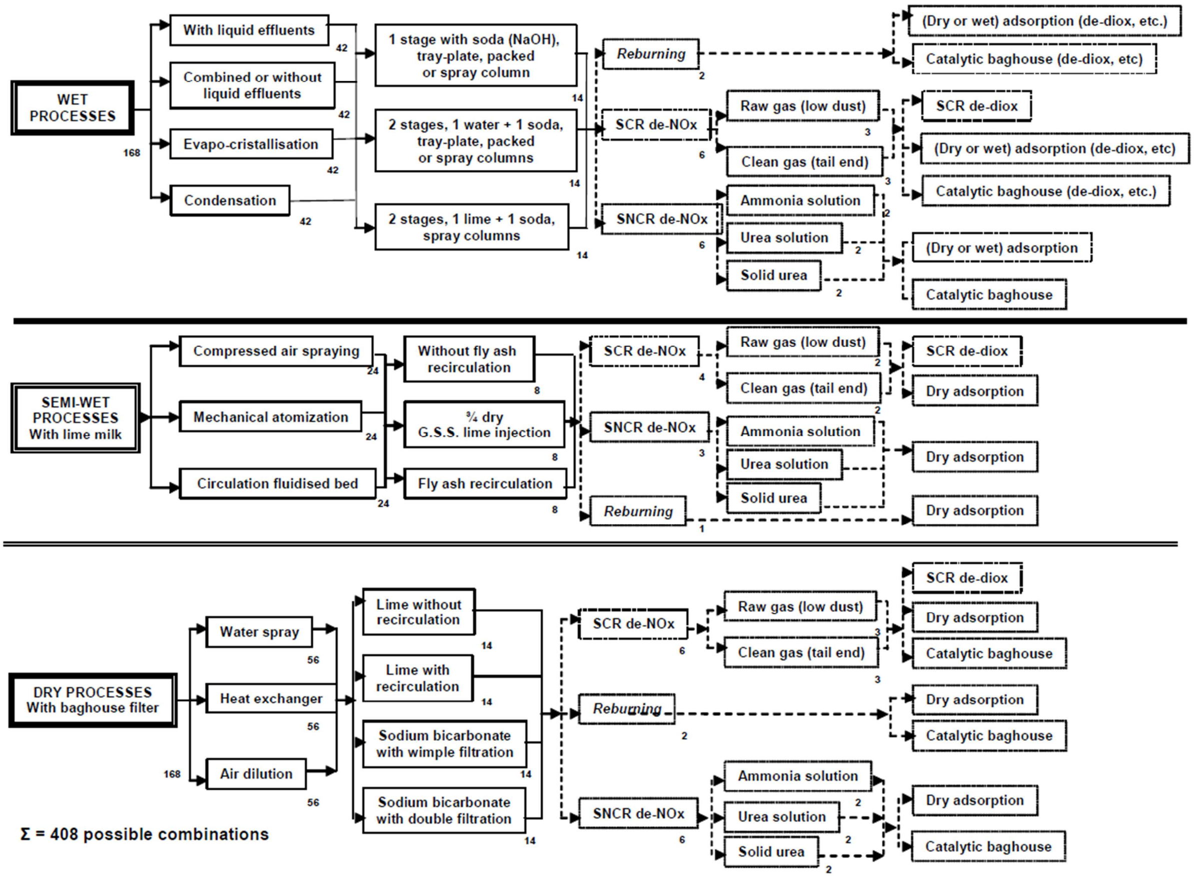


Figure 29: Overview of potential combinations of flue-gas treatment systems (European Commission, 2006b)

14.2.2 Ash and slag derivates

The use of **raw ashes** in agriculture can be associated with 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, DCP, 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 wastewater 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 wastewater is adjusted to a pH of 7 and is discharged either to a wastewater 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 aluminium 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 wastewater 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 sulphide 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 and 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 initial test results show the following recovering rates of the total potential in the sewage sludge incineration ashes: P >95% (as H_3PO_4); Fe: >70% (as iron-chloride solution), metals/metalloids > 85% as metal concentrate for metal recycling and >95% of mineral fraction applicable in cement industry. 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 wastewater 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.

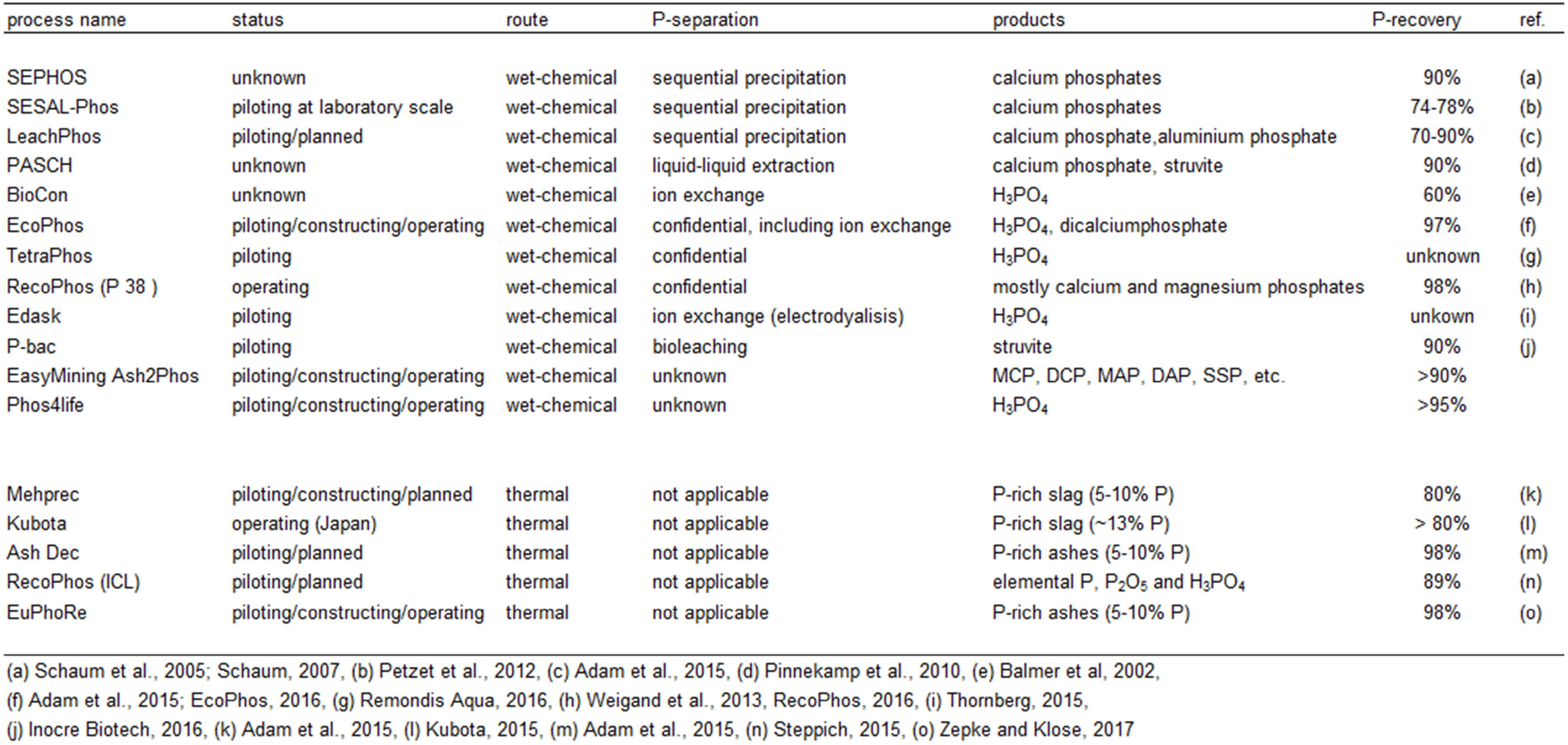
- Type II: Thermal processes
    - Nutrients can be recovered from ashes by high temperature treatments (Table 27). Processes were developed that transfer P into a metallurgical slag by reductive smelting at very high temperature temperatures in a shaft furnace (Scheidig, 2009) or that reduce P to elemental 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 be operated as a 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 wastewater 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

14.2.3 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.

14.3 Pyrolysis & gasification materials

14.3.1 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).

14.3.2 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 (e.g. Someus and Pugliese, 2018). 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.

14.3.3 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 wastewater (active carbon filter, pre-rinsing additive);
- The treatment of drinking water (micro filters);
- Other uses (exhaust filters, carbon fibres, semiconductors, etc.).

15 Chemical composition of STRUBIAS materials

15.1 Precipitated phosphate salts & derivates

15.1.1 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)

15.1.2 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)
wastewater	<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

15.2 Thermal oxidation materials & derivates

15.2.1 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 percentile		*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 percentile		*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., 2005a)
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%												
EU Fertilising Products Regulation (EU) 2019/1009												
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**		

15.2.2 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
											(mg kg^{-1})										
plant biomass																					
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 percentile		***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 percentile		***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 (bottom 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 percentile		***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 percentile		***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)
walnut 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)
	poultry 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 percentile			***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., 2005a)
	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 percentile		***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																					
Wastewater treatment sludge (raw)																					
wastewater treatment sludge (raw)	1																			0.5	(Werther et al., 2000)
wastewater treatment sludge (raw)	1																			0.5	(Wei et al., 2005)
wastewater treatment sludge (raw)	1																			0.5	STRUBIAS contribution - confidential data provider
wastewater 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)
wastewater 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
wastewater 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
wastewater 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)
wastewater 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 percentile		***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***	
Wastewater 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 percentile		***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

15.3 Pyrolysis & gasification materials

15.3.1 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					

15.3.2 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)
citrus 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

16 Methods

16.1 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 30.

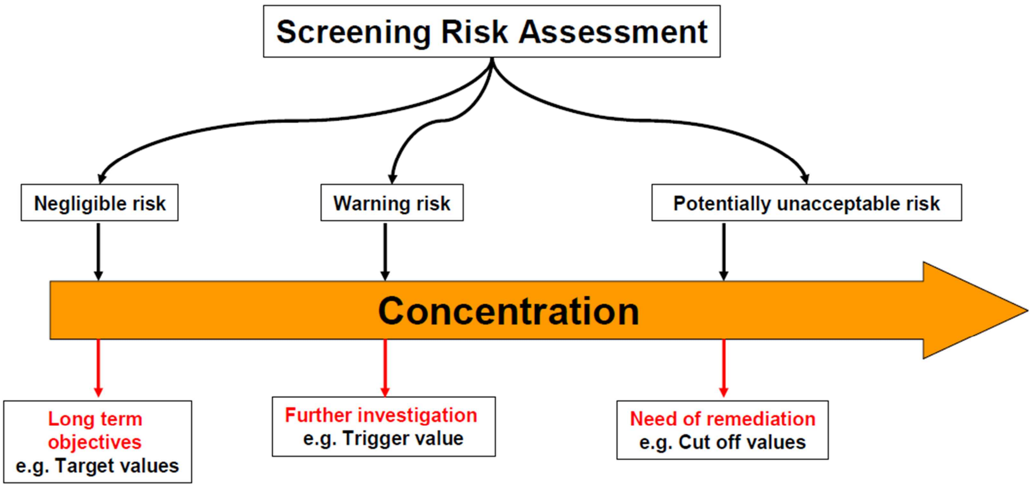


Figure 30: Derivation of soil screening values based on various risk levels and application of the different screening values (adopted from Carlon, 2007)

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 with 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).

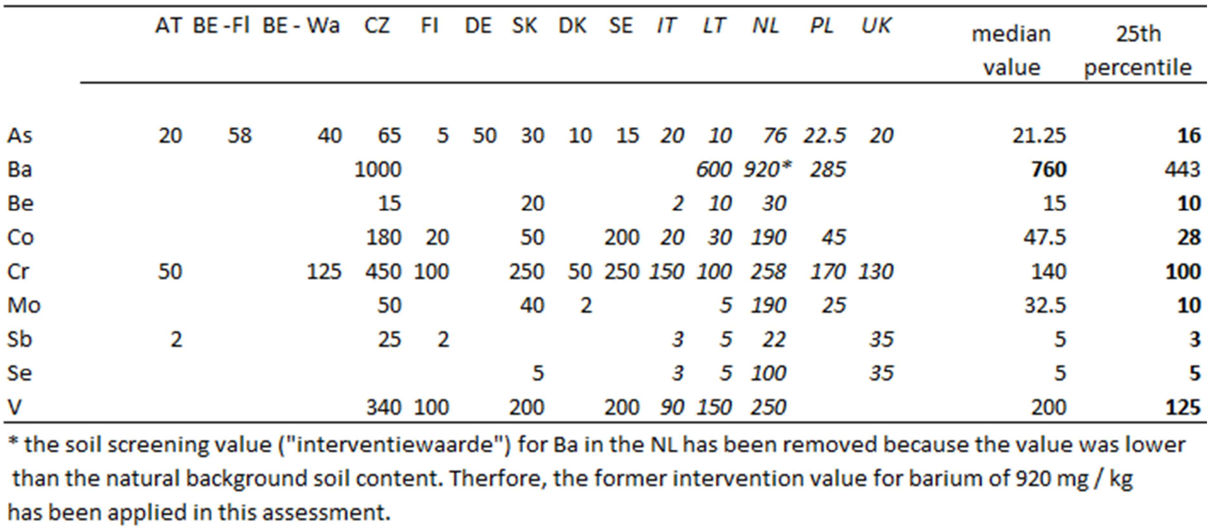
The **predicted accumulation** estimates how much of a trace metal accumulates in soil following annual applications (over years of farming) and takes into account an estimated loss of trace metals in soil from transport of the trace metal into surrounding media – a mass-balance approach (The Weinberg 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 of soil) (Source: Carlon, 2007; updated by the STRUBIAS sub-group for specific EU Member States in case of recent changes). 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*	*76*	*22.5*	*20*	21.25	**16**
Ba				1000							*600*	*920**	*285*		**760**	443
Be				15			20			*2*	*10*	*30*			15	**10**
Co				180	20		50		200	*20*	*30*	*190*	*45*		47.5	**28**
Cr	50		125	450	100		250	50	250	*150*	*100*	*258*	*170*	*130*	140	**100**
Mo				50			40	2			*5*	*190*	*25*		32.5	**10**
Sb	2			25	2					*3*	*5*	*22*		*35*	5	**3**
Se							5			*3*	*5*	*100*		*35*	5	**5**
V				340	100		200		200	*90*	*150*	*250*			200	**125**

* the soil screening value ("interventiewaarde") for Ba in the NL has been removed because the value was lower than the natural background soil content. Therfore, the former intervention value for barium of 920 mg / kg has been applied in this assessment.

- 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 lower value of 1 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}

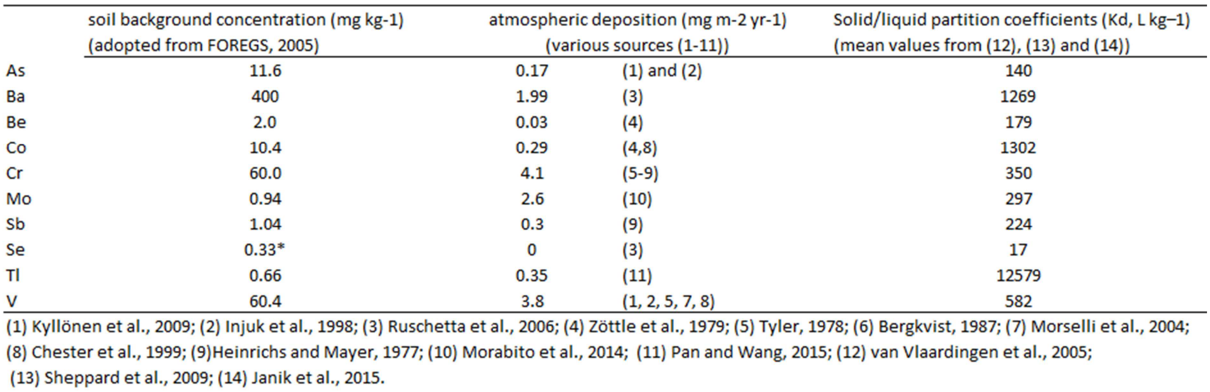
- European *median values* of metals and metalloids for soil background concentrations are used as documented in the European Soil Database from the European Commission – Joint Research Centre (Ispra), that has been developed using approximately 5000 data from topsoil samples belonging to two European databases: a) the Geochemical Atlas of Europe (http://weppi.gtk.fi/publ/foregsatlas/index.php), developed by the Forum of European Geological Surveys (FOREGS); and b) the Geochemical Mapping of Agricultural and Grazing Land Soil in Europe (GEMAS; **http://gemas.geolba.ac.at/**) (FOREGS, 2005; Reimann et al., 2014) (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 with 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. The 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.

16.2 Meta-analyses

16.2.1 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.

16.2.2 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.

16.2.3 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}).

BIBLIOGRAPHY

3R AgroCarbon (2016) 3R AgroCarbon [ONLINE]. Available at: **http://www.3ragrocarbon.com/** [Accessed 26 May 2016].

Abbiramy K.S. & Ross P.R. (2016) Avoidance Behavioural Test with Earthworm, Eisenia Fetida Exposed to the Fertilizer, Potash on Oecd Soil. *International Journal of Contemporary Research and Review* **7**: 20250-20255.

Abbiramy K.S., Ross P.R., Paramanandham J. & Thenmozhi P. (2014) Use of tropical artificial soil as a new substrate for avoidance behavioral test of earthworm, Eisenia foetida exposed to urea. *Journal of Environmental Biology* **35**: 205-210.

Abelha P., Gulyurtlu I., Boavida D., Seabra Barros J., Cabrita I., Leahy J., . . . Leahy M. (2003) Combustion of poultry litter in a fluidised bed combustor☆. *Fuel* **82**: 687-692.

Abma W.R., Driessen W., Haarhuis R. & Loosdrecht M.C.M. (2009) Upgrading of sewage treatment plant by sustainable & cost-effective separate treatment of industrial wastewater. Paper presented at the IWE Nutrient Management in Wastewater Treatment Processes, 6-9 September 2009.

Achat D.L., Daumer M.-L., Sperandio M., Santellani A.-C. & Morel C. (2014a) Solubility and mobility of phosphorus recycled from dairy effluents and pig manures in incubated soils with different characteristics. *Nutrient Cycling in Agroecosystems* **99**: 1-15.

Achat D.L., Sperandio M., Daumer M.-L., Santellani A.-C., Prud'Homme L., Akhtar M. & Morel C. (2014b) Plant-availability of phosphorus recycled from pig manures and dairy effluents as assessed by isotopic labeling techniques. *Geoderma* **232**: 24-33.

Ackerman J.N., Zvomuya F., Cicek N. & Flaten D. (2013) Evaluation of manure-derived struvite as a phosphorus source for canola. *Canadian Journal of Plant Science* **93**: 419-424.

Adam C., Eicher N., Hermann L., Herzel H., Mallon J., Schaaf M. & Stemann J. (2015) Technical comparison on the design, operation and performances of ash processes. Sustainable sewage sludge management fostering phosphorus recovery and energy efficiency (P-REX report). Berlin.

ADEME - Naskeo Rittmo Timab (2016) Démonstrateur industriel de production de struvite sur la STEP de Castres. 77 pages.

Adriano D.C., Woodford T.A. & Ciravolo T.G. (1978) Growth and Elemental Composition of Corn and Bean Seedlings as Influenced by Soil Application of Coal Ash1. *Journal of Environmental Quality* **7**: 416-421.

AEEP (2017) Data and Statistics - Fertilizers. Available at: **http://aeep.eu/data-and-statistics/fertilizers/**.

Agrotechnology Atlas (2016) Livestock manure [ONLINE], available at: **http://agro-technology-atlas.eu/**. Agro Business Park, Tjele, Denmark.

Aitken R.L. & Bell L.C. (1985) Plant uptake and phytotoxicity of boron in Australian fly ashes. *Plant and Soil* **84**: 245-257.

Akhiar A., Battimelli A., Torrijos M. & Carrere H. (2017) Comprehensive characterization of the liquid fraction of digestates from full-scale anaerobic co-digestion. *Waste Management* **59**: 118-128.

Al Seadi T. & Lukehurst C. (2012) Quality management of digestate from biogas plants used as fertiliser. IEA Bioenery Report Task 37 - Energy from Biogas.

Alaya M.N., Girgis B.S. & Mourad W.E. (2000) Activated Carbon from Some Agricultural Wastes Under Action of One-Step Steam Pyrolysis. *Journal of Porous Materials* **7**: 509-517.

Algermissen D., A. Morillon, K.-J. Arlt, N. Wolsfeld, M. Rex, M. Mauhart, . . . S. Tuo-Mikoski S. (2016) Impact of long-term application of blast furnace and steel slags as liming materials on soil fertility, crop yields and plant health (SLAGFERTILISER). European Commission, Research Programme of the Research Fund for Coal and Steel, TG9, 1. Juli 2011 - 30. June 2015, RFSR-CT-2011-00037, Draft Final Report.

Aller M.F. (2016) Biochar properties: Transport, fate, and impact. *Critical Reviews in Environmental Science and Technology* **46**: 1183-1296.

Almeida J.R.M., Bertilsson M., Gorwa-Grauslund M.F., Gorsich S. & Lidén G. (2009) Metabolic effects of furaldehydes and impacts on biotechnological processes. *Applied Microbiology and Biotechnology* **82**: 625.

Almendros G., Gonzalezvila F.J. & Martin F. (1990) FIRE-INDUCED TRANSFORMATION OF SOIL ORGANIC-MATTER FROM AN OAK FOREST - AN EXPERIMENTAL APPROACH TO THE EFFECTS OF FIRE ON HUMIC SUBSTANCES. *Soil Science* **149**: 158-168.

Almendros G., Knicker H. & González-Vila F.J. (2003) Rearrangement of carbon and nitrogen forms in peat after progressive thermal oxidation as determined by solid-state 13C- and 15N-NMR spectroscopy. *Organic Geochemistry* **34**: 1559-1568.

Alonso Camargo-Valero M., Bamelis L., De Clercq L., Delvigne F., Meers E., Michels E., . . . Williams P.T. (2015) Recycling inorganic chemicals from agro- and bio-industrial waste streams. Project/Contract number: 320J - Biorefine. Document number: BIOREFINE – WP2 – A7 – P1, 2, 5 – D.

Alotaibi K.D., Schoenau J.J. & Fonstad T. (2013) Possible utilization of ash from meat and bone meal and dried distillers grains gasification as a phosphorus fertilizer: crop growth response and changes in soil chemical properties. *Journal of Soils and Sediments* **13**: 1024-1031.

Altland J.E. & Locke J.C. (2012) Biochar Affects Macronutrient Leaching from a Soilless Substrate. *HortScience* **47**: 1136-1140.

Álvarez-Ayuso E., Querol X. & Tomás A. (2006) Environmental impact of a coal combustion-desulphurisation plant: Abatement capacity of desulphurisation process and environmental characterisation of combustion by-products. *Chemosphere* **65**: 2009-2017.

Amann A., Zoboli O., Krampe J., Rechberger H., Zessner M. & Egle L. (2018) Environmental impacts of phosphorus recovery from municipal wastewater. *Resources, Conservation and Recycling* **130**: 127-139.

Andrade A. & Schuiling R.D. (2001) The chemistry of struvite crystallization. *Mineralogical Journal of Ukraine* **23**: 37-46.

Angst T.E. & Sohi S.P. (2013) Establishing release dynamics for plant nutrients from biochar. *GCB Bioenergy* **5**: 221-226.

Antakyal D., Kuch B., Preyl V. & Steinmetz H. (2011) Effect of Micropollutants in Wastewater on Recovered Struvite. *Proceedings of the Water Environment Federation, pp. 575-582*.

Antille D.L., Sakrabani R., Tyrrel S.F., Le M.S. & Godwin R.J. (2013) Characterisation of Organomineral Fertilisers Derived from Nutrient-Enriched Biosolids Granules. *Applied and Environmental Soil Science* **2013**: 11.

Antonia López Antón M., Alan Spears D., Somoano M.D. & Rosa Martínez Tarazona M. (2013) Thallium in coal: Analysis and environmental implications. *Fuel* **105**: 13-18.

Antonini S., Arias M.A., Eichert T. & Clemens J. (2012) Greenhouse evaluation and environmental impact assessment of different urine-derived struvite fertilizers as phosphorus sources for plants. *Chemosphere* **89**: 1202-1210.

Apte A.D., Tare V. & Bose P. (2006) Extent of oxidation of Cr(III) to Cr(VI) under various conditions pertaining to natural environment. *Journal of Hazardous Materials* **128**: 164-174.

Arai Y. & Sparks D.L. (2007) Phosphate reaction dynamics in soils and soil components: A multiscale approach. *Advances in Agronomy* **94**: 135-179.

Ayers R.S. & Westcot D.W. (1985) Water quality for agriculture. FAO irrigation drainage paper No. 22-82.

Bachmann H.J., Bucheli T.D., Dieguez-Alonso A., Fabbri D., Knicker H., Schmidt H.-P., . . . Zehetner F. (2016) Toward the Standardization of Biochar Analysis: The COST Action TD1107 Interlaboratory Comparison. *Journal of Agricultural and Food Chemistry* **64**: 513-527.

Baes C.F. & Mesmer R.E. (1976) *The Hydrolysis of Cations.* . John Wiley &Sons,, New York.

Bagge E., Sahlstrom L. & Albihn A. (2005) The effect of hygienic treatment on the microbial flora of bio-waste at biogas plants. *Water Research* **39**: 4879-4886.

Bagge E., Persson M. & Johansson K.E. (2010) Diversity of spore-forming bacteria in cattle manure, slaughterhouse waste and samples from biogas plants. *Journal of Applied Microbiology* **109**: 1549-1565.

Baken S., Larsson M.A., Gustafsson J.P., Cubadda F. & Smolders E. (2012) Ageing of vanadium in soils and consequences for bioavailability. **63**: 839-847.

Ballabio C., Panagos P. & Monatanarella L. (2016) Mapping topsoil physical properties at European scale using the LUCAS database. *Geoderma* **261**: 110-123.

Balmér P., Book K., Hultman B., Jönsson H., Kärrman E., Levlin E., . . . Åberg H. (2002) System för återanvändning av fosfor ur avlopp. Report as manuscript delivered to Statens Naturvårdsverk 2002-01-14.

Bamelis L., Blancke S., Camargo-Valero M.A., De Clercq L., Haumont A., De Keulenaere F., . . . Williams P.T. (2015) Recycling inorganic chemicals from agro- and bio-industrial waste streams. Project/Contract number: 320J - Biorefine. Document number: BIOREFINE – WP2 – A5 – P1, 2, 5, 8 – D.

Barber T.R., Lutes C.C., Doorn M.R.J., Fuchsman P.C., Timmenga H.J. & Crouch R.L. (2003) Aquatic ecological risks due to cyanide releases from biomass burning. *Chemosphere* **50**: 343-348.

Barber W.P.F. (2016) Thermal hydrolysis for sewage treatment: A critical review. *Water Research* **104**: 53-71.

Barbosa R., Lapa N., Lopes H., Gulyurtlu I. & Mendes B. (2011) Stabilization/solidification of fly ashes and concrete production from bottom and circulating ashes produced in a power plant working under mono and co-combustion conditions. *Waste Management* **31**: 2009-2019.

Barnhart J. (1997) Chromium chemistry and implications for environmental fate and toxicity. In: Chromium in Soil: Perspectives in Chemistry, Health, and Environmental Regulation, Proctor et al (eds.), AEHS, CRC Lewis Publishers, Boca Raton, FL.

Barrett G.E., Alexander P.D., Robinson J.S. & Bragg N.C. (2016) Achieving environmentally sustainable growing media for soilless plant cultivation systems - A review. *Scientia Horticulturae* **212**: 220-234.

Barrow N.J. (1984) Modelling the effects of pH on phosphate sorption by soils. *Journal of Soil Science* **35**: 283-297.

Bartsch S., Breuer J., drissen P., Pischke J. & Rex M. (2014) Enrichment of Phosphorus in BOF-slag for Improved Application in Agriculture. The 7th European Oxygen Steelmaking Conference. Třinec, Czech republic, September 9-11, 2014.

Basu M., Pande M., Bhadoria P.B.S. & Mahapatra S.C. (2009) Potential fly-ash utilization in agriculture: A global review. *Progress in Natural Science* **19**: 1173-1186.

Baur R.J. (2009) Waste activated sludge stripping to remove internal phosphorus.

Beck D.A., Johnson G.R. & Spolek G.A. (2011) Amending greenroof soil with biochar to affect runoff water quantity and quality. *Environmental Pollution* **159**: 2111-2118.

Becker R., Dorgerloh U., Helmis M., Mumme J., Diakité M. & Nehls I. (2013) Hydrothermally carbonized plant materials: Patterns of volatile organic compounds detected by gas chromatography. *Bioresource Technology* **130**: 621-628.

Beesley L., Moreno-Jimenez E., Fellet G., Melo L. & Sizmuur T. (2015) Biochar and heavy metals. *Biochar environmental management - science, technology and implementation*, Lehmann J. & Joseph S. (eds.), pages. 563-594. Routledge, Oxon.

Beesley L., Moreno-Jiménez E., Gomez-Eyles J.L., Harris E., Robinson B. & Sizmur T. (2011) A review of biochars' potential role in the remediation, revegetation and restoration of contaminated soils. *Environmental Pollution* **159**: 3269-3282.

Beighle D.E., Boyazoglu P.A., Hemken R.W. & Serumagazake P.A. (1994) Determination of calcium, phosphorus, and magnesium values in rib bones from clinically normal cattle. *American Journal of Veterinary Research* **55**: 85-89.

Berg U. & Schaum C. (2005) Recovery of Phosphorus from sewage sludge and sludge ashes - Applications in Germany and Northern Europe. First National Sludge Symposium NSS 2005, Izmir, Turkey.

Bergkvist B.O. (1987) Soil solution chemistry and metal budgets of spruce forest ecosystems in S. Sweden. *Water, Air, and Soil Pollution* **33**: 131-154.

Bhuiyan M.I.H., Mavinic D.S. & Koch F.A. (2008) Thermal decomposition of struvite and its phase transition. *Chemosphere* **70**: 1347-1356.

Bicudo J.R. (2009) Animal solid manure: storage, handling and disposal. *Agricultural Mechanization and Automation*, McNulty P. & Grace P.M. (eds.), pages. 265-291. Encyclopedia of Life Support Systems, Paris.

Biederman L.A. & Harpole W.S. (2013) Biochar and its effects on plant productivity and nutrient cycling: a meta-analysis. *Global Change Biology Bioenergy* **5**: 202-214.

Billen P., Costa J., Van der Aa L., Van Caneghem J. & Vandecasteele C. (2015) Electricity from poultry manure: a cleaner alternative to direct land application. *Journal of Cleaner Production* **96**: 467-475.

Bio Intelligence Service - Umweltbundesamt - AEA (2010) Preparatory study on food waste across EU27, Contract #: 07.0307/2009/540024/SER/G4 for the European Commission. Brussels.

BIOMACON (2016) BIOMACON [ONLINE]. Available at: **http://www.biomacon.com/index-e.html** [Accessed 26 May 2016].

Bird S.C. & Drizo A. (2009) Investigations on phosphorus recovery and reuse as soil amendment from electric arc furnace slag filters. *Journal of Environmental Science and Health Part a-Toxic/Hazardous Substances & Environmental Engineering* **44**: 1476-1483.

Bischel H.N., Özel Duygan B.D., Strande L., McArdell C.S., Udert K.M. & Kohn T. (2015) Pathogens and pharmaceuticals in source-separated urine in eThekwini, South Africa. *Water Research* **85**: 57-65.

Black Carbon (2016) Black Carbon - a biochar technology [ONLINE]. Available at: **http://www.http://blackcarbon.dk/Biochar** [Accessed 26 May 2016].

Blackwell P., Riethmuller G. & Collins M. (2009) Biochar application to soil. *Biochar for Environmental Management: Science and Technology*, Lehmann J. & Joseph S. (eds.), pages. Earthscan, London.

Blake J.P. & Hess J.B. (2014) Poultry litter ash as a replacement for dicalcium phosphate in broiler diets. *The Journal of Applied Poultry Research* **23**: 101-107.

Blissett R.S. & Rowson N.A. (2012) A review of the multi-component utilisation of coal fly ash. *Fuel* **97**: 1-23.

BMA (2015) Mestverwerkingsloket, Bureau Mest Afzet. Landelijke inventarisatie mestverwerkingscapaciteit, BMA, The Netherlands.

Boateng A.A., Garcia-Perez M., Masek O., Brown R.A. & del Campo B. (2015) Biochar production technology. *Biochar environmental management - science, technology and implementation*, Lehmann J. & Joseph S. (eds.), pages. 63-88. Routledge, Oxon.

Bogush A., Stegemann J.A., Wood I. & Roy A. (2015) Element composition and mineralogical characterisation of air pollution control residue from UK energy-from-waste facilities. *Waste Management* **36**: 119-129.

Bolland M.D.A. & Gilkes R.J. (1990) Rock phosphates are not effective fertilizers in Western Australian soils: A review of one hundred years of research. *Fertilizer research* **22**: 79-95.

Bonvin C., Etter B., Udert K.M., Frossard E., Nanzer S., Tamburini F. & Oberson A. (2015) Plant uptake of phosphorus and nitrogen recycled from synthetic source-separated urine. *Ambio* **44**: S217-S227.

Borenstein M., Hedges L.V., Higgins J.P.T. & Rothstein H.R. (2009) *Introduction to meta-analysis*. John Wiley & Sons, Ltd, Chichester, UK.

Bouzas A., Doñate S., Barat R., Martí N., Borrás L., Grau S., . . . Pastor L. (2016) Implementation of a P-recovery system in Calahorra wastewater treatment plant. Proceedings of the 13th IWA Leading Edge Conference on Water and Wastewater Technologies. 13-16 June, 2016. Jerez de la Frontera (Spain).

Bowden-Green B. & Briens L. (2016) An investigation of drum granulation of biochar powder. *Powder Technology* **288**: 249-254.

Boxall A.B.A. (2012) New and Emerging Water Pollutants arising from Agriculture. OECD Report.

Bradford-Hartke Z., Lane J., Lant P. & Leslie G. (2015) Environmental Benefits and Burdens of Phosphorus Recovery from Municipal Wastewater. *Environmental Science & Technology* **49**: 8611-8622.

Branca T.A., Pistocchi C., Colla V., Ragaglini G., Amato A., Tozzini C., . . . Romaniello L. (2014) Investigation of (BOF) Converter slag use for agriculture in europe. *Metallurgical Research & Technology* **111**: 155-167.

Brennan A., Jiménez E.M., Puschenreiter M., Alburquerque J.A. & Switzer C. (2014) Effects of biochar amendment on root traits and contaminant availability of maize plants in a copper and arsenic impacted soil. *Plant and Soil* **379**: 351-360.

Brentrup F. & Pallière C. (2010) Nitrogen Use Efficiency as an Agro-Environmental Indicator. Proceedings of the OECD Workshop on OECD Agri-environmental Indicators: Lessons Learned and Future Directions, 23-26 March, 2010, Leysin, Switzerland. OECD.

Brink N. (1993) Composting of food waste and catching nitrogen. *Acta Agriculturae Scandinavica* **43**: 114-120.

Brod E., Haraldsen T.K. & Breland T.A. (2012) Fertilization effects of organic waste resources and bottom wood ash: results from a pot experiment. *Agricultural and Food Science* **21**: 332-347.

Brod E., Øgaard A.F., Krogstad T., Haraldsen T.K., Frossard E. & Oberson A. (2016) Drivers of Phosphorus Uptake by Barley Following Secondary Resource Application. *Frontiers in Nutrition* **3**.

Brown R.A., del Camp B., Boateng A.A., Garcia-Perez M. & Masek O. (2015) Fundamentals of biochar production. *Biochar environmental management - science, technology and implementation*, Lehmann J. & Joseph S. (eds.), pages. 39-61. Routledge, Oxon.

Bruun E.W., Ambus P., Egsgaard H. & Hauggaard-Nielsen H. (2012) Effects of slow and fast pyrolysis biochar on soil C and N turnover dynamics. *Soil Biology and Biochemistry* **46**: 73-79.

Bruun S., Hansen T.L., Christensen T.H., Magid J. & Jensen L.S. (2006) Application of processed organic municipal solid waste on agricultural land – a scenario analysis. *Environmental Modeling & Assessment* **11**: 251-265.

Bryers R.W. (1996) Fireside slagging, fouling, and high-temperature corrosion of heat-transfer surface due to impurities in steam-raising fuels. *Progress in Energy and Combustion Science* **22**: 29-120.

Bucheli T.D., Hilber I. & Schmidt H.P. (2015) Polycyclic aromatic hydrocarbons and polychlorinated aromatic compounds in biochar. *Biochar environmental management - science, technology and implementation*, Lehmann J. & Joseph S. (eds.), pages. 595-624. Routledge, Oxon.

Buck J.K., Honston R.J. & Beimborn W.A. (1990) Direct seedling of anthracite refuge using coal flyash as a major soil amendment. In: Proceedings of the mining and reclamation conference and exhibition. West Virginia Univ. Pub. Service No. 2; . pages. 603.

Buckwell A. & Nadeu E. (2016) Nutrient recovery and reuse (NRR) in European agriculture. A review of the issues, opportunities, and actions. Brussels.

Buczek S.B., Cope W.G., McLaughlin R.A. & Kwak T.J. (2017) Acute toxicity of polyacrylamide flocculants to early life stages of freshwater mussels. *Environmental Toxicology and Chemistry* **36**: 2715-2721.

Budai A., Wang L., Gronli M., Strand L.T., Antal M.J., Abiven S., . . . Rasse D.P. (2014) Surface Properties and Chemical Composition of Corncob and Miscanthus Biochars: Effects of Production Temperature and Method. *Journal of Agricultural and Food Chemistry* **62**: 3791-3799.

Busch D., Kammann C., Grünhage L. & Müller C. (2012) Simple Biotoxicity Tests for Evaluation of Carbonaceous Soil Additives: Establishment and Reproducibility of Four Test Procedures. *Journal of Environmental Quality* **41**: 1023-1032.

Buss W. & Mašek O. (2014) Mobile organic compounds in biochar – A potential source of contamination – Phytotoxic effects on cress seed (Lepidium sativum) germination. *Journal of Environmental Management* **137**: 111-119.

Buss W. & Mašek O. (2016) High-VOC biochar—effectiveness of post-treatment measures and potential health risks related to handling and storage. *Environmental Science and Pollution Research* **23**: 19580-19589.

Buss W., Mašek O., Graham M. & Wüst D. (2015) Inherent organic compounds in biochar–Their content, composition and potential toxic effects. *Journal of Environmental Management* **156**: 150-157.

Buss W., Graham M.C., MacKinnon G. & Masek O. (2016) Strategies for producing biochars with minimum PAH contamination. *Journal of Analytical and Applied Pyrolysis* **119**: 24-30.

Butkovskyi A. (2016) Overview of organic contaminants in municipal sewage biosolids and approaches and treatments to decrease these. Workshop on pharmaceuticals and organic chemicals in sewage biosolids: questions for recycling, Malmö, 27 October, 2016.

Cabeza R., Steingrobe B., Römer W. & Claassen N. (2011) Effectiveness of recycled P products as P fertilizers, as evaluated in pot experiments. *Nutrient Cycling in Agroecosystems* **91**: 173-184.

Camargo-Valero M.A., De Clercq L., Delvigne F., Haumont A., Lebuf V., Meers E., . . . Williams P.T. (2015) Recycling inorganic chemicals from agro- and bio-industrial waste streams. Project/Contract number: 320J - Biorefine Document number: BIOREFINE – WP2 – A6 – P1, 2, 3, 5, 8 – D.

Camps-Arbestain M., Amonette J.E., Singh B.P., Wang T. & Schmidt H.P. (2015) A biochar classification system and assocaited test methods. *Biochar environmental management - science, technology and implementation*, Lehmann J. & Joseph S. (eds.), pages. 165-193. Routledge, Oxon.

Camps-Arbestain M., Shen Q., Wang T., van Zwieten L. & Novak J.M. (2017) Available nutrients in biochar. *Biochar - a guide to analytical methods*, Singh B.P., Camps-Arbestain M. & Lehmann J. (eds.), pages. 109-126. CRC Press, Boca Raton, FL.

Canadian Council of Ministers of the Environment (2003) Soil Quality Guidelines for the Protection of Environmental and Human Health. Available at **https://www.ccme.ca/en/resources/canadian_environmental_quality_guidelines/**.

Canadian Council of Ministers of the Environment (CCME) (2003) Summary of existing Canadian environmental quality guidelines.

Cantrell K.B., Hunt P.G., Uchimiya M., Novak J.M. & Ro K.S. (2012) Impact of pyrolysis temperature and manure source on physicochemical characteristics of biochar. *Bioresource Technology* **107**: 419-428.

Carliell-Marquet C. & Cooper J. (2014) Towards closed-loop phosphorus management for the UK Water Industry. In Sustainable Phosphorus Summit, 1-3 September 2014.

Carlon C. (2007) Derivation methods of soil screening values in Europe - a review and evaluation of national procedures towards harmonisation. European Commission, Joint Research Centre, Ispra, EUR 22805-EN, 306 pp.

Carlson C.L., Adriano D.C., Sajwan K.S., Abels S.L., Thoma D.P. & Driver J.T. (1991) Effects of selected trace metals on germinating seeds of six plant species. *Water, Air, and Soil Pollution* **59**: 231-240.

CBS Statistics Netherlands (2015) Urban wastewater treatment per province and river basin district, 1981-2013 Statistics Netherlands (CBS).

Centre E. (2017) Ecoinvent v3.3 database. Zurich, Switserland: Swiss Centre for Life Cycle Inventories. Available at: **http://www.ecoinvent.org/database/ecoinvent-version-3/introduction/**.

Cerrillo M., Palatsi J., Comas J., Vicens J. & Bonmati A. (2015) Struvite precipitation as a technology to be integrated in a manure anaerobic digestion treatment plant - removal efficiency, crystal characterization and agricultural assessment. *Journal of Chemical Technology and Biotechnology* **90**: 1135-1143.

Chabbi A., Lehmann J., Ciais P., Loescher H.W., Cotrufo M.F., Don A., . . . Rumpel C. (2017) Aligning agriculture and climate policy. *Nature Climate Change* **7**: 307.

Chandrajith R. & Dissanayake C.B. (2009) Phosphate mineral fertilizers, trace metals and human health. *Journal of the National Science Foundation of Sri Lanka* **37**: 153-165.

Chandrappa R. & Das D.B. (2012) Waste quantities and characteristics. *Solid waste management, Environmental Science and Engineering*, Chandrappa R. & Das D.B. (eds.), pages. 47-63. Springer-Verlag, Berlin Heidelberg.

Chang A.C., Lund L.J., Page A.L. & Warneke J.E. (1977) Physical Properties of Fly Ash-Amended Soils1. *Journal of Environmental Quality* **6**: 267-270.

Chang L.W., Magos L. & Suzuki T. (1996) *Toxicology of Metals*. CRC Press, Boca Raton. FL, USA.

Charles W., Cord-Ruwisch R., Ho G., Costa M. & Spencer P. (2006) Solutions to a combined problem of excessive hydrogen sulfide in biogas and struvite scaling. *Water Science and Technology* **53**: 203-211.

Chaurand P., Rose J., Domas J. & Bottero J.-Y. (2006) Speciation of Cr and V within BOF steel slag reused in road constructions. *Journal of Geochemical Exploration* **88**: 10-14.

Chen Q.-L., An X.-L., Zhu Y.-G., Su J.-Q., Gillings M.R., Ye Z.-L. & Cui L. (2017) Application of Struvite Alters the Antibiotic Resistome in Soil, Rhizosphere, and Phyllosphere. *Environmental Science & Technology* **51**: 8149-8157.

Chen Q., An X., Li H., Su J., Ma Y. & Zhu Y.-G. (2016) Long-term field application of sewage sludge increases the abundance of antibiotic resistance genes in soil. *Environment International* **92-93**: 1-10.

Cheremisina E. & Schenk J. (2017) Chromium Stability in Steel Slags. **88**: 1700206.

Chester R., Nimmo M. & Preston M.R. (1999) The trace metal chemistry of atmospheric dry deposition samples collected at Cap Ferrat: a coastal site in the Western Mediterranean. *Marine Chemistry* **68**: 15-30.

Chia C.H., Downie A. & Munroe P. (2015) Characteristics of biochar: physical and structural properties. *Biochar environmental management - science, technology and implementation*, Lehmann J. & Joseph S. (eds.), pages. 89-110. Routledge, Oxon.

Churka Blum S., Lehmann J., Solomon D., Caires E.F. & Alleoni L.R.F. (2013) Sulfur forms in organic substrates affecting S mineralization in soil. *Geoderma* **200–201**: 156-164.

Clements W.H., Stahl R.G. & Landis R.C. (2015) Ecological Effects of Biochar on the Structure and Function of Stream Benthic Communities. *Environmental Science & Technology* **49**: 14649-14654.

Cleveland C.C. & Liptzin D. (2007) C : N : P stoichiometry in soil: is there a "Redfield ratio" for the microbial biomass? *Biogeochemistry* **85**: 235-252.

Codling E.E., Chaney R.L. & Sherwell J. (2002) Poultry litter ash as a potential phosphorus source for agricultural crops. *Journal of Environmental Quality* **31**: 954-961.

Collins H. (2008) Use of biochar from teh pyrolsyis of waste organic material as a soil amendment: laboratory and greenhouse analayses. A quarterly progress report prepared for the biochar project. USDA-ARS, Prosser, WA.

Collins H.P., Streubel J., Alva A., Porter L. & Chaves B. (2013) Phosphorus Uptake by Potato from Biochar Amended with Anaerobic Digested Dairy Manure Effluent. *Agronomy Journal* **105**: 989-998.

Conesa J.A., Fullana A. & Font R. (2003) Thermal decomposition of meat and bone meal. *Journal of Analytical and Applied Pyrolysis* **70**: 619-630.

Connolly E.L. & Guerinot M.L. (2002) Iron stress in plants. *Genome Biology* **3**: reviews1024.1021-reviews1024.1024.

Conti R., Fabbri D., Vassura I. & Ferroni L. (2016) Comparison of chemical and physical indices of thermal stability of biochars from different biomass by analytical pyrolysis and thermogravimetry. *Journal of Analytical and Applied Pyrolysis* **122**: 160-168.

Cornel P. & Schaum C. (2009) Phosphorus recovery from wastewater: needs, technologies and costs. *Water Science and Technology* **59**: 1069-1076.

Cornelis G., Johnson C.A., Gerven T.V. & Vandecasteele C. (2008) Leaching mechanisms of oxyanionic metalloid and metal species in alkaline solid wastes: A review. *Applied Geochemistry* **23**: 955-976.

Coutand M., Cyr M., Deydier E., Guilet R. & Clastres P. (2008) Characteristics of industrial and laboratory meat and bone meal ashes and their potential applications. *Journal of Hazardous Materials* **150**: 522-532.

Cox A.E., Camberato J.J. & Smith B.R. (1997) Phosphate availability and inorganic transformation in an alum sludge-affected soil. *Journal of Environmental Quality* **26**: 1393-1398.

Crittenden S., Daumer M.L., Gésan-Guiziou G. & Gaucheron F. (2008) Treatment and Recycling of Phosphorus in Effluent From the Dairy and Agri-Food Industries. Cemagref - BBA Report, France.

Crombie K., Mašek O., Sohi S.P., Brownsort P. & Cross A. (2013) The effect of pyrolysis conditions on biochar stability as determined by three methods. *GCB Bioenergy* **5**: 122-131.

Crossgrove J. & Zheng W. (2004) Manganese toxicity upon overexposure. *NMR in biomedicine* **17**: 544-553.

Cyr M. & Ludmann C. (2006) Low risk meat and bone meal (MBM) bottom ash in mortars as sand replacement. *Cement and Concrete Research* **36**: 469-480.

Czaja Z. & Hermann J. (2011) Adverse environmental impact caused by thermal destruction of animal byproducts. *Contemporary problems of management and environmental protection*, Skibniewska K.A. (ed.), pages. 89-98. Zakład poligraficzny UWM w Olsztynie.

Daghrir R. & Drogui P. (2013) Tetracycline antibiotics in the environment: a review. *Environmental Chemistry Letters* **11**: 209-227.

Davidson D.F. & Sheldon R.P. (1986) Phosphate Deposits of the World: Volume 2, Phosphate Rock Resources.

de Boer M.A., Hammerton M. & Slootweg J.C. (2018) Uptake of pharmaceuticals by sorbent-amended struvite fertilisers recovered from human urine and their bioaccumulation in tomato fruit. *Water Research* **133**: 19-26.

de Brogniez D., Ballabio C., Stevens A., Jones R.J.A., Montanarella L. & van Wesemael B. (2015) A map of the topsoil organic carbon content of Europe generated by a generalized additive model. *European Journal of Soil Science* **66**: 121-134.

De Clercq L., Michels E., Meers E., Buysse J. & Haumont A. (2015) Legal framework of recovered phosphorus (struvite) as fertiliser in North-Western Europe. Document number: BIOREFINE – WP5 – A19 – P1 - D. Ghent, Belgium.

De Graaff L., Odegard I. & Nusselder S. (2017) LCA of thermal conversion of poultry litter at BMC Moerdijk. CE Delft, Publication code 17.2H94.01.

de las Fuentes L., Sanders B., Klemes J. & Urbaniec K. (2003) European Project AWARENET: Agro-food WAstes minimisation and REduction NETwork. 22nd general assembly of the International Commission for Sugar Technology, Madrid.

de Ruijter F.J., van Dijk W., van Middelkoop J.C. & van Reuler H. (2015) Phosphorus recycling from the waste sector. Wagenigen UR, PRI report 641.

De Saeger S., Audenaert K. & Croubels S. (2016) Report from the 5th International Symposium on Mycotoxins and Toxigenic Moulds: Challenges and Perspectives (MYTOX) Held in Ghent, Belgium, May 2016. *Toxins* **8**: 146.

De Temmerman L., Waegeneers N., Thiry C., Du Laing G., Tack F. & Ruttens A. (2014) Selenium content of Belgian cultivated soils and its uptake by field crops and vegetables. *Science of the Total Environment* **468–469**: 77-82.

de Vries W., Leip A., Reinds G.J., Kros J., Lesschen J.P. & Bouwman A.F. (2011) Comparison of land nitrogen budgets for European agriculture by various modeling approaches. *Environmental Pollution* **159**: 3254-3268.

De Windt L., Chaurand P. & Rose J. (2011) Kinetics of steel slag leaching: Batch tests and modeling. *Waste Management* **31**: 225-235.

Decrey L., Udert K.M., Tilley E., Pecson B.M. & Kohn T. (2011) Fate of the pathogen indicators phage ΦX174 and Ascaris suum eggs during the production of struvite fertilizer from source-separated urine. *Water Research* **45**: 4960-4972.

Degryse F., Baird R., da Silva R.C. & McLaughlin M.J. (2017) Dissolution rate and agronomic effectiveness of struvite fertilizers – effect of soil pH, granulation and base excess. *Plant and Soil* **410**: 139-152.

Delin S. (2016) Fertilizer value of phosphorus in different residues. *Soil Use and Management* **32**: 17-26.

Demeyer A., Voundi Nkana J.C. & Verloo M.G. (2001) Characteristics of wood ash and influence on soil properties and nutrient uptake: an overview. *Bioresource Technology* **77**: 287-295.

Demirbas A. (2003) Trace metal concentrations in ashes from various types of biomass species. *Energy Sources* **25**: 743-751.

Demirbas A. (2004) Combustion characteristics of different biomass fuels. *Progress in Energy and Combustion Science* **30**: 219-230.

Demirbas A. (2005) Potential applications of renewable energy sources, biomass combustion problems in boiler power systems and combustion related environmental issues. *Progress in Energy and Combustion Science* **31**: 171-192.

Desmidt E., Ghyselbrecht K., Zhang Y., Pinoy L., Van der Bruggen B., Verstraete W., . . . Meesschaert B. (2015) Global Phosphorus Scarcity and Full-Scale P-Recovery

Techniques: A Review. *Critical Reviews in Environmental Science and Technology* **45**: 336-384.

Dewaele C. (2015) NuReSys - from P-recovery to fertilizer production.

Deydier E., Guilet R., Sarda S. & Sharrock P. (2005a) Physical and chemical characterisation of crude meat and bone meal combustion residue: “waste or raw material?”. *Journal of Hazardous Materials* **121**: 141-148.

Deydier E., Guilet R., Sarda S. & Sharrock P. (2005b) Physical and chemical characterisation of crude meat and bone meal combustion residue: "waste or raw material?". *Journal of Hazardous Materials* **121**: 141-148.

Dikov G., Hermann L., Hukari S., Muskolus A., Nattorp A., Pokorna M. & Gonzales de Ubieta C. (2014) Pre-Normative matrix. Review of fertilization schemes. Review of current legal framework for phosphorus recovery. Sustainable sewage sludge management fostering phosphorus recovery and energy efficiency (P-REX report).

Dobbelare D. (2017) Statistical overview of teh animal by-products industry in the EU in 2016. European Fat Processors and Renderers Association (EFPRA) Congress, 2 June 2017, Hamburg.

Domene X., Enders A., Hanley K. & Lehmann J. (2015) Ecotoxicological characterization of biochars: Role of feedstock and pyrolysis temperature. *Science of the Total Environment* **512-513**: 552-561.

Donatello S. & Cheeseman C.R. (2013) Recycling and recovery routes for incinerated sewage sludge ash (ISSA): A review. *Waste Management* **33**: 2328-2340.

Donatello S., Tyrer M. & Cheeseman C.R. (2010) EU landfill waste acceptance criteria and EU Hazardous Waste Directive compliance testing of incinerated sewage sludge ash. *Waste Management* **30**: 63-71.

Dong J., Wu F. & Zhang G. (2006) Influence of cadmium on antioxidant capacity and four microelement concentrations in tomato seedlings (Lycopersicon esculentum). *Chemosphere* **64**: 1659-1666.

Doring R. & Borrello A. (2014) Economic Performance of the EU Fish Processing Industry Sector - Scientific, Technical and Economic Committee for Fisheries (STECF).

Drogs B., Fuchs W., Al Saedi T., Madsen M. & Kinke B. (2015) Nutrient Recovery by Biogas Digestate Processing. IEA Bioenergy Report.

Drosg B., Fugs W., Al Seadi T., Madsen M. & Linke B. (2015) Nutrient Recovery by Biogas Digestate Processing - IEA Bioenergy report.

Duboc O., Santner J., Fard A.G., Zehetner F., Tacconi J. & Wenzel W.W. (2017) Predicting phosphorus availability from chemically diverse conventional and recycling fertilizers. *Science of the Total Environment* **599**: 1160-1170.

Dumroese R.K., Heiskanen J., Englund K. & Tervahauta A. (2011) Pelleted biochar: Chemical and physical properties show potential use as a substrate in container nurseries. *Biomass & Bioenergy* **35**: 2018-2027.

Dutta T., Kwon E., Bhattacharya S.S., Jeon B.H., Deep A., Uchimiya M. & Kim K.-H. (2016) Polycyclic aromatic hydrocarbons and volatile organic compounds in biochar and biochar-amended soil: a review. *GCB Bioenergy* n/a-n/a.

DWA (2005) Stand der Slarschlammbehandlung und Entsorgung in Deutschland, p. 66.

Dzonzi-Unidm J., Masek O. & Abass O. (2012) Determination of Spontaneous Ignition Behaviour

of Biochar Accumulations. *International Journal of Science and Research* **3**: 656-662.

E-PRTR (2013) European Pollutant Release and Transfer Register (E-PRTR).

EBC (2012) 'European Biochar Certificate - Guidelines for a Sustainable Production of Biochar.' European Biochar Foundation (EBC), Arbaz, Switzerland. **http://www.europeanbiochar.org/en/download**. Version 6.2E of 04th February 2016, DOI: 10.13140/RG.2.1.4658.7043.

ECHA (2010) Guidance on waste and recovered substances. Helsinki.

ECHA (2016) Guidance for identification and naming of substances under REACH and CLP, December 2016, Version 2.0. Available at: **https://echa.europa.eu/documents/10162/23036412/substance_id_en.pdf/ee696bad-49f6-4fec-b8b7-2c3706113c7d**.

ECN (2017) Phyllis database. ECN, Biomass Coal & Environmental Research, Petten.

EcoPhos (2016) EcoPhos - Our technology [ONLINE], Available at: **http://www.ecophos.com/#/en/technology/** [Accessed 20 May 2016].

Ecophos (2018) The Ecophos process for producing dihydrate decalcium phosphate: What makes the difference? Available at: **www.aliphos.com/wp-content/.../The-Ecophos-process-for-producing-dihydrate.pdf**.

EFPRA (2017) The facts about rendering. Available at: **http://efpra.eu/wp-content/uploads/2016/11/The-facts-about-rendering.pdf**.

EFSA (2004) Opinion of the Scientific Panel on Dietetic Products, Nutrition and Allergies on a request from the Commission related to the Tolerable Upper Intake Level of Boron (Sodium Borate and Boric Acid) (Request N° EFSA-Q-2003-018). *The EFSA Journal* **80**: 1-22.

Eghball B., Wienhold B.J., Gilley J.E. & Eigenberg R.A. (2002) Mineralization of manure nutrients. *Journal of Soil and Water Conservation* **57**: 470-473.

Egle L., Rechberger H., Krampe J. & Zessner M. (2016) Phosphorus recovery from municipal wastewater: An integrated comparative technological, environmental and economic assessment of P recovery technologies. *Science of the Total Environment* **571**: 522-542.

Ehlert P.A.I. (2017) Agronomic effectivity of poultry litter ash comparison of results of pot- and incubation experiments with data from literature. Wageningen Uniersity.

Ehlert P.A.I. & Nelemans J.A. (2015a) Efficacy of potassium of hydrated poultry litter ash. Alterra Wageningen UR, Wageningen.

Ehlert P.A.I. & Nelemans J.A. (2015b) Efficacy of phosphorus of hydrated poultry litter ash phosphorus use efficiency of rye grass - Phosphorus use efficiency of rye grass. Alter ra Wageningen UR. Wageningen.

Ehlert P.A.I. & Nelemans J.A. (2015c) Efficacy of phosphorus of hydrated poultry litter ash; Phosphorus use efficiency of green been. Wageningen, Alterra Wageningen UR (University & Research centre), the Netherlands, Alterra report. Commissioned by BMC Moerdijk (**https://www.bmcmoerdijk.nl/en/home.htm**).

Ehlert P.A.I. & Nelemans J.A. (2015d) Efficacy of phosphorus of hydrated poultry litter ash; Phosphorus use efficiency of rye grass. Wageningen, Alterra Wageningen UR

(University & Research centre), the Netherlands, Alterra report. Commissioned by BMC Moerdijk (**https://www.bmcmoerdijk.nl/en/home.htm**).

Ehlert P.A.I., Van Dijk K.C. & Oenema O. (2016a) Towards a legal status for struvite in the Netherlands. Wageningen, Statutory Research Tasks Nature & Environment. Wageningen, Wettelijke Onderzoekstaken Natuur & Milieu, WOt-technical report 69.

Ehlert P.A.I., van Wijnen H.J., Struijs J., Dijk T.A.v., van Scholl L. & de Poorter L.R.M. (2016b) Risicobeoordeling van contaminanten in afval- en reststoffen bestemd voor gebruik als covergistingsmateriaal. Wot-technical report 70, Wageningen, NL.

Elliott A.L., Davis J.G., Waskom R.M., Self J.R. & Christensen D.K. (2007) Phosphorus fertilizers for organic farming systems. Publ. No. 0.569. Colorado State Univ. Coop. Ext., Fort Collins, CO.

Elser J. & Bennett E. (2011) Phosphorus cycle: A broken biogeochemical cycle. *Nature* **478**: 29-31.

Enders A., Hanley K., Whitman T., Joseph S. & Lehmann J. (2012) Characterization of biochars to evaluate recalcitrance and agronomic performance. *Bioresource Technology* **114**: 644-653.

Enell A., Fuhrman F., Lundin L., Warfvinge P. & Thelin G. (2008) Polycyclic aromatic hydrocarbons in ash: Determination of total and leachable concentrations. *Environmental Pollution* **152**: 285-292.

Energy Information Administration of the United States (2001) Emissions of Greenhouse Gases in the United States 2000. Washington.

EPA (2003a) Ecological Screening Level for Aluminum. U.S. Environmental Protection Agency. OSWER Directive 9285.7-60.

EPA (2003b) Ecological Screening Level for Iron. U.S. Environmental Protection Agency. OSWER Directive 9285.7-60.

EPA (2003c) Ecological Screening Level for Manganese. U.S. Environmental Protection Agency. OSWER Directive 9285.7-60.

Eriksson J. (2001) Concentrations of 61 trace elements in sewage sludge, farmyard manure, mineral fertiliser, precipitation and in oil and crops. Swedish Environmental Protection Agency, Naturvårdsverkets reprocentral 2001/10.

Escher B.I., Pronk W., Suter M.J.F. & Maurer M. (2006) Monitoring the removal efficiency of pharmaceuticals and hormones in different treatment processes of source-separated urine with bioassays. *Environmental Science & Technology* **40**: 5095-5101.

ESDAT (2017) Environmental Database Management Software. 7**http://www.esdat.net/Environmental%20Standards/Australia/NEPM%20Tables.pdf**.

ESPP (2016) Scope Newsletter n°122, November 2016. Brussels. available at: **http://phosphorusplatform.eu/images/scope/ScopeNewsletter122.pdf**.

ESPP (2017) Scope Newsletter n°124, February 2017. Brussels. available at: **http://phosphorusplatform.eu/images/scope/ScopeNewsletter124.pdf**.

ETC/SIA (2013) Review of the EU bioenergy potential from a resource efficiency perspective. Background report, Alterra, Wageningen.

Etiegni L. & Campbell A.G. (1991) PHYSICAL AND CHEMICAL CHARACTERISTICS OF WOOD ASH. *Bioresource Technology* **37**: 173-178.

EUBIA (2015) Bio-based Char Market Potentials - An emerging large scale commodity in Europe. EUBIA Report. Brussels, Belgium.

EURAR (2008) European Union Risk Assessment Report - diantimony trioxide.

European Chemicals Bureau (2005) Chromium trioxide, sodium chromate, sodium dichromate, ammonium dichromate, potassium dichromate. European Union Risk Assessment Report. Available at **https://echa.europa.eu/documents/10162/3be377f2-cb05-455f-b620-af3cbe2d570b**.

European Climate Foundation (2013) Biomass for heat and power: Opportunities and economics [ONLINE]. Available at: **http://www.europeanclimate.org/documents/Biomass_report_-_Final.pdf**.

European Commission (2005) Reference Document on Best Available Techniques in the Slaughterhouses and Animal By-products Industries. Available at: eippcb.jrc.ec.europa.eu/reference/BREF/sa_bref_0505.pdf.

European Commission (2006a) Reference Document on Best Available Techniques in the Food, Drink and Milk Industries. Integrated Pollution Prevention and Control Bureau. Seville, Spain.

European Commission (2006b) Integrated Pollution Prevention and Control Reference Document on the Best Available Techniques for Waste Incineration.

European commission (2006c) Reference Document on Best Available Techniques in the Food, Drink and Milk Industries. Integrated Pollution Prevention and Control Bureau, Seville.

European Commission (2010a) Communication from the Commission to the Council and the European Parliament on future steps in bio-waste management in the European Union

European Commission (2010b) Communication from the Commission to the council and the European parliament on future steps in bio-waste managemetn in the European Union (COM(2010)235 Final. Luxembourg.

European Commission (2011) Commission Staff Working paper - 6th Commission Summary on the Implementation of the Urban Wastewater Treatment Directive.

European Commission (2012) Guidance on the interpretation of key provisions of Directive 2008/98/EC on waste. Brussels.

European Commission (2013a) Communication from the Commission to the European parliament, the council, the European Economic and Social Committee and the Committee of the Regions - Consultative Communication on the Sustainable Use of Phosphorus. Brussel.

European Commission (2013b) Report from the Commission to the council and the European Parliament on the implementation of Council Directive 91/676/EEC concerning the protection of waters against pollution caused by nitrates from agricultural sources based on Member State reports for the period 2008–2011.

European commission (2014) Best Available Techniques (BAT) Reference Document for Common Wastewater and Waste Gas Treatment/Management Systems in the Chemical Sector - final draft. European IPPC Bureau.

European Commission (2015a) The Water Framework Directive and the Floods Directive: Actions towards the 'good status' of EU water and to reduce flood risks - 4th implementation report on the Programmes of Measures (March 2015).

European Commission (2015b) Best Available Techniques (BAT) Reference Document for the Production of Pulp, Paper and Board. European IPPC Bureau.

European Commission (2016a) EU agricultural outlook - Prospect for the EU agricultural markets and income 2016-2026.

European Commission (2016b) Limiting the presence of total chromium in fertilising products. Working Document of the Fertilisers Working Group.

European Commission (2017a) Impact of long-term application of blast furnace and steel slags as liming materials on soil fertility, crop yields and plant health (SLAGFERTILISER). Final report - Study. Directorate-General for Research and Innovation, Brussels.

European Commission (2017b) Reference Document on Best Available Techniques on Waste Incineration - First Draft May 2017. Available at: eippcb.jrc.ec.europa.eu/reference/BREF/.

European Commission (2017c) Best Available Techniques (BAT) Reference Document in the Food, Drink and Milk Industries - First Draft (January 2017). European IPPC Bureau.

European Commission (2017d) Facts and Figures about Urban Wastewater Treatment. Available at: **http://ec.europa.eu/environment/water/water-urbanwaste/implementation/factsfigures_en.htm**.

European Environment Agency (2006) How much bioenergy can Europe produce without harming the environment?, EEA Report No 7/2006, European Environment Agency.

European Environment Agency (2012) Soil organic carbon. Indicator Assessment - Data and maps. Available at: **https://www.eea.europa.eu/data-and-maps/indicators/soil-organic-carbon-1/assessment**.

European Environment Agency (2013a) Urban wastewater treatment. Copenhagen.

European Environment Agency (2013b) EU bioenergy potential from a resource-efficiency perspective. EEA Report No 6/2013.

European Environment Agency (2016) Renewable energy in Europe 2016 - Recent growth and knock-on effects. EEA Report No 4/2016.

European Phosphate Fertilizer Alliance (2017) Phosphate fertilizers - Production. Available at: **http://aeep.eu/phosphate-fertilizers/production/** (consulted on 02/10/2017).

Eurostat (2016) Eurostat - your key to European statistics [ONLINE]. available at: **http://ec.europa.eu/eurostat/data/database**. Accesed 17 December 2016.

Ewert W., Hermanussen O., Kabbe C., Mele C., Niewersch C., Stossel E., . . . Stemann J. (2014) Comparison of sludge related processes. Sustainable sewage sludge management fostering phosphorus recovery and energy efficiency (P-REX report).

FAOSTAT-Commodity Balances Livestock (2012) FAOSTAT — Food Balance — Commodity

Balances — Livestock and Fish Primary Equivalent Data., Food and Agriculture Organization of the United Nations (FAO), Rome.

Feng Q.G., Lin Q.Y., Gong F.Z., Sugita S. & Shoya M. (2004) Adsorption of lead and mercury by rice husk ash. *Journal of Colloid and Interface Science* **278**: 1-8.

Ferrari S., Belevi H. & Baccini P. (2002) Chemical speciation of carbon in municipal solid waste incinerator residues. *Waste Management* **22**: 303-314.

Filella M., Belzile N. & Chen Y.-W. (2002a) Antimony in the environment: a review focused on natural waters: I. Occurrence. *Earth-Science Reviews* **57**: 125-176.

Filella M., Belzile N. & Chen Y.-W. (2002b) Antimony in the environment: a review focused on natural waters: II. Relevant solution chemistry. *Earth-Science Reviews* **59**: 265-285.

Flach B., Lieberz S., Rondon M., Williams B. & Teiken C. (2015) EU Biofuels Annual 2015. USDA Foreign Agricultural Service.

Flotats X., Bonmati A., Palatsi J. & Foget H.L. (2013) Trends on manure processing in Europe. Wastes: solutions, treatments and opportunities. 2nd International Conference, 11-13 September 2013, Braga, Portugal.

Flynn H.C., Meharg A.A., Bowyer P.K. & Paton G.I. (2003) Antimony bioavailability in mine soils. *Environmental Pollution* **124**: 93-100.

Foget H.L., Flotats X., Bonmati Blasi A., Palatsi J., Magri A. & Schelde K.M. (2011) Inventory of manure processing activities in Europe. Technical Report No. I concerning “Manure Processing Activities in Europe” to the European Commission, Directorate-General Environment. Brussels.

FOREGS (2005) Forum of the European Geological Survey Directors. Geochemical Atlas of Europe, Geological Survey of Finland, Espoo.

Formisani C. (2003) Trends in bulk blending world wide. Baltimore.

Fowler B. & Goering P. (1991) Antimony. *Metals and Their Compounds in the Environment: Occurrence, Analysis, and Biological Relevance,* Merian E. (ed.), pages. 743-750. VCH, Weinheim.

Franke-Whittle I.H. & Insam H. (2013) Treatment alternatives of slaughterhouse wastes, and their effect on the inactivation of different pathogens: A review. *Critical Reviews in Microbiology* **39**: 139-151.

Franz M. (2008) Phosphate fertilizer from sewage sludge ash (SSA). *Waste Management* **28**: 1809-1818.

Frattini P. (2005) Thallium properties and behaviour - a literature study. Etelä-Suomen yksikkö S41/0000/2005/2, GTK, Finland.

Freddo A., Cai C. & Reid B.J. (2012) Environmental contextualisation of potential toxic elements and polycyclic aromatic hydrocarbons in biochar. *Environmental Pollution* **171**: 18-24.

Freire M., Lopes H. & Tarelho L.A.C. (2015) Critical aspects of biomass ashes utilization in soils: Composition, leachability, PAH and PCDD/F. *Waste Management* **46**: 304-315.

Frieri M., Kumar K. & Boutin A. (2017) Antibiotic resistance. *Journal of Infection and Public Health* **10**: 369-378.

Fröschle B., Messelhäusser U., Höller C. & Lebuhn M. (2015) Fate of Clostridium botulinum and incidence of pathogenic clostridia in biogas processes. *Journal of Applied Microbiology* **119**: 936-947.

Gäbler H.E., Glüh K., Bahr A. & Utermann J. (2009) Quantification of vanadium adsorption by German soils. *Journal of Geochemical Exploration* **103**: 37-44.

Galinato S.P., Yoder J.K. & Granatstein D. (2011) The economic value of biochar in crop production and carbon sequestration. *Energy Policy* **39**: 6344-6350.

García C., T. R., Lynne B., Gali A., Rougé P. & Fabregras C. (2012) Struvite recovery: pilot-scale results and economic assessement of different scenarios. IWA World Water Congress and Exhibition, Busan, Korea.

Garg A.X., Hackam D. & Tonelli M. (2008) Systematic review and meta-analysis: when one study is just not enough. *Clinical Journal of the American Society of Nephrology* **3**: 253-260.

Gascó G., Paz-Ferreiro J. & Méndez A. (2012) Thermal analysis of soil amended with sewage sludge and biochar from sewage sludge pyrolysis. *Journal of Thermal Analysis and Calorimetry* **108**: 769-775.

Gaskin J.W., Steiner C., Harris K., Das K.C. & Bibens B. (2008) Effect of low-temperature pyrolysis conditions on biochar for agricultural use. *Transactions of the Asabe* **51**: 2061-2069.

Ge H., Zhang L., Batstone Damien J., Keller J. & Yuan Z. (2013) Impact of Iron Salt Dosage to Sewers on Downstream Anaerobic Sludge Digesters: Sulfide Control and Methane Production. *Journal of Environmental Engineering* **139**: 594-601.

Geertjes K., Baas K., Verschuren S., Kaashoek R. & Graveland C. (2016) Fosfor in afvalwater en slib. Centraal Bureau voor de Statistiek (NL), Projectnummer 14159.

Gell K., Ruijter F.J.d., Kuntke P., Graaff M.d. & Smit A.L. (2011) Safety and effectiveness of struvite from black water and urine as a phosphorus fertilizer. *Journal of Agricultural Science* **3**.

Gendebien A., Ferguson R., Brink J., Horth H., Sullivan M. & Davis M. (2001) Survey of wastes spread on land - final report.

Gerba C.P. (2015) Chapter 29 - Disinfection. *Environmental Microbiology (Third edition)*, pages. 645-662. Academic Press, San Diego.

Ghidotti M., Fabbri D. & Hornung A. (2017a) Profiles of Volatile Organic Compounds in Biochar: Insights into Process Conditions and Quality Assessment. *ACS Sustainable Chemistry & Engineering* **5**: 510-517.

Ghidotti M., Fabbri D., Mašek O., Mackay C.L., Montalti M. & Hornung A. (2017b) Source and Biological Response of Biochar Organic Compounds Released into Water; Relationships with Bio-Oil Composition and Carbonization Degree. *Environmental Science & Technology* **51**: 6580-6589.

Ghosh G.K., Mohan K.S. & Sarkar A.K. (1996) Characterization of soil-fertilizer P reaction products and their evaluation as sources of P for gram (Cicer arietinum L). *Nutrient Cycling in Agroecosystems* **46**: 71-79.

Godbold D.L., Fritz E. & Hüttermann A. (1988) Aluminum toxicity and forest decline. *Proceedings of the National Academy of Sciences* **85**: 3888-3892.

González-Ponce R., López-de-Sá E.G. & Plaza C. (2009) Lettuce Response to Phosphorus Fertilization with Struvite Recovered from Municipal Wastewater. *HortScience* **44**: 426-430.

Gonzalez Ponce R. & Garcia Lopez De Sa M.E. (2007) Evaluation of struvite as a fertilizer: A comparison with traditional P sources. *Agrochimica* **51**: 301-308.

Good A.G. & Beatty P.H. (2011) Fertilizing Nature: A Tragedy of Excess in the Commons. *Plos Biology* **9**.

Graber E.R., Meller Harel Y., Kolton M., Cytryn E., Silber A., Rav David D., . . . Elad Y. (2010a) Biochar impact on development and productivity of pepper and tomato grown in fertigated soilless media. *Plant and Soil* **337**: 481-496.

Graber E.R., Harel Y.M., Kolton M., Cytryn E., Silber A., David D.R., . . . Elad Y. (2010b) Biochar impact on development and productivity of pepper and tomato grown in fertigated soilless media. *Plant and Soil* **337**: 481-496.

Gratuito M.K.B., Panyathanmaporn T., Chumnanklang R.A., Sirinuntawittaya N. & Dutta A. (2008) Production of activated carbon from coconut shell: Optimization using response surface methodology. *Bioresource Technology* **99**: 4887-4895.

Griffiths B.S., Spilles A. & Bonkowski M. (2012) C:N:P stoichiometry and nutrient limitation of the soil microbial biomass in a grazed grassland site under experimental P limitation or excess. *Ecological Processes* **1**: 6.

Grösslová Z., Vaněk A., Mihaljevič M., Ettler V., Hojdová M., Zádorová T., . . . Ash C. (2015) Bioaccumulation of thallium in a neutral soil as affected by solid-phase association. *Journal of Geochemical Exploration* **159**: 208-212.

Gruda N. (2012) Current and Future Perspective of Growing Media in Europe. *V Balkan Symposium on Vegetables and Potatoes,* Vol. 960, Balliu A. & Gruda N. (eds.), pages. 37-43.

Gulyurtlu I., Crujeira A.T., Abelha P. & Cabrita I. (2007) Measurements of dioxin emissions during co-firing in a fluidised bed. *Fuel* **86**: 2090-2100.

Gunes A., Inal A., Sahin O., Taskin M.B., Atakol O. & Yilmaz N. (2015) Variations in mineral element concentrations of poultry manure biochar obtained at different pyrolysis temperatures, and their effects on crop growth and mineral nutrition. *Soil Use and Management* **31**: 429-437.

Günther L., Dockhorn T., Dichtl N., Müller J., Phan L.-C., Urban I., . . . Bayerle N. (2007) Technical and scientific monitoring of the large-scale seaborne technology at the WWTP Gifhorn. IWA Specialised Conference on Facing Sludge Diversities: Challenges, Risks and Opportunities, Antalya, Turkey.

Hach company (2017) The Cost Of Spilled Milk: Analyzing Product Loss In The Dairy Industry.

Haglund N. & Expertsgroup (2008) Guideline for classification of ash from solid biofuels and peat utilised for recycling and fertilizing in forestry and agriculture. 0283-7234, NT TECHN REPORT 613, Approved 2008-06, Norway.

Hamilton H.A., Brod E., Hanserud O.S., Gracey E.O., Vestrum M.I., Bøen A., . . . Brattebø H. (2016) Investigating Cross-Sectoral Synergies through Integrated Aquaculture, Fisheries, and Agriculture Phosphorus Assessments: A Case Study of Norway. *Journal of Industrial Ecology* **20**: 867-881.

Hammond J. & White P. (2005) Is struvite a valuable phosphate source for agriculture? Entrust project report 675382.006.

Hao X.-D., Wang C.-C., Lan L. & van Loosdrecht M.C.M. (2008) Struvite formation, analytical methods and effects of pH and Ca^{2 +}. *Water Science and Technology* **58**: 1687-1692.

Hao X., Wang C., van Loosdrecht M.C.M. & Hu Y. (2013) Looking Beyond Struvite for P-Recovery. *Environmental Science & Technology* **47**: 4965-4966.

Hapke H.-J. (1996) Heavy metal transfer in the food chain to humans. *Fertilizers and Environment: Proceedings of the International Symposium “Fertilizers and Environment”, held in Salamanca, Spain, 26–29, September, 1994*, Rodriguez-Barrueco C. (ed.), pages. 431-436. Springer Netherlands, Dordrecht.

Haraldsen T.K., Pedersen P.A. & Krogstad T. (2011) Mixtures of bottom wood ash and meat and bone meal as NPK fertilizer. *Recycling of Biomass Ashes*, Insam H. & Knapp B. (eds.), pages. 33-44. Springer-Verlag, Berlin.

Hass A., Gonzalez J.M., Lima I.M., Godwin H.W., Halvorson J.J. & Boyer D.G. (2012) Chicken Manure Biochar as Liming and Nutrient Source for Acid Appalachian Soil. *Journal of Environmental Quality* **41**: 1096-1106.

He Y.D., Zhai Y.B., Li C.T., Yang F., Chen L., Fan X.P., . . . Fu Z.M. (2010) The fate of Cu, Zn, Pb and Cd during the pyrolysis of sewage sludge at different temperatures. *Environmental Technology* **31**: 567-574.

Heijerick D. & Van Sprang P.A. (2004) Probabilistic Distribution of Boron in European Surface Waters. EURAS, Gent, Belgium. Report prepared for IAOIA (International Antimony Oxide Industry Association).

Heinrichs H. & Mayer R. (1977) Distribution and Cycling of Major and Trace Elements in Two Central European Forest Ecosystems1. *Journal of Environmental Quality* **6**: 402-407.

Hejcman M., Szaková J., Schellberg J., Šrek P. & Tlustoš P.J.N.C.i.A. (2009) The Rengen Grassland Experiment: soil contamination by trace elements after 65 years of Ca, N, P and K fertiliser application. **83**: 39-50.

Helsen L., Van den Bulck E., Van den Broeck K. & Vandecasteele C. (1997) Low-temperature pyrolysis of CCA-treated wood waste: Chemical determination and statistical analysis of metal input and output; mass balances. *Waste Management* **17**: 79-86.

Hemme A., Spark M., Wolf P., Paschertz H. & Kamphues J. (2005) Effects of different phosphorus sources in the diet on bone composition and stability (breaking strength) in broilers. *Journal of Animal Physiology and Animal Nutrition* **89**: 129-133.

Henrich E., Bürkle S., Meza-Renken Z.I. & Rumpel S. (1999) Combustion and gasification kinetics of pyrolysis chars from waste and biomass. *Journal of Analytical and Applied Pyrolysis* **49**: 221-241.

Hernandez A.B., Ferrasse J.-H., Chaurand P., Saveyn H., Borschneck D. & Roche N. (2011) Mineralogy and leachability of gasified sewage sludge solid residues. *Journal of Hazardous Materials* **191**: 219-227.

Herrman L. (2009) Recovery of phosphorus from wastewater treatment. A review. Umwelt Wissen Nr. 0929. Bundesamt für Umwelt (BAFU). Bern.

Herzel H., Kruger O., Hermann L. & Adam C. (2016) Sewage sludge ash - A promising secondary phosphorus source for fertilizer production. *Science of the Total Environment* **542**: 1136-1143.

Hess T., Aldaya M., Fawell J., Franceschini H., Ober E., Schaub R. & Schulze-Aurich J. (2014) Understanding the impact of crop and food production on the water environment - using sugar as a model. *Journal of the Science of Food and Agriculture* **94**: 2-8.

Hicks J. & Yager J. (2006) Airborne Crystalline Silica Concentrations at Coal-Fired Power Plants Associated with Coal Fly Ash. *Journal of Occupational and Environmental Hygiene* **3**: 448-455.

Hilber I., Bastos A.C., Loureiro S., Soja G., Marsz A., Cornelissen G. & Bucheli T.D. (2017) The different faces of biochar: contamination risk versus remediation tool. *Journal of Environmental Engineering and Landscape Management* **25**: 86-104.

Hilt K., Harrison J., Bowers K., Stevens R., Bary A. & Harrison K. (2016) Agronomic response of crops fertilized with struvite derived from dairy manure. *Water, Air, & Soil Pollution* **227**: 388.

Hinsinger P. (2001) Bioavailability of soil inorganic P in the rhizosphere as affected by root-induced chemical changes: a review. *Plant and Soil* **237**: 173-195.

Hjelmar O. (1990) Leachate from land disposal of coal fly ash. *Waste Management & Research* **8**: 429-449.

Hjerne O. & Hansson S. (2002) The role of fish and fisheries in Baltic Sea nutrient dynamics. *Limnology and Oceanography* **47**: 1023-1032.

Hjorth M., Christensen K.V., Christensen M.L. & Sommer S.G. (2010) Solid—liquid separation of animal slurry in theory and practice. A review. *Agronomy for Sustainable Development* **30**: 153-180.

Hoffman T.C., Zitomer D.H. & McNamara P.J. (2016) Pyrolysis of wastewater biosolids significantly reduces estrogenicity. *Journal of Hazardous Materials* **317**: 579-584.

Horowitz A.J. (1991) A Primer on Sediment-Trace Element Chemistry, 2nd. Edition. United States Geological Survey Open-File Report 91-76. Denver, Colorado.

Horrel R., Metherell A.K., Ford S. & Doscher C. (1999) Fertiliser evenness – losses and costs: A study on the economic benefits of uniform applications of fertiliser. *Proceedings of the New Zealand Grassland Association* **61**: 215-220.

Horswell J., Prosser J.A., Siggins A., van Schaik A., Ying L., Ross C., . . . Northcott G. (2014) Assessing the impacts of chemical cocktails on the soil ecosystem. *Soil Biology and Biochemistry* **75**: 64-72.

Hossain M.K., Strezov V., Yin Chan K. & Nelson P.F. (2010) Agronomic properties of wastewater sludge biochar and bioavailability of metals in production of cherry tomato (Lycopersicon esculentum). *Chemosphere* **78**: 1167-1171.

Hua K., Liu L. & Bureau D.P. (2005) Determination of Phosphorus Fractions in Animal Protein Ingredients. *Journal of Agricultural and Food Chemistry* **53**: 1571-1574.

Huang H., Liu J. & Jiang Y. (2015) Crystallization and precipitation of phosphate from swine wastewater by magnesium metal corrosion. *Scientific Reports* **5**: 16601.

Hultman B., Levlin E., Löwén M., Mossakowska A. & Stark K. (2001) Extraction of phosphorus and other products from sludge and ash. Interim report. Stockholm Vatten AB. Vattenvårdsteknik, KTH.

Hunsinger H., Jay K. & Vehlow J. (2002) Formation and destruction of PCDD/F inside a grate furnace. *Chemosphere* **46**: 1263-1272.

IAEA (2003) Assessing radiation doses to the public from radionuclides in timber and wood products. IAEA-TECDOC-1376. Vienna, Austria.

Ibrahim l.A.A. (2015) Chemical characterization and mobility of metal species in fly ash–water system. *Water Science* **29**: 109-122.

IFA (2013) Assessment of fertiliser use by crop at the global level 2010-2010/11.

Imtiaz M., Rizwan M.S., Xiong S., Li H., Ashraf M., Shahzad S.M., . . . Tu S. (2015) Vanadium, recent advancements and research prospects: A review. *Environment International* **80**: 79-88.

Injuk J., Van Grieken R. & De Leeuw G. (1998) Deposition of atmospheric trace elements into the North Sea: Coastal, ship, platform measurements and model predictions. *Atmospheric Environment* **32**: 3011-3025.

Inocre Biotech (2016) Inocre biotech - P-bac process [ONLINE], Available at: **http://fritzmeier-umwelttechnik.com/p-bac-process/?lang=en** [Accessed 20 May 2016].

Insam H. & Seewald M.S.A. (2010) Volatile organic compounds (VOCs) in soils. *Biology and Fertility of Soils* **46**: 199-213.

Insam H. & Knapp B. (2011) *Recycling of biomass ash.* Springer-Verlag, Berlin.

INTECUS GmbH (2013) Technical Guide on the Treatment and Recycling Techniques for Sludge from municipal Wastewater Treatment with references to Best Available Techniques (BAT). Federal Environment Ministry’s Advisory Assistance Programme for Environmental Protection in Central and Eastern Europe, the Caucasus and Central Asia. Project No. FKZ 380 01 233 Report No. (UBA-FB) 001927/E,1.

International Biochar Initiative (2016a) IBI (International Biochar Initiative). Standardized Product Definition and Product Testing Guidelines for Biochar That Is Used in Soil (IBI-STD-01.1, April 11, 2013) [ONLINE]. Available at: **http://www.biochar-international.org/sites/default/files/IBI_Biochar_Standards_V1.1.pdf** [Accessed 26 May 2016].

International Biochar Initiative (2016b) State of the biochar industry 2015 [ONLINE], Available at: **http://www.biochar-international.org/State_of_Industry_2015** [Accessed 26 May 2016].

International Manganese Institute (2012) Assessing the potential terrestrial risks from manganese.

Ippolito J.A., Spokas K.A., Novak J.M., Lentz R.D. & Cantrell K.B. (2015) Biochar elemental composition and factors influencing nutrient retention. *Biochar environmental management - science, technology and implementation*, Lehmann J. & Joseph S. (eds.), pages. 139-163. Routledge, Oxon.

Iqbal Y., Lewandowski I., Weinreich A., Wippel B., Pforte B., Hadai O., . . . Peters D. (2016) Maximising the yield of biomass from residues of agricultural crops and biomass from forestry. BIENL 15082, report for DG Energy.

IUPAC (1990) Glossary of atmospheric chemistry terms. International Union of Pure and Applied Chemistry, Applied Chemistry Division, Commission on Atmospheric Chemistry. *Pure and Applied Chemistry* **62**: 2167-2219.

Izquierdo M., Moreno N., Font O., Querol X., Alvarez E., Antenucci D., . . . Fernández-Pereira C. (2008) Influence of the co-firing on the leaching of trace pollutants from coal fly ash. *Fuel* **87**: 1958-1966.

Jaison S. & Muthukumar T. (2017) Chromium Accumulation in Medicinal Plants Growing Naturally on Tannery Contaminated and Non-contaminated Soils. *Biological Trace Element Research* **175**: 223-235.

Jala S. & Goyal D. (2006) Fly ash as a soil ameliorant for improving crop production—a review. *Bioresource Technology* **97**: 1136-1147.

James A., Thring R., Helle S. & Ghuman H. (2012) Ash Management Review - Applications of Biomass Bottom Ash. *Energies* **5**: 3856.

Janik L.J., Forrester S.T., Kirby J.K., McLaughlin M.J., Soriano-Disla J.M. & Reimann C. (2015a) Prediction of metal and metalloid partitioning coefficients (Kd) in soil using mid-infrared diffuse reflectance spectroscopy. *Chemistry of Europe's agricultural soils – Part B: General background information and further analysis of the GEMAS data set. Geologisches Jahrbuch (Reihe B 103),* Reimann C., Birke M., Demetriades A., Filzmoser P. & O'Connor P. (eds.), pages. 183-188. Schweizerbarth.

Janik L.J., Forrester S.T., Soriano-Disla J.M., Kirby J.K., McLaughlin M.J., Reimann C. & The G.P.T. (2015b) GEMAS: Prediction of solid-solution phase partitioning coefficients (Kd) for oxoanions and boric acid in soils using mid-infrared diffuse reflectance spectroscopy. *Environmental Toxicology and Chemistry* **34**: 235-246.

Janssen H. & Koopman R. (2005) Total phosphorus in soil, bio-waste and sewage sludge, prepared by CEN/BT/Task Force 151 – Horizontal Standards in the Field of Sludge, Bio-waste and Soil, the secretariat of which is held by Danish Standards.

Jeffery S., Abalos D., Spokas K.A. & Verheijen F.G.A. (2015) Biochar effects on crop yield. *Biochar environmental management - science, technology and implementation,* Lehmann J. & Joseph S. (eds.), pages. 301-326. Routledge, Oxon.

Jeffery S., Abalos D., Prodana M., Bastos A.C., van Groenigen J.W., Hungate B.A. & Verheijen F. (2017) Biochar boosts tropical but not temperate crop yields. *Environmental Research Letters* **12**: 053001.

Jensen A., Dam-Johansen K., Wójtowicz M.A. & Serio M.A. (1998) TG-FTIR Study of the Influence of Potassium Chloride on Wheat Straw Pyrolysis. *Energy & Fuels* **12**: 929-938.

Jewell S. & Kimball S. (2014) Mineral commodity summaries. US Geological Survey, pp. 83. Available at: **https://minerals.usgs.gov/minerals/pubs/mcs/2014/mcs2014.pdf**.

Jing C., Liu S., Korfiatis G.P. & Meng X. (2006) Leaching behavior of Cr(III) in stabilized/solidified soil. *Chemosphere* **64**: 379-385.

Johnson C.J., Pedersen J.A., Chappell R.J., McKenzie D. & Aiken J.M. (2007) Oral Transmissibility of Prion Disease Is Enhanced by Binding to Soil Particles. *PLoS Pathogens* **3**: e93.

Johnston A.E. & Steen I. (1999) Understanding Phsophorus and its use in agriculture. European Fertilizer Manufacturers' Association, Brussels.

Johnston A.E. & Richards I.R. (2003) Effectiveness of different precipitated phosphates as phosphorus sources for plants. *Soil Use and Management* **19**: 45-49.

Joint Research Centre (2009) Linking soil degradation processes, soil-friendly farming practices and soil-relevant policy measures. Ispra.

Jones A., Panagos P., Barcelo S. & al E. (2010) The State of Soil in Europe: A contribution of the JRC to the European Environment Agency’s Environment State and Outlook Report— SOER 2010. European Commission - Joint Research Centre, Institute for Environment and Sustainability, Ispra, Italy.

Jordan-Meille L., Rubæk G.H., Ehlert P.A.I., Genot V., Hofman G., Goulding K., . . . Barraclough P. (2012) An overview of fertilizer-P recommendations in Europe: soil testing, calibration and fertilizer recommendations. *Soil Use and Management* **28**: 419-435.

Jossa P. & Remy C. (2015) Life Cycle Assessment of selected processes for P recovery from sewage sludge, sludge liquor, or ash. Sustainable sewage sludge management fostering phosphorus recovery and energy efficiency (P-REX report). Berlin.

Jovic V. (1993) Thallium in rocks, soils and plants: past progress and future needs. . *Neues Jahrbuch für Mineralogie, Abhandlungen* **166**: 43-52.

Kabata-Pendias A. (2011) *Trace elements in soils and plants - Fourth edition*. CRC Press, Boca Raton.

Kabata-Pendias A. & Mukherjee A.B. (2007) *Trace elements from soil to human.* Springer, New York.

Kabbe C. (2017) Overview of phosphorus recovery from the wastewater stream facilities operating or under construction (March 2017) - P-REX Document.

Kabbe C., Kraus F. & Nattorp A. (2015) Integral guidance document for phosphorus recovery and recycling. Sustainable sewage sludge management fostering phosphorus recovery and energy efficiency (P-REX report).

Kahiluoto H., Kuisma M., Ketoja E., Salo T. & Heikkinen J. (2015) Phosphorus in Manure and Sewage Sludge More Recyclable than in Soluble Inorganic Fertilizer. *Environmental Science & Technology* **49**: 2115-2122.

Kalmykova Y. & Fedje K.K. (2013) Phosphorus recovery from municipal solid waste incineration fly ash. *Waste Management* **33**: 1403-1410.

Kalmykova Y. & Karlfeldt Fedje K. (2013) Phosphorus recovery from municipal solid waste incineration fly ash. *Waste Management* **33**: 1403-1410.

Kambo H.S. & Dutta A. (2015) A comparative review of biochar and hydrochar in terms of production, physico-chemical properties and applications. *Renewable and Sustainable Energy Reviews* **45**: 359-378.

Kaplan D.I., Sajwan K.S., Adriano D.C. & Gettier S. (1990) Phytoavailability and toxicity of beryllium and vanadium. *Water, Air, and Soil Pollution* **53**: 203-212.

Karbowska B. (2016) Presence of thallium in the environment: sources of contaminations, distribution and monitoring methods. *Environmental Monitoring and Assessment* **188**: 640.

Karltun E., Saarsalmi A., Ingerslev M., Mandre M., Andersson S., Gaitnieks T., . . . Varnagiryte-Kabasinskiene I. (2008) Wood ash recycling - possibilities and risks. *Sustainable use of forest biomass for energy: a synthesis with focus on the*, Röser D., Asikainen A., Raulund-Rasmussen K. & Stupak I. (eds.), pages. 79-108. Springer, Dordrecht.

Karvelas M., Katsoyiannis A. & Samara C. (2003) Occurrence and fate of heavy metals in the wastewater treatment process. *Chemosphere* **53**: 1201-1210.

Kasina M., Kowalski P.R. & Michalik M. (2016) Metals Accumulation During Thermal Processing of Sewage Sludge - Characterization of Fly Ash and Air Pollution Control (APC) Residues. *Energy Procedia* **97**: 23-30.

Kataki S., West H., Clarke M. & Baruah D.C. (2016) Phosphorus recovery as struvite: Recent concerns for use of seed, alternative Mg source, nitrogen conservation and fertilizer potential. *Resources, Conservation and Recycling* **107**: 142-156.

Katanda Y., Zvomuya F., Flaten D. & Cicek N. (2016) Hog-Manure-Recovered Struvite: Effects on Canola and Wheat Biomass Yield and Phosphorus Use Efficiencies. *Soil Science Society of America Journal* **80**: 135-146.

Kemacheevakul P., Otani S., Matsuda T. & Shimizu Y. (2012) Occurrence of micro-organic pollutants on phosphorus recovery from urine. *Water Science and Technology* **66**: 2194-2201.

Kercher A.K. & Nagle D.C. (2003) Microstructural evolution during charcoal carbonization by X-ray diffraction analysis. *Carbon* **41**: 15-27.

Kern J., Tammeorg P., Shanskiy M., Sakrabani R., Knicker H., Kammann C., . . . Glaser B. (2017) Synergistic use of peat and charred material in growing media – an option to reduce the pressure on peatlands? *Journal of Environmental Engineering and Landscape Management* **25**: 160-174.

Khan M.R., Wajid & Khan M. (1996) The effect of fly ash on plant growth and yield of tomato. *Environmental Pollution* **92**: 105-111.

Kim A.G., Kazonich G. & Dahlberg M. (2003) Relative Solubility of Cations in Class F Fly Ash. *Environmental Science & Technology* **37**: 4507-4511.

Kim S. & Dale B.E. (2004) Global potential bioethanol production from wasted crops and crop residues. *Biomass & Bioenergy* **26**: 361-375.

Kleber M., Hockaday W.C. & Nico P.S. (2015) Characteristics of biochar: macro-molecular properties. *Biochar environmental management - science, technology and implementation*, Lehmann J. & Joseph S. (eds.), pages. 11-138. Routledge, Oxon.

Kloss S., Zehetner F., Dellantonio A., Hamid R., Ottner F., Liedtke V., . . . Soja G. (2012) Characterization of Slow Pyrolysis Biochars: Effects of Feedstocks and Pyrolysis Temperature on Biochar Properties. *Journal of Environmental Quality* **41**: 990-1000.

Knowles O.A., Robinson B.H., Contangelo A. & Clucas L. (2011) Biochar for the mitigation of nitrate leaching from soil amended with biosolids. *Science of the Total Environment* **409**: 3206-3210.

Komiyama T., Kobayashi A. & Yahagi M. (2013) The chemical characteristics of ashes from cattle, swine and poultry manure. *Journal of Material Cycles and Waste Management* **15**: 106-110.

Kopittke P.M., Menzies N.W. & Blamey F.P.C. (2005) Rhizotoxicity of aluminate and polycationic aluminium at high pH. *Plant and Soil* **266**: 177-186.

Korving L. (2012) Trends in slibontwatering, STOWA Report 2012-46. p. 108.

Koutcheiko S., Monreal C.M., Kodama H., McCracken T. & Kotlyar L. (2007) Preparation and characterization of activated carbon derived from the thermo-chemical conversion of chicken manure. *Bioresource Technology* **98**: 2459-2464.

Kratz S., Schick J. & Schnug E. (2016) Trace elements in rock phosphates and P containing mineral and organo-mineral fertilizers sold in Germany. *Science of the Total Environment* **542**: 1013-1019.

Kraus F. & Seis W. (2015) Quantitative risk assessment of potential hazards for humans and the environment: quantification of potential hazards resulting from agricultural use of the manufactured fertilizers. Sustainable sewage sludge management fostering phosphorus recovery and energy efficiency (P-REX report). Berlin.

Kremer A.M. (2013) Nutrient budgets - Methodology and Handbook, version 1.02. Luxembourg.

Kretschmer B., Smith C., Watkins E., Allen B., Buckwell A., Desbarats J. & Kieve D. (2013) Technology options for feeding 10 billion people - Recycling agricultural, forestry & food wastes and residues for sustainable bioenergy and biomaterials. Institute for European Environmental Policy (IEEP).

Krewski D., Yokel R.A., Nieboer E., Borchelt D., Cohen J., Harry J., . . . Rondeau V. (2007) Human health risk assessment for aluminium, aluminium oxide and aluminium hydroxide. *Journal of toxicology and environmental health. Part B, Critical reviews* **10**: 1-269.

Kröppl M., Muñoz I.L. & Zeiner M. (2011) Trace elemental characterization of fly ash. *Toxicological & Environmental Chemistry* **93**: 886-894.

Krüger O. & Adam C. (2015) Recovery potential of German sewage sludge ash. *Waste Management* **45**: 400-406.

Krüger O., Grabner A. & Adam C. (2014) Complete Survey of German Sewage Sludge Ash. *Environmental Science & Technology* **48**: 11811-11818.

Kubota (2015) Sustainable phosphorus cycle by Kubota sludge melting system. European Sustainable Phosphorus Conference, Berlin.

Kuhn M., Spiegel H., Lopez F.A., Rex M. & Erdmann R. (2006) Sustainable agriculture using blast furnace and steel slags as liming agents. ECSC/RFCS Final Report for Contract No 7210-PR/267 for Directorate-General for Research, European Commission.

Kuligowski K., Poulsen T.G., Rubæk G.H. & Sørensen P. (2010) Plant-availability to barley of phosphorus in ash from thermally treated animal manure in comparison to other manure based materials and commercial fertilizer. *European Journal of Agronomy* **33**: 293-303.

Kuokkanen T., Pöykiö R., Nurmesniemi H. & Rämö J. (2006) Sequential leaching of heavy metals and sulfur in bottom ash and fly ash from the co-combustion of wood and peat at a municipal district heating plant. *Chemical Speciation & Bioavailability* **18**: 131-142.

Kuppusamy S., Thavamani P., Megharaj M., Venkateswarlu K. & Naidu R. (2016) Agronomic and remedial benefits and risks of applying biochar to soil: Current knowledge and future research directions. *Environment International* **87**: 1-12.

Kuzyakov Y. & Domanski G. (2000) Carbon input by plants into the soil. Review. *Journal of Plant Nutrition and Soil Science* **163**: 421-431.

Kyllonen K., Karlsson V. & Ruoho-Airola T. (2009) Trace element deposition and trends during a ten year period in Finland. *Science of the Total Environment* **407**: 2260-2269.

Lal R. (2009) Soils and food sufficiency. A review. *Agronomy for Sustainable Development* **29**: 113-133.

Langenkamp H. & Part P. (2001) Organic contaminants in sewage sludge for agricultural use. European Commision, Joint Research Centre, Institute for Environment and Substainability, Soil and Waste Unit.

Langeveld C.P. & Ten Wolde K.W. (2013) Phosphate recycling in mineral fertiliser production. *Proceedings of the International Fertiliser Society* **727**: 1-24.

Lanzerstorfer C. (2015) Investigation of the contamination of a fly ash sample during sample preparation by air classification. *International Journal of Environmental Science and Technology* **12**: 1437-1442.

Lapa N., Barbosa R., Lopes M.H., Mendes B., Abelha P., Gulyurtlu I. & Santos Oliveira J. (2007) Chemical and ecotoxicological characterization of ashes obtained from sewage sludge combustion in a fluidised-bed reactor. *Journal of Hazardous Materials* **147**: 175-183.

Lapuerta M., Hernandez J.J., Pazo A. & Lopez J. (2008) Gasification and co-gasification of biomass wastes: Effect of the biomass origin and the gasifier operating conditions. *Fuel Processing Technology* **89**: 828-837.

Larsson M.A., Baken S., Gustafsson J.P., Hadialhejazi G. & Smolders E. (2013) Vanadium bioavailability and toxicity to soil microorganisms and plants. *Environmental Toxicology and Chemistry* **32**: 2266-2273.

Larsson M.A., Baken S., Smolders E., Cubadda F. & Gustafsson J.P. (2015a) Vanadium bioavailability in soils amended with blast furnace slag. *Journal of Hazardous Materials* **296**: 158-165.

Larsson M.A., D'Amato M., Cubadda F., Raggi A., Öborn I., Kleja D.B. & Gustafsson J.P. (2015b) Long-term fate and transformations of vanadium in a pine forest soil with added converter lime. *Geoderma* **259-260**: 271-278.

Lavric E.D., Konnov A.A. & De Ruyck J. (2004) Dioxin levels in wood combustion - a review. *Biomass & Bioenergy* **26**: 115-145.

Le Corre K.S., Valsami-Jones E., Hobbs P. & Parsons S.A. (2009) Phosphorus Recovery from Wastewater by Struvite Crystallization: A Review. *Critical Reviews in Environmental Science and Technology* **39**: 433-477.

Lehmann J. & Joseph S. (2015) *Biochar for environmental management - science, technology and implementation, 2nd edition*. Routledge, New York.

Lehmann J., Abiven S., Kleber M., Pan G., Singh B.P., Sohi S.P. & Zimmerman A.R. (2015) Persistence of bichar in soil. *Biochar for Environmental Management - Science, Technology and Implementation, 2nd edition.*, Lehmann J. & Joseph S. (eds.), pages. 235-282. Routledge - Earthscan, London.

Leip A., Weiss F., Lesschen J.P. & Westhoek H. (2014) The nitrogen footprint of food products in the European Union. *Journal of Agricultural Science* **152**: S20-S33.

Lemieux P.M., Lee C.W., Ryan J.V. & Lutes C.C. (2001) Bench-scale studies on the simultaneous formation of PCBs and PCDD/Fs from combustion systems. *Waste Management* **21**: 419-425.

Li B., Zhang H., Ma Y. & McLaughlin M.J. (2011) Influences of soil properties and leaching on nickel toxicity to barley root elongation. *Ecotoxicology and Environmental Safety* **74**: 459-466.

Li F., Cao X., Zhao L., Wang J. & Ding Z. (2014) Effects of Mineral Additives on Biochar Formation: Carbon Retention, Stability, and Properties. *Environmental Science & Technology* **48**: 11211-11217.

Li J., Liu B., Zeng Y. & Wang Z. (2017a) Mineralogical determination and geo-chemical modeling of chromium release from AOD slag: Distribution and leachability aspects. *Chemosphere* **167**: 360-366.

Li J., Liu B., Zeng Y., Wang Z. & Gao Z. (2017b) Maximum availability and mineralogical control of chromium released from AOD slag. *Environmental Monitoring and Assessment* **189**: 113.

Li M., Wang C., Cen K., Ni M. & Li X. (2018) Emission characteristics and vapour/particulate phase distributions of PCDD/F in a hazardous waste incinerator under transient conditions. *Royal Society Open Science* **5**.

Li Y.-x., Zhang X.-l., Li W., Lu X.-f., Liu B. & Wang J. (2013) The residues and environmental risks of multiple veterinary antibiotics in animal faeces. *Environmental Monitoring and Assessment* **185**: 2211-2220.

Liang X., Jin Y., He M., Niyungeko C., Zhang J., Liu C., . . . Arai Y. (2017) Phosphorus speciation and release kinetics of swine manure biochar under various pyrolysis temperatures. *Environmental Science and Pollution Research*.

Libra J.A., Ro K.S., Kammann C., Funke A., Berge N.D., Neubauer Y., . . . Emmerich K.-H. (2011) Hydrothermal carbonization of biomass residuals: a comparative review of the chemistry, processes and applications of wet and dry pyrolysis. *Biofuels* **2**: 71-106.

Lidén G. (2006) Dustiness Testing of Materials Handled at Workplaces. *The Annals of Occupational Hygiene* **50**: 437-439.

Lievens C., Carleer R., Cornelissen T. & Yperman J. (2009) Fast pyrolysis of heavy metal contaminated willow: Influence of the plant part. *Fuel* **88**: 1417-1425.

Lillo-Ródenas M.A., Marco-Lozar J.P., Cazorla-Amorós D. & Linares-Solano A. (2007) Activated carbons prepared by pyrolysis of mixtures of carbon precursor/alkaline hydroxide. *Journal of Analytical and Applied Pyrolysis* **80**: 166-174.

Lin Y., Munroe P., Joseph S., Henderson R. & Ziolkowski A. (2012) Water extractable organic carbon in untreated and chemical treated biochars. *Chemosphere* **87**: 151-157.

Lindsay W.L. & De Ment J.D. (1961) Effectiveness of some iron phosphates as sources of phosphorus for plants. *Plant and Soil* **14**: 118-126.

Liu H., Pu C., Yu X., Sun Y. & Chen J. (2018) Removal of tetracyclines, sulfonamides, and quinolones by industrial-scale composting and anaerobic digestion processes. *Environmental Science and Pollution Research*.

Liu X., Li Z., Zhang Y., Feng R. & Mahmood I.B. (2014) Characterization of human manure-derived biochar and energy-balance analysis of slow pyrolysis process. *Waste Management* **34**: 1619-1626.

Liu X., Tao Y., Wen G., Kong F., Zhang X. & Hu Z. (2016) Influence of Soil and Irrigation Water pH on the Availability of Phosphorus in Struvite Derived from Urine through a Greenhouse Pot Experiment. *Journal of Agricultural and Food Chemistry* **64**: 3324-3329.

Liu Y., Rahman M.M., Kwag J.H., Kim J.H. & Ra C. (2011) Eco-friendly Production of Maize Using Struvite Recovered from Swine Wastewater as a Sustainable Fertilizer Source. *Asian-Australasian Journal of Animal Sciences* **24**: 1699-1705.

Lizarraga E., Zabaleta C., Palop J.A.J.J.o.T.A. & Calorimetry (2007) Thermal stability and decomposition of pharmaceutical compounds. **89**: 783-792.

Locke J.C., Altland J.E. & Ford C. (2013) Gasified rice hull biochar affects nutrition and growth of horticultural crops in container substrates. *Journal of Horticultural Science* **31**: 195-202.

Looft T., Johnson T.A., Allen H.K., Bayles D.O., Alt D.P., Stedtfeld R.D., . . . Stanton T.B. (2012) In-feed antibiotic effects on the swine intestinal microbiome. *Proceedings of the National Academy of Sciences* **109**: 1691-1696.

Lopes H., Gulyurtlu I., Abelha P., Crujeira T., Salema D., Freire M., . . . Cabrita I. (2009) Particulate and PCDD/F emissions from coal co-firing with solid biofuels in a bubbling fluidised bed reactor. *Fuel* **88**: 2373-2384.

Lopez Anton M.A., Spears D.A., Diaz Somoano M. & Martinez Tarazona M.R. (2013) Thallium in coal: Analysis and environmental implications. *Fuel* **105**: 13-18.

Lou Y., Ye Z.-L., Chen S., Wei Q., Zhang J. & Ye X. (2018) Influences of dissolved organic matters on tetracyclines transport in the process of struvite recovery from swine wastewater. *Water Research* **134**: 311-326.

Lövren L. (2012) Roll pelletizing of ash—Cost efficient handling and improved product with accelerated carbonatization. In Procedings of Conference on Ash Utilization 2012: Ashes in a Sustainable Society, Stockholm, Sweden, 25–27.

Lu T., Yuan H., Wang Y., Huang H. & Chen Y. (2016) Characteristic of heavy metals in biochar derived from sewage sludge. *Journal of Material Cycles and Waste Management* **18**: 725-733.

Lukehurst C.T., Frost P. & Al Seadi T. (2010) Utilisation of digestate from biogas plants as biofertiliser. IEA Bioenergy Report - Task 37.

Lundin L. & Marklund S. (2005) Thermal Degradation of PCDD/F in Municipal Solid Waste Ashes in Sealed Glass Ampules. *Environmental Science & Technology* **39**: 3872-3877.

Lundin M., Olofsson M., Pettersson G.J. & Zetterlind H. (2004) Environmental and economic assessment of sewage sludge handling options. *Resources, Conservation and Recycling* **41**: 255-278.

Luo Y., Xu L., Rysz M., Wang Y., Zhang H. & Alvarez P.J.J. (2011) Occurrence and Transport of Tetracycline, Sulfonamide, Quinolone, and Macrolide Antibiotics in the Haihe River Basin, China. *Environmental Science & Technology* **45**: 1827-1833.

Lynch D., Low F., Henihan A.M., Garcia A., Kwapinski W., Zhang L. & Leahy J.J. (2014) Behavior of Heavy Metals during Fluidized Bed Combustion of Poultry Litter. *Energy & Fuels* **28**: 5158-5166.

Ma Y., Oliveira R.S., Freitas H. & Zhang C. (2016) Biochemical and Molecular Mechanisms of Plant-Microbe-Metal Interactions: Relevance for Phytoremediation. *Frontiers in Plant Science* **7**: 918.

Ma Y.L. & Matsunaka T. (2013) Biochar derived from dairy cattle carcasses as an alternative source of phosphorus and amendment for soil acidity. *Soil Science and Plant Nutrition* **59**: 628-641.

Macdonald G.K., Jarvie H.P., Withers P.J.A., Doody D.G., Keeler B.L., Haygarth P.M., . . . Zhang T. (2016) Guiding phosphorus stewardship for multiple ecosystem services. *Ecosystem Health and Sustainability* **2**: e01251.

Madhiyanon T., Sathitruangsak P. & Soponronnarit S. (2009) Co-combustion of rice husk with coal in a cyclonic fluidized-bed combustor (Psi-FBC). *Fuel* **88**: 132-138.

Maestrini B., Nannipieri P. & Abiven S. (2015) A meta-analysis on pyrogenic organic matter induced priming effect. *GCB Bioenergy* **7**: 577-590.

Mahmoud A., Olivier J., Vaxelaire J. & Hoadley A.F.A. (2011) Electro-dewatering of wastewater sludge: Influence of the operating conditions and their interactions effects. *Water Research* **45**: 2795-2810.

Makridis C. & Amberger A. (1996) Thallium concentration in soils and crops and critical values with respect to food chain. *Fertilizers and Environment: Proceedings of the International Symposium “Fertilizers and Environment”, held in Salamanca, Spain, 26–29, September, 1994,* Rodriguez-Barrueco C. (ed.), pages. 443-448. Springer Netherlands, Dordrecht.

Mankasingh U., Choi P.-C. & Ragnarsdottir V. (2011) Biochar application in a tropical, agricultural region: A plot scale study in Tamil Nadu, India. *Applied Geochemistry* **26, Supplement**: S218-S221.

Manteau U. (2012) Wood Flows in Europe. Commissioned by CEPI: Confederation of European Paper Industries, and CEI-Bois: European Confederation of Woodworking Industries [ONLINE]. Available at: **http://www.cepi.org/system/files/public/documents/publications/forest/2012/CEPIWoodFlowsinEurope2012.pdf**.

Marchi A., Geerts S., Weemaes M., Wim S. & Christine V. (2015) Full-scale phosphorus recovery from digested wastewater sludge in Belgium - part I: technical achievements and challenges. *Water Science and Technology* **71**: 487-494.

Marculescu C. & Stan C. (2012) Non-oxidant Thermal Treatment for Organic Waste Neutralization. *Energy Procedia* **18**: 545-551.

Marti N., Ferrer J., Seco A. & Bouzas A. (2008) Optimisation of sludge line management to enhance phosphorus recovery in WWTP. *Water Research* **42**: 4609-4618.

Martí N., Pastor L., Bouzas A., Ferrer J. & Seco A. (2010) Phosphorus recovery by struvite crystallization in WWTPs: Influence of the sludge treatment line operation. *Water Research* **44**: 2371-2379.

Martin H.W., Kaplan D.I.J.W., Air, & Pollution S. (1998) Temporal Changes in Cadmium, Thallium, and Vanadium Mobility in Soil and Phytoavailability under Field Conditions. **101**: 399-410.

Martinez-Sanchez V., Kromann M.A. & Astrup T.F. (2015) Life cycle costing of waste management systems: Overview, calculation principles and case studies. *Waste Management* **36**: 343-355.

Martins Abdao dos Passos A., Milanez de Rezende P., Reis Carvalho E. & William de Avila F. (2015) Biochar, farmyard manure and poultry litter on chemical attributes of a Distrophic Cambissol and soybean crop. *Agrária - Revista Brasileira de Ciências Agrárias* **3**: 382-388.

Masia A.A.T., Buhre B.J.P., Gupta R.P. & Wall T.F. (2007) Characterising ash of biomass and waste. *Fuel Processing Technology* **88**: 1071-1081.

Massé D.I., Cata Saady N.M. & Gilbert Y. (2014) Potential of Biological Processes to Eliminate Antibiotics in Livestock Manure: An Overview. *Animals : an Open Access Journal from MDPI* **4**: 146-163.

Massey M.S., Davis J.G., Ippolito J.A. & Sheffield R.E. (2009) Effectiveness of recovered magnesium phosphates as fertilizers in neutral and slightly alkaline soils. *Agronomy Journal* **101**: 323-329.

Masto R.E., Sarkar E., George J., Jyoti K., Dutta P. & Ram L.C. (2015) PAHs and potentially toxic elements in the fly ash and bed ash of biomass fired power plants. *Fuel Processing Technology* **132**: 139-152.

Mätzing H., Baumann W., Becker B., Jay K., Paur H.-R. & Seifert H. (2001) Adsorption of PCDD/F on MWI fly ash. *Chemosphere* **42**: 803-809.

Mayer B.K., Baker L.A., Boyer T.H., Drechsel P., Gifford M., Hanjra M.A., . . . Rittmann B.E. (2016) Total Value of Phosphorus Recovery. *Environmental Science & Technology* **50**: 6606-6620.

McLaren R.G., Naidu R., Smith J. & Tiller K.G. (1998) Fractionation and Distribution of Arsenic in Soils Contaminated by Cattle Dip. *Journal of Environmental Quality* **27**: 348-354.

McLaughlin M.J. (2002) Measuring P availability in soils fertilized with water-soluble P fertilizers using ^{32}P methodologies. IAEA Tecdoc 1272. Vienna.

Mehta C.M., Khunjar W.O., Nguyen V., Tait S. & Batstone D.J. (2015) Technologies to recover nutrients from waste streams: a critical review. *Critical Reviews in Environmental Science and Technology* **45**: 385-427.

Meij R., Nagengast S. & te Winkel H. (2000) The Occurrence of Quartz in Coal Fly Ash Particles. *Inhalation Toxicology* **12**: 109-116.

Méndez A., Gómez A., Paz-Ferreiro J. & Gascó G. (2012) Effects of sewage sludge biochar on plant metal availability after application to a Mediterranean soil. *Chemosphere* **89**: 1354-1359.

Merchant S.S. (2010) The Elements of Plant Micronutrients. *Plant Physiology* **154**: 512-515.

Meyer S., Genesio L., Vogel I., Schmidt H.-P., Soja G., Someus E., . . . Glaser B. (2017) Biochar standardization and legislation harmonization. *Journal of Environmental Engineering and Landscape Management* 1-17.

Michalowski T. & Pietrzyk A. (2006) A thermodynamic study of struvite plus water system. *Talanta* **68**: 594-601.

Midwest Research Institute (1998) Emission Factor Documentation for AP-42, Section 9.2.1, Fertilizer Application, Draft Report, Midwest Research Institute (MRI) for the Office of Air Quality Planning and Standards (OAQPS), U. S. Environmental Protection Agency (EPA), Contract No. 68-D2-0159.

Miles T.R., Baxter L.L., Bryers R.W., Jenkins B.M. & Oden L.L. (1996) Boiler deposits from firing biomass fuels. *Biomass & Bioenergy* **10**: 125-138.

Miles T.R., Miles J.T.R., Baxter L.L., Bryers R.W., Jenkins B.M. & Oden L.L. (1995) Alkali deposits found in biomass power plants. A prelimimary investigation of their extent and nature. Report of the National Renewable Energy Laboratory (NREL/TZ-2-11226-1; TP-433-8142). Golden, CO, USA.

Milieu Ltd - WRc - RPA (2010a) Environmental, economic and social impacts of the use of sewage sludge on land - final report, Study Contract DG ENV.G.4/ETU/2008/0076r (for the European Commission, DG Environment).

Milieu Ltd - WRc - RPA (2010b) Environmental, economic and social impacts of the use of sewage sludge on land - Final report. Part I - Overview Report. Report report prepared for the European Commission, DG Environment under Study Contract DG ENV.G.4/ETU/2008/0076r.

Milieu Ltd - WRc - RPA (2010c) Environmental, economic and social impacts of the use of sewage sludge on land - Final report. Part II - Report on Options and Impacts. Report report prepared for the European Commission, DG Environment under Study Contract DG ENV.G.4/ETU/2008/0076r.

Milosavljevic I., Oja V. & Suuberg E.M. (1996) Thermal effects in cellulose pyrolysis: Relationship to char formation processes. *Industrial & Engineering Chemistry Research* **35**: 653-662.

Miravet R., López-Sánchez J.F. & Rubio R. (2006) Leachability and analytical speciation of antimony in coal fly ash. *Analytica Chimica Acta* **576**: 200-206.

Misra M.K., Ragland K.W. & Baker A.J. (1993) WOOD ASH COMPOSITION AS A FUNCTION OF FURNACE TEMPERATURE. *Biomass & Bioenergy* **4**: 103-116.

Mithöfer A., Schulze B. & Boland W. (2004) Biotic and heavy metal stress response in plants: evidence for common signals. *FEBS Letters* **566**: 1-5.

Moilanen A. (2006) Thermogravimetric characterisations of biomass and waste for gasification processes. VTT Technical Research Centre of Finland. Publications No 607; p. 103.

Mok W.S.L. & Antal M.J. (1983) Effects of pressure on biomass pyrolysis. II. Heats of reaction of cellulose pyrolysis. *Thermochimica Acta* **68**: 165-186.

Moller K. (2015) Assessment of alternative phosphorus fertilisers for organic farming - meat and bone meal. Stuttgart.

Moller K. (2016) Assessment of alternative phosphorus fertilisers for organic farming - chars, ashes and slags. Stuttgart.

Möller K. & Müller T. (2012) Effects of anaerobic digestion on digestate nutrient availability and crop growth: A review. *Engineering in Life Sciences* **12**: 242-257.

Moloney A., Fotoushi J. & Steiger J. (2014) Industrial water use [ONLINE], available at: **http://www.wwdmag.com/industrial-wastewater-recyclingreuse/industrial-revolution**.

Montag D. & Pinnekamp J. (2009) The PASH process for P-recovery and overview of the German Funding Programme "Recycling management of plant nutrients, especially phosphorous". BALTIC 21 Phosphorus Recycling and Good Agricultural Management Practice, Berlin.

Monte M.C., Fuente E., Blanco A. & Negro C. (2009) Waste management from pulp and paper production in the European Union. *Waste Management* **29**: 293-308.

Montes N., Otero M., Coimbra R.N., Méndez R. & Martín-Villacorta J. (2015) Removal of tetracyclines from swine manure at full-scale activated sludge treatment plants. *Environmental Technology* **36**: 1966-1973.

Moore C. (2013) Can chicken litter be economically utilised at farm scale for energy production and improve the profitability of the chicken farmer by reducing costs and increasing revenues? Master thesis, DCU.

Mooshammer M., Wanek W., Zechmeister-Boltenstern S. & Richter A. (2014) Stoichiometric imbalances between terrestrial decomposer communities and their resources: mechanisms and implications of microbial adaptations to their resources. *Frontiers in Microbiology* **5**.

Morabito E., Contini D., Belosi F., Stortini A.M., Manodori L. & Gambaro A. (2014) Atmospheric Deposition of Inorganic Elements and Organic Compounds at the Inlets of the Venice Lagoon. *Advances in Meteorology* **2014**: 10.

Morselli L., Brusori B., Passarini F., Bernardi E., Francaviglia R., Gataleta L., . . . Olivieri P. (2004) Heavy metals monitoring at a Mediterranean natural ecosystem of Central Italy. Trends in different environmental matrixes. *Environment International* **30**: 173-181.

Morshedizad M., Panten K., Klysubun W. & Leinweber P. (2018) Bone char effects on soil: sequential fractionations and XANES spectroscopy. *SOIL* **4**: 23-35.

Müller-Stöver D., Ahrenfeldt J., Holm J.K., Shalatet S.G.S., Henriksen U. & Hauggaard-Nielsen H. (2012) Soil application of ash produced by low-temperature fluidized bed gasification: effects on soil nutrient dynamics and crop response. *Nutrient Cycling in Agroecosystems* **94**: 193-207.

Muller J., Reinhardt M., Gunther L., Dockhorn T., Dichtl N., Urban I., . . . Vesterager N.O. (2005) Fundamentals and feasibility of nutrient recycling using the seaborne process. Conference on the Management of Residues Emanating from Water and Wastewater Treatment, Johannesburg, South Africa.

Münch E.V. & Barr K. (2001) Controlled struvite crystallisation for removing phosphorus from anaerobic digester sidestreams. *Water Research* **35**: 151-159.

Nanzer S., Oberson A., Berger L., Berset E., Hermann L. & Frossard E. (2014) The plant availability of phosphorus from thermo-chemically treated sewage sludge ashes as studied by 33P labeling techniques. *Plant and Soil* **377**: 439-456.

Nattorp A., Remmen K. & Remy C. (2017) Cost assessment of different routes for phosphorus recovery from wastewater using data from pilot and production plants. *Water Science and Technology* **76**: 413-424.

Neal C., Williams R.J., Bowes M.J., Harrass M.C., Neal M., Rowland P., . . . Jarvie H.P. (2010) Decreasing boron concentrations in UK rivers: Insights into reductions in detergent formulations since the 1990s and within-catchment storage issues. *Science of the Total Environment* **408**: 1374-1385.

Nelissen V., Rutting T., Huygens D., Staelens J., Ruysschaert G. & Boeckx P. (2012) Maize biochars accelerate short-term soil nitrogen dynamics in a loamy sand soil. *Soil Biology & Biochemistry* **55**: 20-27.

Nelson N.O. & Janke R.R. (2007) Phosphorus sources and management in organic production systems. *Horttechnology* **17**: 442-454.

Neupane G. & Donahoe R.J. (2013) Leachability of elements in alkaline and acidic coal fly ash samples during batch and column leaching tests. *Fuel* **104**: 758-770.

Neves D., Thunman H., Matos A., Tarelho L. & Gómez-Barea A. (2011) Characterization and prediction of biomass pyrolysis products. *Progress in Energy and Combustion Science* **37**: 611-630.

Neyroud J.A. & Lischer P. (2003) Do different methods used to estimate soil phosphorus availability across Europe give comparable results? *Journal of Plant Nutrition and Soil Science-Zeitschrift Fur Pflanzenernahrung Und Bodenkunde* **166**: 422-431.

Nielsen F.H. (2014) Update on human health effects of boron. *Journal of Trace Elements in Medicine and Biology* **28**: 383-387.

Nieminen J. (2010) Phosphorus recovery and recycling from municipal wastewater sludge. Thesis, Aalto University.

NIOSH—Publications Dissemination (2002) Hazard Review: Health Effects of Occupational Exposure to Respirable Crystalline Silica. National Institutes of Occupational Safety and Health (NIOSH); Cincinnati, OH, USA.

Novak J.M., Lima I., Xing B., Gaskin J.W., Steiner C., Das K.C., . . . Schomberg H. (2009) Characterization of Designer Biochar Produced at Different Temperatures and Their Effects on a Loamy Sand. *2009* **3**.

Nriagu J.O. (1998) History, production and uses of thallium. In J. O. Nriagu (Ed.), Thallium in the environment 29 (pp. 1–14). New York: Wiley.

Nutalapati D., Gupta R., Moghtaderi B. & Wall T.F. (2007) Assessing slagging and fouling during biomass combustion: A thermodynamic approach allowing for alkali/ash reactions. *Fuel Processing Technology* **88**: 1044-1052.

Obernberger I. & Supancic K. (2009) Possibilities of ash utilisation from biomass combustion plants. Proceedings of teh 17th European Biomass Conference & exhibition, 29 June - 03 July Hamburg, Germany.

Ochecova P., Tlustos P. & Szakova J. (2014) Wheat and Soil Response to Wood Fly Ash Application in Contaminated Soils. *Agronomy Journal* **106**: 995-1002.

Oenema O., Chardon W., Ehlert P.A.I., Van Dijk K.C., Schoumans O. & Rulkens W.H. (2012) Phosphorus fertilisers from by-products and wastes. Proceedings 717, International Fertiliser Society, Leek, UK.

Oorts K., Ghesquiere U. & Smolders E. (2007) Leaching and aging decrease nickel toxicity to soil microbial processes in soils freshly spiked with nickel chloride. **26**: 1130-1138.

Oosterhuis F.H., Brouwer F.M. & Wijnants H.J. (2000) A possible EU wide charge on cadmium in phosphate fertilisers: Economic and environmental implications. Institute for Environmental Studies, Report number E-00/02 commissioned by the European Commission. Amsterdam.

Page A.L., Elseewi A.A. & Straughan I.R. (1979) Physical and chemical properties of fly ash from coal-fired power plants with reference to environmental impacts. *Residue Reviews: Residues of Pesticides and Other Contaminants in the Total Environment*, Gunther F.A. & Gunther J.D. (eds.), pages. 83-120. Springer New York, New York, NY.

Pagenkopf G.K. & Connolly J.M. (1982) Retention of boron by coal ash. *Environmental Science & Technology* **16**: 609-613.

Paisley L.G. & Hostrup-Pedersen J. (2005a) A quantitative assessment of the BSE risk associated with fly ash and slag from the incineration of meat-and-bone meal in a gas-fired power plant in Denmark. *Preventive Veterinary Medicine* **68**: 263-275.

Paisley L.G. & Hostrup-Pedersen J. (2005b) A quantitative assessment of the BSE risk associated with fly ash and slag from the incineration of meat-and-bone meal in a gas-fired power plant in Denmark. *Preventive Veterinary Medicine* **68**: 263-275.

Pan Y.P. & Wang Y.S. (2015) Atmospheric wet and dry deposition of trace elements at 10 sites in Northern China. *Atmospheric Chemistry and Physics* **15**: 951-972.

Panagos P., Van Liedekerke M., Jones A. & Montanarella L. (2012) European Soil Data Centre: Response to European policy support and public data requirements. *Land use policy* **29**: 329-338.

Pandelova M., Lenoir D. & Schramm K.W. (2006) Correlation between PCDD/F, PCB and PCBz in coal/waste combustion. Influence of various inhibitors. *Chemosphere* **62**: 1196-1205.

Pandey V.C. & Singh N. (2010) Impact of fly ash incorporation in soil systems. *Agriculture, Ecosystems & Environment* **136**: 16-27.

Parker D.L., Kumar H.D., Rai L.C. & Singh J.B. (1997) Potassium salts inhibit growth of the cyanobacteria Microcystis spp. in pond water and defined media: Implications for control of microcystin-producing aquatic blooms. *Applied and Environmental Microbiology* **63**: 2324-2329.

Paul E., Laval M.L. & Sperandio M. (2001) Excess Sludge Production and Costs Due to Phosphorus Removal. *Environmental Technology* **22**: 1363-1371.

Pell Frischmann Consultants Ltd (2012) Enhancement and treatment of digestates from anaerobic digestion, WRAP Project code: OMK006 - 002, UK.

Peng X., Deng Y., Peng Y. & Yue K. (2018) Effects of biochar addition on toxic element concentrations in plants: A meta-analysis. *Science of the Total Environment* **616-617**: 970-977.

Pensis I., Luetzenkirchen F. & Friede B. (2014) SWeRF—A Method for Estimating the Relevant Fine Particle Fraction in Bulk Materials for Classification and Labelling Purposes. *Annals of Occupational Hygiene* **58**: 501-511.

Peralta-Videa J.R., Lopez M.L., Narayan M., Saupe G. & Gardea-Torresdey J. (2009) The biochemistry of environmental heavy metal uptake by plants: Implications for the food chain. *The International Journal of Biochemistry & Cell Biology* **41**: 1665-1677.

Peter A.L.J. & Viraraghavan T. (2005) Thallium: a review of public health and environmental concerns. *Environment International* **31**: 493-501.

Petrie B., Barden R. & Kasprzyk-Hordern B. (2015) A review on emerging contaminants in wastewaters and the environment: Current knowledge, understudied areas and recommendations for future monitoring. *Water Research* **72**: 3-27.

Pettersen J. & Hertwich E.G. (2008) Critical Review: Life-Cycle Inventory Procedures for Long-Term Release of Metals. *Environmental Science & Technology* **42**: 4639-4647.

Petzet S., Peplinski B. & Cornel P. (2012) On wet chemical phosphorus recovery from sewage sludge ash by acidic or alkaline leaching and an optimized combination of both. *Water Research* **46**: 3769-3780.

Pichtel J. (2016) Oil and Gas Production Wastewater: Soil Contamination and Pollution Prevention. *Applied and Environmental Soil Science* **2016**: 24.

Pillay K., von Blottnitz H. & Petersen J. (2003) Ageing of chromium(III)-bearing slag and its relation to the atmospheric oxidation of solid chromium(III)-oxide in the presence of calcium oxide. *Chemosphere* **52**: 1771-1779.

Pinnekamp J., Montag D. & Everding W. (2010) Recycling of plant nutrients, especially phosphorus - overview and preliminary results of the German Funding Programme. 7. FONA-FORUM, Berlin.

Pitman R.M. (2006) Wood ash use in forestry – a review of the environmental impacts. *Forestry: An International Journal of Forest Research* **79**: 563-588.

Pizzorno L. (2015) Nothing Boring About Boron. *Integrative Medicine: A Clinician's Journal* **14**: 35-48.

Plants Database (2016) Nutrient content of crops [ONLINE], Available at: **http://plants.usda.gov/npk/main** [Accessed 22 March 2016]. United States Department of Agriculture.

Plaza C., Sanz R., Clemente C., Fernandez J.M., Gonzalez R., Polo A. & Colmenarejo M.F. (2007) Greenhouse evaluation of struvite and sludges from municipal wastewater treatment works as phosphorus sources for plants. *Journal of Agricultural and Food Chemistry* **55**: 8206-8212.

Pogue J. & Yusuf S. (1998) Overcoming the limitations of current meta-analysis of randomised controlled trials. *Lancet* **351**: 47-52.

Pokhrel D. & Viraraghavan T. (2004) Treatment of pulp and paper mill wastewater—a review. *Science of the Total Environment* **333**: 37-58.

Polubesova T., Zadaka D., Groisman L. & Nir S. (2006) Water remediation by micelle–clay system: Case study for tetracycline and sulfonamide antibiotics. *Water Research* **40**: 2369-2374.

Popowska M., Rzeczycka M., Miernik A., Krawczyk-Balska A., Walsh F. & Duffy B. (2012) Influence of Soil Use on Prevalence of Tetracycline, Streptomycin, and Erythromycin Resistance and Associated Resistance Genes. *Antimicrobial Agents and Chemotherapy* **56**: 1434-1443.

Pöykiö R., Nurmesniemi H., Dahl O. & Makela M. (2014) Chemical fractionation method for characterization of biomass-based bottom and fly ash fractions from large-sized power plant of an integrated pulp and paper mill complex. *Transactions of Nonferrous Metals Society of China* **24**: 588-596.

Pöykiö R., Rönkkömäki H., Nurmesniemi H., Perämäki P., Popov K., Välimäki I. & Tuomi T. (2009) Chemical and physical properties of cyclone fly ash from the grate-fired boiler incinerating forest residues at a small municipal district heating plant (6 MW). *Journal of Hazardous Materials* **162**: 1059-1064.

Prabakar J., Dendorkar N. & Morchhale R.K. (2004) Influence of fly ash on strength behavior of typical soils. *Construction and Building Materials* **18**: 263-267.

Press F. & Siever R. (1974) *Earth*. Freeman, W.H. and co., San Francisco.

Priester T., Köster R. & Eberle S.H. (1996) *Charakterisierung kohlenstoffhaltiger Bestandteile in Hausmüllverbrennungsschlacken unter besonderer Berücksichtigung organischer Stoffe. In: Müll und Abfall*. Institut für Technische Chemie - Bereich Wasser- und Geotechnologie (ITC-WGT) (ITC-WGT), Karlsruhe.

Proctor D.M., Fehling K.A., Shay E.C., Wittenborn J.L., Green J.J., Avent C., . . . Zak M.A. (2000) Physical and Chemical Characteristics of Blast Furnace, Basic Oxygen Furnace, and Electric Arc Furnace Steel Industry Slags. *Environmental Science & Technology* **34**: 1576-1582.

PYREG (2016) PYREG GmbH [ONLINE], Available at: **http://www.pyreg.de/home.html** [Accessed 26 May 2016].

Qadir S., Qureshi M.I., Javed S. & Abdin M.Z. (2004) Genotypic variation in phytoremediation potential of Brassica juncea cultivars exposed to Cd stress. *Plant Science* **167**: 1171-1181.

Quintana M., Colmenarejo M.F., Barrera J., García G., García E. & Bustos A. (2004) Use of a Byproduct of Magnesium Oxide Production To Precipitate Phosphorus and Nitrogen as Struvite from Wastewater Treatment Liquors. *Journal of Agricultural and Food Chemistry* **52**: 294-299.

Rahman M.M., Salleh M.A.M., Rashid U., Ahsan A., Hossain M.M. & Ra C.S. (2014) Production of slow release crystal fertilizer from wastewaters through struvite crystallization – A review. *Arabian Journal of Chemistry* **7**: 139-155.

Ravi S., Sharratt B.S., Li J.R., Olshevski S., Meng Z.J. & Zhang J.G. (2016) Particulate matter emissions from biochar-amended soils as a potential tradeoff to the negative emission potential. *Scientific Reports* **6**.

RecoPhos (2016) RecoPhos, Close the circle, Wie Phoenix aus der Asche [ONLINE], Available at: **http://www.recophos.de/index.php?id=12** [Accessed 17 May 2016].

Regenis - Bio Energie Technologie (2016) MAX Pyrolysereaktor Available at: **http://www.rewenergy.de/produkte/max-pyrolysereaktor.html#content** [Accessed 26 May 2016].

Rehder D. (2015) The role of vanadium in biology. *Metallomics* **7**: 730-742.

Reibe K., Roß C.-L. & Ellmer F. (2015) Hydro-/Biochar application to sandy soils: impact on yield components and nutrients of spring wheat in pots. *Archives of Agronomy and Soil Science* **61**: 1055-1060.

Reijkonen I. (2017) Chemical bioavailability of chromium and vanadium species in soil: risk assessment of the use of steel industry slags as liming materials. PhD thesis, University of Helsinki, Finland.

Reijonen I. (2017) Chemical bioavailability of chromium and vanadium species in soil: Risk assessment of the use of steel industry slags as liming materials. PhD thesis, University of Helsinki.

Reijonen I., Metzler M. & Hartikainen H. (2016) Impact of soil pH and organic matter on the chemical bioavailability of vanadium species: The underlying basis for risk assessment. *Environmental Pollution* **210**: 371-379.

Reimann C., Birke M., Demetriades A., Filzmoser P. & Ȯ́Connor P. (2014) Chemistry of Europe's Agricultural Soils. Part B: General Background Information and Further Analysis of Gemas Data Set. Geologisches Jahrbuch Reihe B, Hannover.

Reiter M.S. & Middleton A. (2016) Nutrient availability from poultry litter co-products - Appendix F. Virginia.

Remondis Aqua (2016) Phoenix from the ashes - Remondis Acqua paves the way for recovering phosphorus from sewage sludge ashes [ONLINE], Available at: **http://www.remondis-aktuell.com/en/remondis-aktuell/032014/water/phoenix-from-the-ashes/** [Accessed 20 May 2016].

Repetto G., del Peso A. & & Repetto M. (1998) Human thallium toxicity. In J. O. Nriagu (Ed.), Thallium in the environment 29 (pp. 167–199). New York: Wiley.

Reuter H.I., Lado L.R., Hengl T. & Montanarella L. (2008) Continental-scale digital Soil Mapping using European Soil Profile Data: Soil pH. *SAGA – Seconds Out. Hamburger Beiträge zur Physischen Geographie und Landschaftsökologie,* Vol. 19, Böhner J., Blaschke T. & Montanarella L. (eds.), pages. 113-126. Universität Hamburg Institut für Geographie, Hamburg.

Rex M., Drissen P., Bartsch S., Breuer J. & Pischke J. (2013) Pflanzenverfügbarkeit von Phosphaten aus Klärschlamm- und Tiermehlaschen nach Aufschluss in flüssiger Konverterschlacke. VDLUFA Schriftenreihe 69.

Rey-Salgueiro L., Omil B., Merino A., Martínez-Carballo E. & Simal-Gándara J. (2016) Organic pollutants profiling of wood ashes from biomass power plants linked to the ash characteristics. *Science of the Total Environment* **544**: 535-543.

Richard F.C. & Bourg A.C.M. (1991) Aqueous geochemistry of chromium: A review. *Water Research* **25**: 807-816.

Rigby H., Dowding A., Fernandes A., Humphries D., Petch R., Reynolds C., . . . Smith S. (2015) Organic Contaminant Content and Physico-Chemical Characteristics of Waste Materials Recycled in Agriculture. *Agriculture* **5**: 1289.

Risnes H., Fjellerup J., Henriksen U., Moilanen A., Norby P., Papadakis K., . . . Sorensen L.H. (2003) Calcium addition in straw gasification. *Fuel* **82**: 641-651.

Ro K.S., Cantrell K.B. & Hunt P.G. (2010) High-Temperature Pyrolysis of Blended Animal Manures for Producing Renewable Energy and Value-Added Biochar. *Industrial & Engineering Chemistry Research* **49**: 10125-10131.

Rodríguez-Reinoso F. & Molina-Sabio M. (1992) Activated carbons from lignocellulosic materials by chemical and/or physical activation: an overview. *Carbon* **30**: 1111-1118.

Rohr A.C., Campleman S.L., Long C.M., Peterson M.K., Weatherstone S., Quick W. & Lewis A. (2015) Potential Occupational Exposures and Health Risks Associated with Biomass-Based Power Generation. *International Journal of Environmental Research and Public Health* **12**: 8542-8605.

Römbke J., Moser T. & Moser H. (2009) Ecotoxicological characterisation of 12 incineration ashes using 6 laboratory tests. *Waste Management* **29**: 2475-2482.

Römheld V. & Marschner H. (1986) Mobilization of iron in the rhizosphere of different plant species. *Advances in Plant Nutrition, Volume 2*, Tinker B. & Läuchli A. (eds.), pages. 155-204. Praeger Scientific, New York.

Ronteltap M., Maurer M. & Gujer W. (2007) The behaviour of pharmaceuticals and heavy metals during struvite precipitation in urine. *Water Research* **41**: 1859-1868.

Ross J.J., Zitomer D.H., Miller T.R., Weirich C.A. & McNamara P.J. (2016) Emerging investigators series: pyrolysis removes common microconstituents triclocarban, triclosan, and nonylphenol from biosolids. *Environmental Science-Water Research & Technology* **2**: 282-289.

Rossell J.B. (2001) *Frying: improving quality*. Woodhead, Cambridge, England.

Roy R.N., Finck A., Blair G.J. & Tandon H.L.S. (2006) Nutrient management guidelines for some major field crops. *Plant nutrition for food productivity - A guide for integrated nutrient management*, FAO (ed.), pages. 235-348. FAO, Rome.

Ruamrungsri S., Bundithya W., Potapohn N., Ohtake N., Sueyoshi K. & Ohyama T. (2011) Effect of NPK Levels on Growth and Bulb Quality of Some Geophytes in Substrate Culture. *X International Symposium on Flower Bulbs and Herbaceous Perennials,* Vol. 886, VanDenEnde J.E., Krikke A.T. & DenNijs A.P.M. (eds.), pages. 213-218.

Ruiz Diaz D.A., Mueller N.D., Heller K. & Nelson N.O. (2010) Phosphorus recovered from feedlot manure as fertilizer source for corn and soybean. Kansas City.

Ruschetta S., Mosello R. & Carcano A. (2003) Trace metal measurements in atmospheric depositions at three sites in Northern Italy. Methodology and preliminary results. *Forest Snow and Landscape Research* **802**: 191-200.

Rutherford D.W., Wershaw R.L. & Cox L.G. (2004) Changes in Composition and Porosity Occurring During the Thermal Degradation of Wood and Wood Components. US Geological Survey, Scientific Investigation Report 2004-5292.

Saarsalmi A., Smolander A., Kukkola M. & Arola M. (2010) Effect of wood ash and nitrogen fertilization on soil chemical properties, soil microbial processes, and stand growth in two coniferous stands in Finland. *Plant and Soil* **331**: 329-340.

Sahlstrom L., Bagge E., Emmoth E., Holmqvist A., Danielsson-Tham M.L. & Albihn A. (2008) A laboratory study of survival of selected microorganisms after heat treatment of bio-waste used in biogas plants. *Bioresource Technology* **99**: 7859-7865.

Salado R., Daly E., Vencovsky D., Zamparutti T. & Palfrey R. (2009) Environmental, economic and social impact of the use of sewage sludge on land. Consultation report on options and impacts. Brussels.

Sardans J. & Peñuelas J. (2015) Trees increase their P:N ratio with size. *Global ecology and biogeography : a journal of macroecology* **24**: 147-156.

Sartaj M. & Fernandes L. (2005) Adsorption of boron from landfill leachate by peat and the effect of environmental factors. *Journal of Environmental Engineering and Science* **4**: 19-28.

Sarteel M., Tostivint C., Landowski A. & et al (2016) Resource Efficiency in Practice – Closing Mineral Cycles. Final report. BIO by Deloitte, Ecologic Institute, AMEC, Milan University, DTU, LEI. Report for European Commission, DG Environment. Brussels, Belgium.

Saunders S.E., Bartelt-Hunt S.L. & Bartz J.C. (2008) Prions in the environment: Occurrence, fate and mitigation. *Prion* **2**: 162-169.

Saveyn H. & Eder P. (2014) End-of-Waste criteria for biodegradable waste subjected to biological treatment (compost and digestate) - technical proposals. Publications Office of the European Commission.

Scarlat N., Martinov M. & Dallemand J.-F. (2010) Assessment of the availability of agricultural crop residues in the European Union: Potential and limitations for bioenergy use. *Waste Management* **30**: 1889-1897.

Scharf S., Schneider M. & Zethner G. (1997) Zur Situation der Verwertung und Entsorgung des kommunalen Klärschlammes in Österreich. MONOGRAPHIEN Band 095 M-095, Federal Environment Agency – Austria. .

Schaum C. (2007) Processes for future sewage sludge treatment - sewage sludge conditioning and phosphorus recovery from sewage sludge ash. Thesis, Technische Universitat Darmstadt, Darmstadt, Germany.

Scheidig K. (2009) Economic and energy aspects of phosphorus recycling from sewage sludge. *Korrespondenz Abwasser Abfall* **56**: 1138-1146.

Schiemenz K. & Eichler-Löbermann B. (2010) Biomass ashes and their phosphorus fertilizing effect on different crops. *Nutrient Cycling in Agroecosystems* **87**: 471-482.

Schiemenz K., Kern J., Paulsen H.-M., Bachmann S. & Eichler-Löbermann B. (2011) Phosphorus fertilizing effects of biomass ashes. *Recycling of Biomass Ashes*, Insam H. & Knapp B.A. (eds.), pages. 17-31. Springer Berlin Heidelberg, Berlin, Heidelberg.

Schimmelpfennig S. & Glaser B. (2012) One Step Forward toward Characterization: Some Important Material Properties to Distinguish Biochars. *Journal of Environmental Quality* **41**: 1001-1013.

Schimmelpfennig S., Müller C., Grünhage L., Koch C. & Kammann C. (2014) Biochar, hydrochar and uncarbonized feedstock application to permanent grassland—Effects on greenhouse gas emissions and plant growth. *Agriculture, Ecosystems & Environment* **191**: 39-52.

Schmidt H.P. & Wilson K. (2016) The 55 uses of biochar - the Biochar Journal 2014, Arbaz, Switzerland. [ONLINE], Available at: **www.biochar-journal.org/en/ct/2** [Accessed 26 May 2016].

Schoderboeck L., Mühlegger S., Losert A., Gausterer C. & Hornek R. (2011) Effects assessment: Boron compounds in the aquatic environment. *Chemosphere* **82**: 483-487.

Schönberg A., Samiei K., Kern H. & Raupenstrauch H. (2014) The recophos process: recovering phosphorus from sewage. *Österreichische Wasser- und Abfallwirtschaft* **66**: 403-407.

Schoumans O. (2015) Phosphorus leaching from soils: process description, risk assessment and mitigation. PhD thesis, Wageningen University, Wageningen, NL.

Schoumans O.F., Rulkens W.H., Oenema O. & Ehlert P.A.I. (2010) Phosphorus recovery from animal manure - technical opportunities and agro-economical perspectives. Alterra report 2158. Wageningen.

Schoumans O.F., Bouraoui F., Kabbe C., Oenema O. & van Dijk K.C. (2015) Phosphorus management in Europe in a changing world. *Ambio* **44**: S180-S192.

Schulze M., Mumme J., Funke A. & Kern J. (2016) Effects of selected process conditions on the stability of hydrochar in low-carbon sandy soil. *Geoderma* **267**: 137-145.

Schuurmans Stekhoven S. (2015) Memo energie en grondstoffenfabriek. Utrecht, RVO.

Schwarzer G., Carpenter J.R. & Rücker G. (2015) An Introduction to meta-analysis in R. *Meta-Analysis with R*, pages. Springer International Publishing, Cham.

Scurlock J.M.O., Dayton D.C. & Hames B. (2000) Bamboo: an overlooked biomass resource? *Biomass & Bioenergy* **19**: 229-244.

Searle S. & Malins C. (2013) Availability of cellulosic residues and wastes in the EU. New York.

Secretariat of the Stockholm Convention on Persistent Organic Pollutants (2008) Waste incinerators - guidelines on best available techniques and provisional guidance on best environmental practices relevant to Article 5 and Annex C of the Stockholm Convention on Persistent Organic Pollutants. Geneva.

Seufert V. & Ramankutty N. (2017) Many shades of gray—The context-dependent performance of organic agriculture. *Science Advances* **3**.

Severin M., Breuer J., Rex M., Stemann J., Adam C., Weghe H.v.d. & Kücke M. (2014) Phosphate fertilizer value of heat treated sewage sludge ash. *Plant, Soil and Environment* **60**: 555-561.

Shackley S., Sohi S., Ibarrola R., Hammond J., Mašek O., Brownsort P., . . . Haszeldine S. (2013) Biochar, Tool for Climate Change Mitigation and Soil Management. *Geoengineering Responses to Climate Change: Selected Entries from the Encyclopedia of Sustainability Science and Technology*, Lenton T. & Vaughan N. (eds.), pages. 73-140. Springer New York, New York, NY.

Shahid M., Shamshad S., Rafiq M., Khalid S., Bibi I., Niazi N.K., . . . Rashid M.I. (2017) Chromium speciation, bioavailability, uptake, toxicity and detoxification in soil-plant system: A review. *Chemosphere* **178**: 513-533.

Sheppard S., Long J., Sanipelli B. & Sohlenius G. (2009) Solid/liquid partition coefficients (Kd) for selected soils and sediments at Forsmark and Laxemar-Simpevarp. SKB Rapport R-09-27. Swedish Nuclear Fuel

and Waste Management Co.

Shibamoto T., Yasuhara A. & Katami T. (2007) Dioxin Formation from Waste Incineration. *Reviews of Environmental Contamination and Toxicology: Continuation of Residue Reviews*, Ware G.W., Whitacre D.M. & Gunther F.A. (eds.), pages. 1-41. Springer New York, New York, NY.

Shigeru M., Hideto F., Emi Y., Takayoshi K., Akira O. & Kensuke N. (1997) Bioaccumulation of Antimony by Chlorella vulgaris and the Association Mode of Antimony in the Cell. *Applied Organometallic Chemistry* **11**: 393-396.

Siciliano A. (2016) Assessment of fertilizer potential of the struvite produced from the treatment of methanogenic landfill leachate using low-cost reagents. *Environmental Science and Pollution Research* **23**: 5949-5959.

Siddiqui S. & Adams W.A. (2002) The fate of diesel hydrocarbons in soils and their effect on the germination of perennial ryegrass. *Environmental Toxicology* **17**: 49-62.

Siebers N., Godlinski F. & Leinweber P. (2014) Bone char as phosphorus fertilizer involved in cadmium immobilization in lettuce, wheat, and potato cropping. *Journal of Plant Nutrition and Soil Science* **177**: 75-83.

SIFCO (2017) Rapport d’activité 2015 - Syndicat des Industries Françaises des COproduits.

Sigurnjak I., Michels E., Crappé S., Buysens S., Tack F.M.G. & Meers E. (2016) Utilization of derivatives from nutrient recovery processes as alternatives for fossil-based mineral fertilizers in commercial greenhouse production of Lactuca sativa L. *Scientia Horticulturae* **198**: 267-276.

Silva F.C., Borrego C., Keizer J.J., Amorim J.H. & Verheijen F.G.A. (2015) Effects of moisture content on wind erosion thresholds of biochar. *Atmospheric Environment* **123, Part A**: 121-128.

Singer A.C., Shaw H., Rhodes V. & Hart A. (2016) Review of Antimicrobial Resistance in the Environment and Its Relevance to Environmental Regulators. *Frontiers in Microbiology* **7**: 1728.

Singh R., Gautam N., Mishra A. & Gupta R. (2011a) Heavy metals and living systems: An overview. *Indian Journal of Pharmacology* **43**: 246-253.

Singh S., Ram L.C., Masto R.E. & Verma S.K. (2011b) A comparative evaluation of minerals and trace elements in the ashes from lignite, coal refuse, and biomass fired power plants. *International Journal of Coal Geology* **87**: 112-120.

Six L., Hoxha A. & Langeveld K. (2014) Is therea potential for P recyclingin the mineral fertilizer industry? Sustainable Phosphorus Summit, 1-3 September, Le Corum, Montpellier, France.

Skodras G., Prokopidou M. & Sakellaropoulos G.P. (2006) Leaching and toxicity behavior of coal–biomass waste cocombustion ashes. *Environmental Toxicology* **21**: 317-323.

Smit C.E. (2012) *Environmental risk limits for vanadium in water - A proposal for water quality standards in accordance with the Water Framework Directive.* . RIVM Letter Report 601714021/2012, Bilthoven, the Netherlands.

Smith C.R., Buzan E.M. & Lee J.W. (2013a) Potential Impact of Biochar Water-Extractable Substances on Environmental Sustainability. *ACS Sustainable Chemistry & Engineering* **1**: 118-126.

Smith C.R., Hatcher P.G., Kumar S. & Lee J.W. (2016) Investigation into the Sources of Biochar Water-Soluble Organic Compounds and Their Potential Toxicity on Aquatic Microorganisms. *ACS Sustainable Chemistry & Engineering* **4**: 2550-2558.

Smith P.G., Boutin C. & Knopper L. (2013b) Vanadium Pentoxide Phytotoxicity: Effects of Species Selection and Nutrient Concentration. *Archives of Environmental Contamination and Toxicology* **64**: 87-96.

Smith S. & Kwan M.K.H. (1989) Use of aquatic macrophytes as a bioassay method to assess relative toxicity, uptake kinetics and accumulated forms of trace metals. *Hydrobiologia* **188**: 345-351.

Smith S.R. (2009) Organic contaminants in sewage sludge (biosolids) and their significance for agricultural recycling. *Philosophical Transactions of the Royal Society A: Mathematical, Physical and Engineering Sciences* **367**: 4005-4041.

Smolders E., Oorts K., Van Sprang P., Schoeters I., Janssen C.R., McGrath S.P. & Mclaughlin M.J. (2009) Toxicity of Trace Metals in Soil as Affected by Soil Type and Aging After Contamination: Using Calibrated Bioavailability Models to Set Ecological Soil Standards. **28**: 1633-1642.

Sokka L., Antikainen R. & Kauppi P. (2004) Flows of nitrogen and phosphorus in municipal waste: a substance flow analysis in Finland. *Progress in Industrial Ecology* **1**: 165-186.

Someus E. (2015) Risk of biochar use - organic pollutants: PCBs and PCDDs/PCDFs. Refertil Newsletter - Issue 2015.

Someus E. & Pugliese M. (2018) Concentrated Phosphorus Recovery from Food Grade Animal Bones. *Sustainability* **10**.

Song W., Ding Y., Chiou C.T. & Li H. (2010) Selected Veterinary Pharmaceuticals in Agricultural Water and Soil from Land Application of Animal Manure. *Journal of Environmental Quality* **39**: 1211-1217.

Spellman F.R. & Whiting N.E. (2007) *Environmental Management of Concentrated Animal Feeding Operations (CAFOs)*. Taylor and Francis, Boca Raton.

Spokas K.A., Novak J.M., Stewart C.E., Cantrell K.B., Uchimiya M., DuSaire M.G. & Ro K.S. (2011) Qualitative analysis of volatile organic compounds on biochar. *Chemosphere* **85**: 869-882.

Srivastava S., Tripathi R.D. & Dwivedi U.N. (2004) Synthesis of phytochelatins and modulation of antioxidants in response to cadmium stress in Cuscuta reflexa – an angiospermic parasite. *Journal of Plant Physiology* **161**: 665-674.

Staroń P., Kowalski Z., Staroń A., Seidlerová J. & Banach M. (2016) Residues from the thermal conversion of waste from the meat industry as a source of valuable macro- and micronutrients. *Waste Management* **49**: 337-345.

Steenari B.M., Schelander S. & Lindqvist O. (1999) Chemical and leaching characteristics of ash from combustion of coal, peat and wood in a 12 MW CFB – a comparative study. *Fuel* **78**: 249-258.

Steiner C., Sanchez-Monedero M.A. & Kammann C. (2015) biochar as an additive to compost. *Biochar environmental management - science, technology and implementation*, Lehmann J. & Joseph S. (eds.), pages. 717-738. Routledge, Oxon.

Steinhauser G., Brandl A. & Johnson T.E. (2014) Comparison of the Chernobyl and Fukushima nuclear accidents: A review of the environmental impacts. *Science of the Total Environment* **470–471**: 800-817.

Stenmark A., Jense C., AQuested T. & Moates G. (2016) Estimates of European food waste levels. FUSIONS EU project.

Steppich D. (2015) The thermo-reductive RecoPhos process – From sewage sludge ashes to elementary phosphorus for industrial applications. RecoPhos – Demonstration Event Leoben, Leoben, Austria.

Storer N.L. & Nelson T.S. (1968) The effect of various aluminum compounds on chick performance. *Poultry Science* **47**: 244-246.

STOWA (2011) Fosfaatterugwinning in communale afvalwaterzuiveringsinstallaties. STOWA Report 24/2011.

STOWA (2015) Verkenning van de kwaliteit van struviet uit de communale afvalwaterketen (in dutch). STOWA Report 2015-35, Amersfoort, NL.

STOWA (2016a) Levenscyclusanalyse van grondstoffen uit rioolwater. STOWA Rapport 22.2016, Amersfoort.

STOWA (2016b) Struviet en struviethoudende producten uit communaal afvalwater. STOWA 2016-12. Amersfoort, NL.

Straughan I.R., Elseewi A.A. & Page A.L. (1978) Mobilization of selected trace elements in residues from coal combustions with a special reference to fly ash. *Trace Substances and Environmental Health* **12**: 389-402.

Styles D., Adams P., Thelin G., Vaneeckhaute C., Chadwick D. & Withers P.J.A. (2018) Life Cycle Assessment of Biofertilizer Production and Use Compared with Conventional Liquid Digestate Management. *Environmental Science & Technology* **52**: 7468-7476.

Subbarao G.V., Ito O., Berry W.L. & Wheeler R.M. (2003) Sodium—A Functional Plant Nutrient. *Critical Reviews in Plant Sciences* **22**: 391-416.

Sütterlin H., Trittler R., Bojanowski S., Stadlbauer E.A. & Kümmerer K. (2007) Fate of Benzalkonium Chloride in a Sewage Sludge Low Temperature Conversion Process Investigated by LC-LC/ESI-MS/MS. **35**: 81-87.

Sutton M.A., Howard C.M., Erisman J.W., Billen G., Bleeker A., Grennfelt P., . . . Grizetti B. (2011) *The European Nitrogen Assessment*. Cambridge University Press, Cambridge.

Świetlik R., Trojanowska M., Karbowska B. & Zembrzuski W. (2016) Speciation and mobility of volatile heavy metals (Cd, Pb, and Tl) in fly ashes. *Environmental Monitoring and Assessment* **188**: 637.

Syers J.K., Johnston A.E. & Curtin D. (2008) Efficiency of soil and fertilizer phosphorus use - Reconciling changing concepts of soil phosphorus behaviour with agronomic information. FAO Fertilizer and plant nutrition bulletin No18. FAO, Rome.

Takahashi M., Kato S., Shima H., Sarai E., Ichioka T., Hatyakawa S. & Miyajiri H. (2001) Technology for recovering phosphorus from incinerated wastewater treatment sludge. *Chemosphere* **44**: 23-29.

Talboys P.J., Heppell J., Roose T., Healey J.R., Jones D.L. & Withers P.J.A. (2016) Struvite: a slow-release fertiliser for sustainable phosphorus management? *Plant and Soil* **401**: 109-123.

Tan Z. & Lagerkvist A. (2011) Phosphorus recovery from the biomass ash: A review. *Renewable and Sustainable Energy Reviews* **15**: 3588-3602.

The Ellen MacArthur Foundation (2014) Towards the circular economy - Accelerating the scale-up across global supply chains. Cowes, UK.

The Weinberg Group Inc. (2000) Health risk evaluation of select metals in inorganic fertilizers post application. Report prepared for the Fertilizer Institute (TFI). Washington, 140 pp.

Theis M., Skrifvars B.J., Hupa M. & Tran H.H. (2006) Fouling tendency of ash resulting from burning mixtures of biofuels. Part 1: Deposition rates. *Fuel* **85**: 1125-1130.

Thompson L.B. (2013) Field evaluation of the availability for corn and soybean of phosphorus recovered as struvite from corn fiber processing for bioenergy. Graduate Theses and Dissertations. Paper 13173.

Thornberg D. (2015) Genanvendelse af fosfor fra aske og rejektvand - projekt EDASK og GAFOKIR. EDASK Project Meeting, Kopenhagen.

Thy P., Lesher C.E. & Jenkins B.M. (2000) Experimental determination of high-temperature elemental losses from biomass slag. *Fuel* **79**: 693-700.

Thy P., Jenkins B.M., Grundvig S., Shiraki R. & Lesher C.E. (2006) High temperature elemental losses and mineralogical changes in common biomass ashes. *Fuel* **85**: 783-795.

Thy P., Lesher C.E., Jenkins B.M., Gras M.A. & Shiraki R. (2008) Trace mobilization during comustion of biomass fuels. Pier final project report. Commission contract No 500-02-004. Davis.

Tillman D.A. (2000) Biomass cofiring: the technology, the experience, the combustion consequences. *Biomass & Bioenergy* **19**: 365-384.

Titirici M.-M., Antonietti M. & Baccile N. (2008) Hydrothermal carbon from biomass: a comparison of the local structure from poly- to monosaccharides and pentoses/hexoses. *Green Chemistry* **10**: 1204-1212.

Tomasi Morgano M., Leibold H., Richter F., Stapf D. & Seifert H. (2018) Screw pyrolysis technology for sewage sludge treatment. *Waste Management* **73**: 487-495.

Tong L., Li P., Wang Y. & Zhu K. (2009) Analysis of veterinary antibiotic residues in swine wastewater and environmental water samples using optimized SPE-LC/MS/MS. *Chemosphere* **74**: 1090-1097.

Tortosa Masiá A.A., Buhre B.J.P., Gupta R.P. & Wall T.F. (2007) Characterising ash of biomass and waste. *Fuel Processing Technology* **88**: 1071-1081.

Tóth G., Guicharnaud R.-A., Tóth B. & Hermann T. (2014) Phosphorus levels in croplands of the European Union with implications for P fertilizer use. *European Journal of Agronomy* **55**: 42-52.

Tschan M., Robinson B. & Schulin R. (2008) Antimony uptake by Zea mays (L.) and Helianthus annuus (L.) from nutrient solution. *Environmental Geochemistry and Health* **30**: 187-191.

Twining B.S., Twiss M.R. & Fisher N.S. (2003) Oxidation of Thallium by Freshwater Plankton Communities. *Environmental Science & Technology* **37**: 2720-2726.

Tyler G. (1978) Leaching rates of heavy metal ions in forest soil. *Water, Air, and Soil Pollution* **9**: 137-148.

Uchimiya M. (2014) Changes in Nutrient Content and Availability During the Slow Pyrolysis of Animal Wastes. *Applied Manure and Nutrient Chemistry for Sustainable Agriculture and Environment*, He Z. & Zhang H. (eds.), pages. 53-68. Springer Netherlands, Dordrecht.

Uchimiya M., Cantrell K.B., Hunt P.G., Novak J.M. & Chang S. (2012) Retention of Heavy Metals in a Typic Kandiudult Amended with Different Manure-based Biochars. *Journal of Environmental Quality* **41**: 1138-1149.

Udikovic-Kolic N., Wichmann F., Broderick N.A. & Handelsman J. (2014) Bloom of resident antibiotic-resistant bacteria in soil following manure fertilization. *Proceedings of the National Academy of Sciences* **111**: 15202-15207.

Ueno Y. & Fujii M. (2001) Three Years Experience of Operating and Selling Recovered Struvite from Full-Scale Plant. *Environmental Technology* **22**: 1373-1381.

Umantaheswaran K. & Batra V.S. (2008) Physico-chemical characterisation of Indian biomass ashes. *Fuel* **87**: 628-638.

UMK-AG (2000) Arbeitsgruppe der Umweltministerkonferenz "Ursachen der Klärschlammbelastung mit gefährlichen Stoffen, Maßnahmenplan" (2000): Abschlussbericht "Ursachen der Klärschlammbelastung mit gefährlichen Stoffen, Maßnahmenplan". Preprint, 350 p.

UNEP (2000) Cleaner production assessment in fish processing. Paris, France.

UNEP (2004) General Technical Guideline on the Environmentally Sound Management of Wastes Consisting of, Containing or Contaminated with Persistent Organic Pollutants. Open-ended Working Group Report of the Basel Convention on the Control of Transboundary Movements of Hazardous Wastes and Their Disposal.

United Nations Economic Commission for Europe (UN ECE) (2009) Recent Results and Updating of Scientific and Technical Knowledge: Health Risks of Air Pollution from Biomass Combustion. Report by the Task Force on Health, Executive Body for the Convention on Long-range Transboundary Air Pollution, ECE/EB.AIR/WG.1/2009/12. United Nations Economic Commission for Europe (UN ECE); Geneva, Switzerland: p. 12.

Uysal A. & Kuru B. (2015) The fertilizer effect of struvite recovered from dairy industry wastewater on the growth and nutrition of maize plant. *Fresenius Environmental Bulletin* **24**: 3155-3162.

Uysal A., Yilmazel Y.D. & Demirer G.N. (2010) The determination of fertilizer quality of the formed struvite from effluent of a sewage sludge anaerobic digester. *Journal of Hazardous Materials* **181**: 248-254.

Uysal A., Demir S., Sayilgan E., Eraslan F. & Kucukyumuk Z. (2014) Optimization of struvite fertilizer formation from baker’s yeast wastewater: growth and nutrition of maize and tomato plants. *Environmental Science and Pollution Research* **21**: 3264-3274.

Valeur I. (2011) Speciation of heavy metals and nutrient elements in digestate. PhD thesis, Norwegian University of Life Sciences.

Vamvuka D. & Zografos D. (2004) Predicting the behaviour of ash from agricultural wastes during combustion. *Fuel* **83**: 2051-2057.

Vamvuka D. & Kakaras E. (2011) Ash properties and environmental impact of various biomass and coal fuels and their blends. *Fuel Processing Technology* **92**: 570-581.

Vamvuka D., Hahladakis J. & Pentari D. (2005) Leaching of toxic elements from lignite and agroresidue ashes in cultivated soils of Crete. *Energy & Fuels* **19**: 807-812.

Van Caneghem J. & Vandecasteele C. (2014) Characterisation of polycyclic aromatic hydrocarbons in flue gas and residues of a full scale fluidized bed combustor combusting non-hazardous industrial waste. *Waste Management* **34**: 2407-2413.

Van Caneghem J., Block C., Vermeulen I., Van Brecht A., Van Royen P., Jaspers M., . . . Vandecasteele C. (2010) Mass balance for POPs in a real scale fluidized bed combustor co-incinerating automotive shredder residue. *Journal of Hazardous Materials* **181**: 827-835.

van der Voet E., Lifset R.J. & Luo L. (2010) Life-cycle assessment of biofuels, convergence and divergence. *Biofuels* **1**: 435-449.

van Dijk K.C., Lesschen J.P. & Oenema O. (2016) Phosphorus flows and balances of the European Union Member States. *Science of the Total Environment* **542**: 1078-1093.

Van Eijk R.J., te Winkel H. & Stam A.F. (2011) Van Eijk R.J., te Winkel H., Stam A.F. Environmental and health aspects of coal and biomass co-combustion ashes; Proceedings of the World of Coal Ash (WOCA) Conference; Denver, CO, USA. 9–12 May; p. 11.

van Eijk R.J., Obernberger I. & Supancic K. (2012) Options for increased utilization of ash from biomass combustion and co-firing. IEA Bioenergy Task 32 Deliverable D4 - Document number 30102040-PGR/R&E 11-2142.

Van Grinsven H.J.M., Holland M., Jacobsen B.H., Klimont Z., Sutton M.a. & Jaap Willems W. (2013) Costs and Benefits of Nitrogen for Europe and Implications for Mitigation. *Environmental Science & Technology* **47**: 3571-3579.

Van Loo S. & Koppejan J. (2008) *The Handbook of Biomass Combustion and Co-Firing*. Earthscan, London.

van Vlaardingen P.L.A., Posthumus R. & Posthuma-Doodema C.J.A.M. (2005) Environmental Risk Limits for Nine Trace Elements, RIVM report 601501029/2005.

Van Wesenbeeck S., Prins W., Ronsse F. & Antal M.J. (2014) Sewage Sludge Carbonization for Biochar Applications. Fate of Heavy Metals. *Energy & Fuels* **28**: 5318-5326.

Vaneeckhaute C., Janda J., Vanrolleghem P.A., Tack F.M.G. & Meers E. (2016) Phosphorus use efficiency of bio-based fertilizers: bioavailability and fractionation. *Pedosphere* **26**: 310-325.

Vaněk A., Grösslová Z., Mihaljevič M., Trubač J., Ettler V., Teper L., . . . Ash C. (2016) Isotopic Tracing of Thallium Contamination in Soils Affected by Emissions from Coal-Fired Power Plants. *Environmental Science & Technology* **50**: 9864-9871.

Vanhoof C. & Tirez K. (2014) Analysemethode voor het bepalen van de zuiverheid van struviet. OVAM - Study Report 2014/SCT/R/26.

Vassilev S.V. & Braekman-Danheux C. (1999) Characterization of refuse-derived char from municipal solid waste - 2. Occurrence, abundance and source of trace elements. *Fuel Processing Technology* **59**: 135-161.

Vassilev S.V. & Vassileva C.G. (2007) A new approach for the classification of coal fly ashes based on their origin, composition, properties, and behaviour. *Fuel* **86**: 1490-1512.

Vassilev S.V. & Vassileva C.G. (2009) A new approach for the combined chemical and mineral classification of the inorganic matter in coal. 1. Chemical and mineral classification systems. *Fuel* **88**: 235-245.

Vassilev S.V., Braekman-Danheux C. & Laurent P. (1999) Characterization of refuse-derived char from municipal solid waste - 1. Phase-mineral and chemical composition. *Fuel Processing Technology* **59**: 95-134.

Vassilev S.V., Eskenazy G.M. & Vassileva C.G. (2000) Contents, modes of occurrence and origin of chlorine and bromine in coal. *Fuel* **79**: 903-921.

Vassilev S.V., Baxter D., Andersen L.K. & Vassileva C.G. (2010) An overview of the chemical composition of biomass. *Fuel* **89**: 913-933.

Vassilev S.V., Baxter D., Andersen L.K. & Vassileva C.G. (2013a) An overview of the composition and application of biomass ash.: Part 2. Potential utilisation, technological and ecological advantages and challenges. *Fuel* **105**: 19-39.

Vassilev S.V., Baxter D., Andersen L.K. & Vassileva C.G. (2013b) An overview of the composition and application of biomass ash. Part 1. Phase–mineral and chemical composition and classification. *Fuel* **105**: 40-76.

Vehlow J., Bergfeldt B. & Hunsinger H. (2006) PCDD/F and related compounds in solid residues from municipal solid waste incineration - a literature review. *Waste Management & Research* **24**: 404-420.

Venderbosch R.H. & Prins W. (2010) Fast pyrolysis technology development. *Biofuels, Bioproducts and Biorefining* **4**: 178-208.

Vesterinen P. (2003) Wood ash recycling - state of the art in Finland and Sweden, VTT Processes, PRO2/6107/03.

Visioli G., Conti F.D., Menta C., Bandiera M., Malcevschi A., Jones D.L. & Vamerali T. (2016) Assessing biochar ecotoxicology for soil amendment by root phytotoxicity bioassays. *Environmental Monitoring and Assessment* **188**: 166.

Vogel T., Nelles M. & Eichler-Löbermann B. (2015) Phosphorus application with recycled products from municipal wastewater to different crop species. *Ecological Engineering* **83**: 466-475.

Vollaro M., Galioto F. & Viaggi D. (2016) The circular economy and agriculture: new opportunities for re-using Phosphorus as fertilizer. *Bio-Based and Applied Economics* **5**: 267-285.

vom Eyser C., Schmidt T.C. & Tuerk J. (2016) Fate and behaviour of diclofenac during hydrothermal carbonization. *Chemosphere* **153**: 280-286.

Wallace J.S., Garner E., Pruden A. & Aga D.S. (2018) Occurrence and transformation of veterinary antibiotics and antibiotic resistance genes in dairy manure treated by advanced anaerobic digestion and conventional treatment methods. *Environmental Pollution* **236**: 764-772.

Wang H., Yan B.J. & Li F. (2015) Analysis of Cr with Various Valence States in Industrial EAF Slag for Making Stainless Steel. *Isij International* **55**: 1425-1431.

Wang L. & Nancollas G.H. (2008) Calcium Orthophosphates: Crystallization and Dissolution. *Chemical Reviews* **108**: 4628-4669.

Wang L., Li A. & Chang Y. (2016) Hydrothermal treatment coupled with mechanical expression at increased temperature for excess sludge dewatering: Heavy metals, volatile organic compounds and combustion characteristics of hydrochar. *Chemical Engineering Journal* **297**: 1-10.

Wang L., Hustad J.E., Skreiberg Ø., Skjevrak G. & Grønli M. (2012a) A Critical Review on Additives to Reduce Ash Related Operation Problems in Biomass Combustion Applications. *Energy Procedia* **20**: 20-29.

Wang S. & Shi X. (2001) Molecular mechanisms of metal toxicity and carcinogenesis. *Mol Cell Biochem* **222**: 3-9.

Wang T., Camps-Arbestain M., Hedley M. & Bishop P. (2012b) Predicting phosphorus bioavailability from high-ash biochars. *Plant and Soil* **357**: 173-187.

Wang Z., Wang F., Cao J. & Wang J. (2010) Pyrolysis of pine wood in a slowly heating fixed-bed reactor: Potassium carbonate versus calcium hydroxide as a catalyst. *Fuel Processing Technology* **91**: 942-950.

Wanty R.B. & Goldhaber M.B. (1992) Thermodynamics and kinetics ofreactions involving vanadium in natural systems: Accumulation ofvanadium in sedimentary rocks. *Geochimica Et Cosmochimica Acta* **56**: 1471-1483.

WCA environment (2014) Assessment of Risks to Soil Quality and Human Health from Organic Contaminants in Materials Commonly Spread on Land in Scotland: Final Report.

Wei X., Schnell U. & Hein K.R.G. (2005) Behaviour of gaseous chlorine and alkali metals during biomass thermal utilisation. *Fuel* **84**: 841-848.

Weidelener A., Brechtel K., Maier W., Krampe J. & Rott U. (2005) Recovery of phosphorus from sewage sludge as MAP. IWA-/WISA-Conference on the Management of Residues Emanating from Water and Wastewater Treatment, 9–12 August 2005, Johannesburg, South Africa.

Weigand H., Bertau M., Huebner W., Bohndick F. & Bruckert A. (2013) RecoPhos: Full-scale fertilizer production from sewage sludge ash. *Waste Management* **33**: 540-544.

Weiner B., Baskyr I., Poerschmann J. & Kopinke F.-D. (2013) Potential of the hydrothermal carbonization process for the degradation of organic pollutants. *Chemosphere* **92**: 674-680.

Weinfurtner K., Gäth S., Kördel W. & Waida C. (2009) Ecological testing of products from phosphorus recovery processes - first results. International conference on nutrient recovery from wastewater streams. IWA Publishing, London, UK.

Wells D.E. (2013) Poultry litter ash as a phosphorus source for greenhouse crop production. PhD thesis, Louisiana State University.

Werther J., Saenger M., Hartge E.U., Ogada T. & Siagi Z. (2000) Combustion of agricultural residues. *Progress in Energy and Combustion Science* **26**: 1-27.

WHO (1990) Environmental Health Criteria 107: Barium. International Programme on Chemical Safety, Geneva: 121. **http://www.inchem.org/documents/ehc/ehc/ehc107.htm**

WHO (1996) Thallium, Internation programme on chemical safety, Environmental Health Criteria, 182. Geneva.

WHO (2006) Guidelines for drinking-water quality, third edition, incorporating first addendum. Volume 1 — Recommendations. Geneva.

WHO (2018) Circular economy and health: opportunities and risks. WHO Regional Office for Europe, Copenhagen, Denmark.

Widen P. (1993) Composting of sorted household waste in Uppsala. Thesis, Sveriges Lantbruksinversiteit, Uppsala.

Wiechmann B., Dienemann C., Kabbe C., Vogel I. & Roskosch A. (2013a) Sewage sludge management in Germany. Umweltbundesamt (UBA).

Wiechmann B., Dienemann C., Christian Kabbe C., Simone Brandt S., Ines Vogel I. & Andrea Roskosch A. (2013b) Sewage sludge management in Germany. Umweltbundesamt (UBA), Dessau-Roßlau, Germany.

Wieck-Hansen K., Overgaard P. & Larsen O.H. (2000) Cofiring coal and straw in a 150 MWe power boiler experiences. *Biomass & Bioenergy* **19**: 395-409.

Wigley F., Williamson J., Malmgren A. & Riley G. (2007) Ash deposition at higher levels of coal replacement by biomass. *Fuel Processing Technology* **88**: 1148-1154.

Wijnands J.H.M. & Linders G.-J. (2013) Competitiveness Proofing - Fertilising Materials Exante evaluation of competitiveness impacts of the European Commission's policy proposal Review of Regulation (EC) no 2003/2003 relating to fertilisers. Technical report, ECORYS & WUR, Den Haag, Netherlands.

Wilfert P., Kumar P.S., Korving L., Witkamp G.-J. & van Loosdrecht M.C.M. (2015) The Relevance of Phosphorus and Iron Chemistry to the Recovery of Phosphorus from Wastewater: A Review. *Environmental Science & Technology* **49**: 9400-9414.

Wilken V., Zapka O. & Muskolus A. (2015) Product quality: fertilizing efficiency, results of pot and field tests. Report of the P-REX project - Sustainable sewage sludge management fostering phosphorus recovery and energy efficiency. Institute of Agricultural and Urban Ecological Projects affiliated to Berlin Humboldt University (IASP).

Wilson S.C., Lockwood P.V., Ashley P.M. & Tighe M. (2010) The chemistry and behaviour of antimony in the soil environment with comparisons to arsenic: A critical review. *Environmental Pollution* **158**: 1169-1181.

Winship K.A. (1987) Toxicity of antimony and its compounds. *Adverse Drug Reactions and Toxicological Reviews* **6**: 67-90.

Wojtkowiak T., Karbowska B., Zembrzuski W., Siepak M. & Lukaszewski Z. (2016) Miocene colored waters: A new significant source of thallium in the environment. *Journal of Geochemical Exploration* **161**: 42-48.

Woolf D., Amonette J.E., Street-Perrott F.A., Lehmann J. & Joseph S. (2010) Sustainable biochar to mitigate global climate change. *Nature Communications* **1**.

Wopenka B. & Pasteris J.D. (2005) A mineralogical perspective on the apatite in bone. *Materials Science & Engineering C-Biomimetic and Supramolecular Systems* **25**: 131-143.

World Health Organization (1996) *Trace elements in human nutrition and health*. Geneva.

World Health Organization (2004) Manganese and its compounds : environmental aspects. (Concise international chemical assessment document ; 63). Stuttgart.

Wragge V. (2015) Product quality and fertiliser value of recovered products. P-REX project final meeting, Berlin.

Wu H., Lai C., Zeng G., Liang J., Chen J., Xu J., . . . Wan J. (2017) The interactions of composting and biochar and their implications for soil amendment and pollution remediation: a review. *Critical Reviews in Biotechnology* **37**: 754-764.

Wuana R.A. & Okieimen F.E. (2011) Heavy Metals in Contaminated Soils: A Review of Sources, Chemistry, Risks and Best Available Strategies for Remediation. *ISRN Ecology* **2011**: 20.

Xia W., Du X., Zheng T., Zhang B., Li Y., Bassig B.A., . . . Xu S. (2016) A Case–Control Study of Prenatal Thallium Exposure and Low Birth Weight in China. *Environmental Health Perspectives* **124**: 164-169.

Xiao T., Guha J., Boyle D., Liu C.-Q., Zheng B., Wilson G.C., . . . Chen J. (2004) Naturally occurring thallium: a hidden geoenvironmental health hazard? *Environment International* **30**: 501-507.

Xiao X., Chen Z. & Chen B. (2016) H/C atomic ratio as a smart linkage between pyrolytic temperatures, aromatic clusters and sorption properties of biochars derived from diverse precursory materials. *Scientific Reports* **6**: 22644.

Xu C., Salsali H., Weese S. & Warriner K. (2015) Inactivation of Clostridium difficile in sewage sludge by anaerobic thermophilic digestion. *Canadian Journal of Microbiology* **62**: 16-23.

Yao F.X., Arbestain M.C., Virgel S., Blanco F., Arostegui J., Maciá-Agulló J.A. & Macías F. (2010) Simulated geochemical weathering of a mineral ash-rich biochar in a modified Soxhlet reactor. *Chemosphere* **80**: 724-732.

Yao Z.T., Ji X.S., Sarker P.K., Tang J.H., Ge L.Q., Xia M.S. & Xi Y.Q. (2015) A comprehensive review on the applications of coal fly ash. *Earth-Science Reviews* **141**: 105-121.

Ye Z.-L., Deng Y., Lou Y., Ye X. & Chen S. (2018) Occurrence of veterinary antibiotics in struvite recovery from swine wastewater by using a fluidized bed. *Frontiers of Environmental Science & Engineering* **12**: 7.

Ye Z.-L., Deng Y., Lou Y., Ye X., Zhang J. & Chen S. (2017) Adsorption behavior of tetracyclines by struvite particles in the process of phosphorus recovery from synthetic swine wastewater. *Chemical Engineering Journal* **313**: 1633-1638.

Zepke F. & Klose S. (2017) Vom Klärschlamm zum Phosphat – Dünger Das EuPhoRe®-Verfahren. .

Zevenhoven-Onderwater M., Blomquist J.P., Skrifvars B.J., Backman R. & Hupa M. (2000) The prediction of behaviour of ashes from five different solid fuels in fluidised bed combustion. *Fuel* **79**: 1353-1361.

Zevenhoven M., Yrjas P., Skrifvars B.-J. & Hupa M. (2012) Characterization of Ash-Forming Matter in Various Solid Fuels by Selective Leaching and Its Implications for Fluidized-Bed Combustion. *Energy & Fuels* **26**: 6366-6386.

Zhang H.-L., Fang W., Wang Y.-P., Sheng G.-P., Zeng R.J., Li W.-W. & Yu H.-Q. (2013) Phosphorus Removal in an Enhanced Biological Phosphorus Removal Process: Roles of Extracellular Polymeric Substances. *Environmental Science & Technology* **47**: 11482-11489.

Zhang S., Herbell J.-D. & Gaye-Haake B. (2004) Biodegradable organic matter in municipal solid waste incineration bottom ash. *Waste Management* **24**: 673-679.

Zheng H., Wang Z., Deng X., Zhao J., Luo Y., Novak J., . . . Xing B. (2013) Characteristics and nutrient values of biochars produced from giant reed at different temperatures. *Bioresource Technology* **130**: 463-471.

Zöttle H.W., Stahr K. & Hädrich F. (1979) Umsatz von Spurenelementen in der Bärhalde und ihren Ökosystemen. *Mitteilung der Deutschen Bodenkundlichen Gesellschaft* **29**: 569-576.

Zwetsloot M.J., Lehmann J., Bauerle T., Vanek S., Hestrin R. & Nigussie A. (2016) Phosphorus availability from bone char in a P-fixing soil influenced by root-mycorrhizae-biochar interactions. *Plant and Soil* **408**: 95-105.

Zwick T.C., Arthur M.F., Tolle D.A. & Van Voris P. (1984) A unique laboratory method for evaluating agro-ecosystem effects of an industrial waste product. *Plant and Soil* **77**: 395-399.

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