DRAFT STRUBIAS Interim Report

DRAFT market study for recovered phosphate salts, ash-based materials and pyrolysis materials in view of their possible inclusion as Component Material Categories in the Revised Fertiliser Regulation

Interim Report

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STRUBIAS materials as CMCs in the Revised EU Fertiliser Regulation

The Fertilisers Regulation revision aims at establishing a regulatory framework enabling production of fertilisers from recycled bio-wastes and other secondary raw materials, in line with the Bioeconomy strategy\(^1\), which encompasses the production of renewable biological resources and the conversion of these resources and waste streams into value added products.

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 phosphorus (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 wish to raise concerning the sustainable use of P. Phosphorus recycling has also been addressed by FP7 research projects, the results of which have been analysed during the workshop 'Circular approaches to phosphorus: from research to deployment', held in Berlin on 4 March 2015.

One of the identified priorities was to revise the EU Fertiliser Regulation to extend its scope to nutrients from secondary sources (e.g. recycled phosphates) and organic sources. At that moment, regulatory barriers to market rollout were noted, with the message that coherent interpretation of existing relevant EU and national legislation is a precondition for widespread implementation. Hence, a strong message from participants was the need for policy support for P recycling and for the nutrient circular economy.

The presence of secondary raw materials in the initial list of CMCs is very limited. That is due partially to the lack of agreed recovery rules and partially to the legal construction of the Animal By-products Regulation (which the Commission therefore proposes to amend). For that reason mainly, the Commission is empowered to extend the list of CMCs to additional categories by Delegated acts. In recital 55 it has declared its intention to do so for waste and animal by-products "when the manufacturing processes have been scientifically analysed and process requirements have been established at Union level".

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 removing of P from waste water in the form of struvite, incinerating and post-processing 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 a 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.

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\(^1\) http://ec.europa.eu/research/bioeconomy/index.cfm
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 biochar, ashes and struvite. These could possibly be considered as CMCs in the revised Fertiliser Regulation. The acronym STRUBIAS (STRUvite, Biochar, or incineration Ashes)\(^2\) 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 as follows:

- Techno-scientific information obtained from the STRUBIAS sub-group\(^3\) indicated that struvite that is precipitated with the intention of P-recovery is often a mixture of different minerals. Moreover, P-recovery processes that rely on the principle of precipitation can also intentionally target Ca- and Mg-phosphates other than struvite (e.g. dicalcium phosphates). Therefore, the JRC STRUBIAS Interim Report on nutrient recovery rules proposed to expand the scope of this CMC and to change the name of the CMC from "struvite" to "recovered phosphate salts". This proposal received a large degree of support from the STRUBIAS sub-group. Recovered phosphate salts involve substances and mixtures dominantly composed of calcium and magnesium phosphates that have been precipitated and isolated deliberately under controlled conditions and serve as an ingredient for a plant nutrition product.

- The designation of ash-based materials was maintained and comprises both raw ashes obtained from the incineration process as well as ashes that have been further processed with the aim to partly remove heavy metals, increase the availability of plant nutrients in the ash complexes or to form a complex multi-nutrient mineral fertiliser. Ash-based materials involve substances or mixtures that contain materials that have undergone a thermal oxidation process. Oxygen is not limiting the chemical reactions that transform input materials into ashes and/or slags.

- The designation pyrolysis material is proposed for any material that is produced from eligible input materials via production processes that cover the pyrolysis technology spectrum including gasification and hydrothermal carbonisation techniques. Both heterogeneous substances rich in aromatic carbon and minerals are considered. This terminology offers the advantage that a clear reference is made to the production technology in the name of the CMC. Pyrolysis materials involve substances or mixtures that have undergone a pyrolysis, hydrothermal carbonisation or gasification process and

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\(^2\)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)

\(^3\) The JRC is supported in the process by a technical working group, referred to as the STRUBIAS sub-group, that constitutes a sub-group 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 possible recovery rules for nutrients from eligible input materials into STRUBIAS materials.
serve as an ingredient for a plant nutrition product. The process takes 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 or pyrogenic carbonaceous
materials.
Objectives of the JRC STRUBIAS work

The Joint Research Centre (JRC) of the European Commission is assessing the existing techno-scientific evidence in view of a possible inclusion of materials containing STRUBIAS as Component Material Categories (CMC) in the Revised EC Fertiliser Regulation\(^4\). This assessment should form the basis for any technical proposals on the requirements that those candidate materials shall comply with. After assessment, STRUBIAS materials could possibly be used as component materials for the different Product Function Categories (PFCs) included in the proposal for the Revised Fertiliser Regulation, more specifically fertiliser, liming material, soil improver, growing medium, agronomic additive, plant biostimulant, and fertilising product blend.

The JRC is supported in the process by a technical working group that constitutes a subgroup of the Commission expert group on Fertilisers (hereafter STRUBIAS sub-group), 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 possible recovery rules for nutrients from eligible input materials into STRUBIAS materials and market aspects related to these materials.

The JRC assesses STRUBIAS materials against following criteria in order to provide information on the appropriateness to include recovered phosphate salts, ash-based materials and pyrolysis materials into Annex II of the Revised Fertilising Products Regulation by delegated act:

I. 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 proposal for the Revised EC Fertiliser Regulation];

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

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

Should the European Commission conclude that it is appropriate to include recovered phosphate salts, ash-based materials and pyrolysis materials into Annex II of the revised Fertilising Products Regulation by delegated act, it would be worthwhile to assess the possible impacts. This report therefore builds on the Interim Report on nutrient recovery rules and supplements it by providing for a description of the impacts of the proposed recovery rules elaborated under that work on:

(1) the foreseen EU market for fertilising products containing STRUBIAS materials and the reasonable replacement potential of conventional fertilisers by such products;

(2) the environment and human health impacts of replacing conventional fertilisers by fertilising products containing STRUBIAS materials resulting from the opening of the EU market to such products; and

(3) the production and compliance costs of products complying with the proposed recovery rules proposed and the impacts on the economy of replacing conventional fertilisers by recovered fertilising products;

As the expected impacts obviously will depend on the precise formulation of proposed nutrient recovery rules, the reports on nutrient recovery rules and the market study will be strongly intertwined, and nutrient recovery rules may have to be established in an iterative manner, taking into account the corresponding possible impacts. Therefore, it is important to run this project concurrently with the work under nutrient recovery rules.

This Interim Report starts with a description of the properties and the current fate of eligible input materials for STRUBIAS production processes (section 3), followed by an overview of the STRUBIAS production processes (section 4). This contextual information is required to make an informed choice for (1) the selection of relevant pathways for which detailed process inventories and costs assessments (section 7.2) will be performed (to be completed at a later stage in the STRUBIAS project), and (2) an assessment of processes that are associated to a high technological readiness level to estimate the market for STRUBIAS materials for the year 2030 (section 8.2).

**Agronomic efficiency** is a critical variable to determine the equivalence of P-fertilisers derived from STRUBIAS fertilisers with mined and synthetic P-fertilisers. The market outlook for STRUBIAS materials (section 8.2) is tightly linked to this parameter. Indirectly, environmental and human health impacts are also related to this parameter that the fertiliser application rates are interconnected to plant-available P, rather than to total P in the fertilising material. The agronomic efficiency of STRUBIAS materials within a relevant European context has been evaluated using meta-analyses techniques (section 5).

Possible adverse environmental or human health issues will be enforced by STRUBIAS nutrient recovery rules, but the impacts of the production and use phase of STRUBIAS materials will be evaluated in follow-up documents to this report.

**Economic aspects**, including estimates of production costs and sales prices, are presented in section 7.

Finally, an outlook of future market for STRUBIAS materials has been given as well as an estimation of the substitution effect of mined and synthetic fertilisers by fertilising products containing recovered phosphate salts, ash-based materials and pyrolysis materials.
for the year 2030 (section 8). Therefore, this document will cover aspects that relate to all three criteria against which STRUBIAS materials will be assessed.
3 Potential input materials for fertilising products containing STRUBIAS materials

3.1 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 1). 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 8, 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 1). Animal production and non-food production are associated to lower losses (Figure 1) (van Dijk et al., 2016).

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

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

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

The losses from non-food production, as designated by van Dijk et al. (2016), relate to losses from forest-based industries (woodworking, furniture industry, pulp and paper industry). The total losses equal 77 kt P yr\(^{-1}\), with wood industry waste being the dominant fraction (65 kt P yr\(^{-1}\)).
Figure 1. Phosphorus (P) use for the EU-27 in 2005 [kt P yr⁻¹] aggregated at the food and non-food production–consumption–waste chain; showing the imports (blue), exports (purple), losses (red) and internal upward/downward flows (black) for crop production (CP), animal production (AP), food processing (FP), non-food production (NF) and human consumption (HC) sectors (indicated with square blocks); the arrow thickness shows the relative flow sizes; the positive balance of +924 in CP represents annual net accumulation of P in agricultural soils in 2005 (adopted from van Dijk et al., 2016).

In the following sections, the characteristics of the designated potential input material streams by van Dijk (2016) and Buckwell and Nadeu (2016) will be reviewed in view of their potential to be used as an eligible input material for STRUBIAS production processes. Furthermore, the STRUBIAS sub-group pointed towards additional input materials that are suitable and already used for STRUBIAS production processes. Often, these input materials are used for a process aimed at the production of a different primary product (e.g. energy, clean water, steel) and STRUBIAS materials are produced or can be manufactured from residues from the process.
### 3.2 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 1) (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 1). 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 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 1) (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 1). 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 1) (Kremer, 2013; Plants Database, 2016). Grain maize has, for instance, a nutrient content that is about 3 to 4 times higher than wheat.

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 1).
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

### Table 1: 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))

<table>
<thead>
<tr>
<th>crop</th>
<th>harvest (Mt yr(^{-1}))</th>
<th>residue-to-harvest ratio</th>
<th>residue dry matter content (Mt yr(^{-1}))</th>
<th>residue nutrient concentration</th>
<th>nutrient content N (kt yr(^{-1}))</th>
<th>nutrient content P (kt yr(^{-1}))</th>
</tr>
</thead>
<tbody>
<tr>
<td>wheat and spelt</td>
<td>136</td>
<td>1.3</td>
<td>150</td>
<td>0.28</td>
<td>0.05</td>
<td>1353</td>
</tr>
<tr>
<td>barley</td>
<td>57</td>
<td>1.2</td>
<td>58</td>
<td>0.43</td>
<td>0.09</td>
<td>419</td>
</tr>
<tr>
<td>grain maize and corn-cob-mix</td>
<td>61</td>
<td>1.0</td>
<td>47</td>
<td>0.81</td>
<td>0.20</td>
<td>248</td>
</tr>
<tr>
<td>oats and spring cereal mixes</td>
<td>12</td>
<td>1.3</td>
<td>14</td>
<td>0.70</td>
<td>0.14</td>
<td>101</td>
</tr>
<tr>
<td>rye and winter cereal mixes</td>
<td>8</td>
<td>1.6</td>
<td>12</td>
<td>0.48</td>
<td>0.11</td>
<td>57</td>
</tr>
<tr>
<td>triticale</td>
<td>10</td>
<td>1.3</td>
<td>11</td>
<td>1.30</td>
<td>0.24</td>
<td>144</td>
</tr>
<tr>
<td>rice</td>
<td>3</td>
<td>1.4</td>
<td>3</td>
<td>0.67</td>
<td>0.13</td>
<td>23</td>
</tr>
<tr>
<td>root crops</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>potatoes</td>
<td>59</td>
<td>0.4</td>
<td>20</td>
<td>1.10</td>
<td>0.18</td>
<td>224</td>
</tr>
<tr>
<td>sugar beet</td>
<td>115</td>
<td>0.2</td>
<td>20</td>
<td>2.28</td>
<td>0.45</td>
<td>452</td>
</tr>
<tr>
<td>oil seeds</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>sunflower seeds</td>
<td>7</td>
<td>2.1</td>
<td>12</td>
<td>2.84</td>
<td>0.58</td>
<td>351</td>
</tr>
<tr>
<td>rape and turnip rape seeds</td>
<td>19</td>
<td>1.5</td>
<td>26</td>
<td>1.53</td>
<td>0.31</td>
<td>392</td>
</tr>
<tr>
<td>overall total</td>
<td>350</td>
<td></td>
<td>374</td>
<td></td>
<td>2795</td>
<td>559</td>
</tr>
</tbody>
</table>

\(^{1}\) calculated based on residue-to-harvest ratio and residue dry matter content
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 1, 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 recovered phosphate salts (after anaerobic digestion, although concerns exist due to the low phosphate content of anaerobically digested crop residues), ash-based materials (Insam and Knapp, 2011), and pyrolysis materials (EBC, 2012).

### 3.3 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 2) (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 2). 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 economically exported to other areas 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 2). 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 2). 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 2). 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 2). 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 2). All manure types have generally high organic matter contents (range 57% - 82%) (Table 2).

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 2). 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 2). Pig manure contributes for 9% (for N) and 10% (for P) to the total nutrients presents in manure (Table 2). Poultry and pig are often kept inside year-round, for which reasons their manure can be easily collected.
Table 2: Amount of manure in the EU-27 and its composition (sources: Agrotechnology Atlas (2016); Foget et al. (2011); Gendebien et al. (2001))

<table>
<thead>
<tr>
<th></th>
<th>Amount (Mt yr(^{-1}))</th>
<th>Dry matter (%)</th>
<th>Organic matter (%)</th>
<th>N/P concentration</th>
<th>K concentration (kt K yr(^{-1}))</th>
<th>N concentration (kt N yr(^{-1}))</th>
<th>P concentration (kt P yr(^{-1}))</th>
</tr>
</thead>
<tbody>
<tr>
<td>Pig slurry</td>
<td>148.6</td>
<td>5</td>
<td>69</td>
<td>3.7</td>
<td>4.6</td>
<td>8.3</td>
<td>2.2</td>
</tr>
<tr>
<td>Source separated pig manure</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>0.0</td>
<td>0.0</td>
<td>0.0</td>
</tr>
<tr>
<td>Solid</td>
<td>14.2</td>
<td>24</td>
<td>80</td>
<td>2.4</td>
<td>2.0</td>
<td>3.2</td>
<td>1.4</td>
</tr>
<tr>
<td>Liquid</td>
<td>8.8</td>
<td>2</td>
<td>n.a.</td>
<td>5.4</td>
<td>9.1</td>
<td>17.1</td>
<td>3.2</td>
</tr>
<tr>
<td>Pig deep litter</td>
<td>5.3</td>
<td>28</td>
<td>75</td>
<td>2.5</td>
<td>4.0</td>
<td>2.3</td>
<td>0.9</td>
</tr>
<tr>
<td>Total pig</td>
<td>176.9</td>
<td>7</td>
<td>517</td>
<td>6.5</td>
<td>517</td>
<td>847</td>
<td>248</td>
</tr>
<tr>
<td>Cattle slurry</td>
<td>447.8</td>
<td>9</td>
<td>66</td>
<td>4.6</td>
<td>4.7</td>
<td>4.5</td>
<td>1.0</td>
</tr>
<tr>
<td>Source separated cattle manure</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>0.0</td>
<td>0.0</td>
<td>0.0</td>
</tr>
<tr>
<td>Solid</td>
<td>294.9</td>
<td>22</td>
<td>64</td>
<td>3.3</td>
<td>2.1</td>
<td>2.4</td>
<td>0.7</td>
</tr>
<tr>
<td>Liquid</td>
<td>54.6</td>
<td>3</td>
<td>57</td>
<td>12.5</td>
<td>29.4</td>
<td>447</td>
<td>12.2</td>
</tr>
<tr>
<td>Cattle deep litter</td>
<td>294.9</td>
<td>25</td>
<td>77</td>
<td>4.7</td>
<td>3.2</td>
<td>2397</td>
<td>2.9</td>
</tr>
<tr>
<td>Total cattle</td>
<td>1092.1</td>
<td>17</td>
<td>6114</td>
<td>76.9</td>
<td>5694</td>
<td>613</td>
<td>1342</td>
</tr>
<tr>
<td>Poultry slurry</td>
<td>3.4</td>
<td>14</td>
<td>82</td>
<td>3.0</td>
<td>2.6</td>
<td>6.8</td>
<td>2.3</td>
</tr>
<tr>
<td>Poultry deep litter</td>
<td>109.5</td>
<td>57</td>
<td>74</td>
<td>3.1</td>
<td>2.1</td>
<td>4.4</td>
<td>1.4</td>
</tr>
<tr>
<td>Total poultry</td>
<td>112.9</td>
<td>55</td>
<td>1319</td>
<td>16.6</td>
<td>1319</td>
<td>2746</td>
<td>883</td>
</tr>
<tr>
<td>Overall total</td>
<td>1381.9</td>
<td>19</td>
<td>69</td>
<td>4.4</td>
<td>7950</td>
<td>9287</td>
<td>2473</td>
</tr>
</tbody>
</table>
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 2. The reasons for this discrepancy relate to the different reference years that are used in the studies, and the uncertainty in average values for the nutrient concentrations and dry matter contents of the diverse manure fractions. Hence, estimating manure nutrient inputs is associated to a significant degree of uncertainty, and the absolute numbers given should be interpreted with the necessary caution.

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

Landspreading of manure involves a risk for the accumulation of potentially toxic elements, pathogen recycling, and odour nuisance (Gendebien et al., 2001). Manure may contain significant amounts of metals/metalloids due to the use of mineral supplements and veterinary products. This is particularly true for pig slurry, which can contain up to 600 mg kg\(^{-1}\) dry matter of Cu, and up to 900 mg kg\(^{-1}\) dry matter of Zn. Also, cattle slurries contain high amounts of Zn (up to 750 mg kg\(^{-1}\) dry matter) (Gendebien et al., 2001). Considering the large volumes of manure applied, metals/metalloids may accumulate as a result of long-term agricultural use. Accumulation of metals/metalloids could not only affect the soil fertility, but also promote metal migration through leaching and runoff. Additionally, there is also growing concern regarding the environmental fate and potential impacts of the veterinary pharmaceuticals on human and ecosystem health as animal feeding pharmaceuticals are excreted into manure either as parent compounds or as bioactive metabolites (Song et al., 2010). Finally, animal manures contain pathogenic elements in variable 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 to high transport costs of the large-volume material and unbalanced nutrient supply stoichiometry. This may lead to P accumulation in soil and water eutrophication after the leaching of nutrients from the soil.
In the legal sense, the animal by-products regulation (EC/1069/2009) classifies manure as an animal by-product of category 2. According to this regulation, **manure can be applied on land without further processing**. Possible manure **use restrictions** are, however, defined in other EU legislation.

The **Water Framework Directive** (WFD, 2000/60/EC) establishes an integrated and co-ordinated framework for the sustainable management of water, including prevention of deterioration of water bodies (lakes, rivers, coastal and transitional, groundwater), promotion of sustainable water use, and ensuring “enhanced protection and improvement of the aquatic environment”. The **Nitrates Directives** (91/676/EEC) indirectly limits the return of P to land by limiting the amount of N in manure that can be applied on land. However, the stoichiometric N/P ratios documented for soil microbes and plants (around 6 - 8; Cleveland and Liptzin, 2007) are higher than the N/P ratios of most types of manure (on average 4.4, Table 2). 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 framework 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 (MBA, 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 residuals 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 2). In all Member States, with the exception of Latvia, the potential P surplus between 2010 and 2014 was lower than between 2000 and 2004.

Figure 2: 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. At current, 7.8% of the manure (108 Mt, 556 kt N, 139 kt P) is 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) (Figure 3).
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 (polyacrylamide) have adverse environmental effect. Their use also has a strong effect on the further treatment potential of the solid and liquid fraction (Schoumans et al., 2010), although new technologies are emerging. Usual coagulants and flocculants are polyelectrolytes, aluminum and iron sulphate, aluminum and iron chloride, calcium oxides and calcium hydroxides, and also magnesium oxide and magnesium hydroxides. In case of targeting P-recovery in agriculture, the use of metal-containing salts might potentially impact upon the adeptness for P-recycling.

Recovery of biogas during anaerobic digestion is advantageous as it can be used for electric energy generation or for the heating and drying during the further processing of manure. Additionally, anaerobic digestion increases the dewatering properties of the slurry as more components end up in the solid fraction following separation of the slurry, and the drying of the solid fraction proceeds more rapidly (Schoumans et al., 2010). The solid fraction can then be dried before pelleting 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.,...
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 recovered phosphate salts (from pig manure). Different companies in the Netherlands and the UK (e.g. BMC Moerdijk, Fibrophos, etc.) produce poultry litter ashes from a combination of slaughtered animals and poultry beds. Finally, Hitachi-Zosen and the BioEcoSim project are evaluating the possibility of establishing a poultry manure pyrolysis facility in central Europe based on experience obtained from their pilot plant in Japan.

3.4 Animal by-products other than manure

3.4.1 Food waste from households, retail and food services

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

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

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

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

In the legal sense, food waste is treated in the same way as normal waste that is non-hazardous if and only if it does not exhibit any properties that may render it hazardous. Products of animal origin, or foodstuffs containing products of animal origin, and catering waste are, however, classified as category 3 Animal By-products. Stringent controls are applied to its transport, handling and storage, treatment and disposal through Animal By-Products Regulation (EC) No 1069/2009. Therefore, a share of the food waste is processed together with slaughter residues by the rendering industry (see paragraph 3.4.4).

3.4.2 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, 2006c). Post-harvest loss occurs during pre-processing, processing, storage and transportation of fishery and aquaculture products. Post-harvest fish losses are, for instance, caused by fish scrap generation during fileting, curing and smoking processes and discarding of bycatch at sea because fish is too small or not valuable enough to bring to land for sale. At the global scale, up to 70% of total processed raw fish (on weight basis) ends up as solid waste in processing plants (UNEP, 2000).

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

Legally, residues from the fish industry are classified as animal by-products and derived
products. The materials show similarity to animal by-products from the meat and dairy
industry, for which reason P-recovery in the form of recovered phosphates, ashes and
pyrolysis 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 3.4.3 and 3.4.4).

3.4.3 Materials from the meat industries

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

Slaughter residues are classified with other animal by-products according to Regulation
(EC) 1069/2009. Slaughter residues are animal derived, and can contain different bacterial,
viral, prion and parasitic pathogens and cannot be discharged into the environment without
proper treatment. Therefore, the use of unprocessed slaughter residues for animal feed has
been banned in the EU since 2000 due to fear of Bovine Spongiform Encephalopathy (BSE).
Determinate category-specific product treatment options should now be undertaken prior to
further utilization (Table 3). 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 (EFPR, 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 4. 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 3: Classification of substrates according to Regulation (EC) 1069/2009, and their respective main treatment routes (adopted from Moller, 2015)

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

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 4) (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 4). A total amount of 14.5 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 4: Overview of the slaughtered animals and the amounts of slaughter residues generated in EU-28 for the year 2005

<table>
<thead>
<tr>
<th></th>
<th>absolute weight of slaughtered animals(^{a,b}) (kt yr(^{-1}))</th>
<th>inedible fraction(^{c,d}) (-)</th>
<th>slaughter refuse (kt yr(^{-1}))</th>
<th>relative contribution (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>bovine meat</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>calf</td>
<td>946</td>
<td>0.42</td>
<td>3417</td>
<td>25</td>
</tr>
<tr>
<td>adult cattle</td>
<td>6819</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>pigmeat</td>
<td>21781</td>
<td>0.34</td>
<td>7405</td>
<td>53</td>
</tr>
<tr>
<td>sheep and goat meat</td>
<td>809</td>
<td>0.48</td>
<td>388</td>
<td>3</td>
</tr>
<tr>
<td>poultry meat</td>
<td>10797</td>
<td>0.25</td>
<td>2699</td>
<td>19</td>
</tr>
<tr>
<td>overall sum</td>
<td>49289</td>
<td>13910</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

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

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

3.4.4 Rendering industry

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

The 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 4.

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 4). Category 2 material is mainly used for the production of biodiesel (fat fraction), and fertilisers (protein fraction) (Dobbelare, 2017) (Figure 4). About 180 kt of category 2 protein material is used for the production of (organic) fertilisers in the year 2016 (Dobbelare, 2017). The fat fraction of Category 3 animal by-products are typically processed to produce Processed Animal Proteins (PAP), that can be used for pet food, animal feed including and fish feed, oleochemicals, edible fats and biodiesel (EFPRA, 2017). The protein fraction for category 3 material equalled 2.7 Mt of material in 2016, and was dominantly used for the production of pet food (~70%), (organic) fertilisers (18%), and to a smaller extent for fish feed, human food (gelatin) and fur feed (EFPRA, 2017) (Figure 4). 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 pant 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 4: Potential application of processed animal by-products of category 1, 2 and 3 (source: EFPRA (2017))

Animal by-products other than manure and derived products not intended for human consumption are currently already used for the manufacturing of STRUBIAS materials. The mono-incineration of 1 tonne of animal derived meal and grist generates about 100-300 kg of ash (Coutand et al., 2008). During the incineration process, all organic matter in the material, including proteins, is transformed to CO$_2$, H$_2$O and nitrous and sulphur oxides, etc. Minerals like Ca, Mg and P are relatively stable in response to heating (Deydier et al., 2005; 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., 2005; Wopenka and Pasteris, 2005; Czaja and Hermann, 2011). The combustion induces a wide range of structural modifications, such as crystalization of calcium phosphate, substitution reactions, etc. These processes reduce the P-solubility and therefore the value of the meat and bone meal ashes as P fertiliser in comparison to the original substrate (Moller, 2015). At present, however, co-incineration is the dominant thermochemical pathway due to the ease of operation and increased energy revenues. Mono-incineration is only applied at specific facilities (e.g. Kalfos – SARIA) that aim at producing meat and bone meal ashes for use as a fertilising material. 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.
3.5 Effluents and residues from municipal waste water treatment plants

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

Municipal sewage is a water-carried waste that is intended to be removed from a community. It consists mostly of greywater (from sinks, tubs, showers, dishwashers, and clothes washers), blackwater (the water used to flush toilets, combined with the human waste that it flushes away); soaps and detergents; and toilet paper. The Urban Waste Water Treatment Directive (91/271/EEC) defines an agglomeration as an area where the population and/or economic activities are sufficiently concentrated for urban waste water to be collected and conducted to an urban waste water treatment plant or to a final discharge point. The size of an agglomeration in terms of generated pollution load is measured in “population equivalent” (p.e.). This is the organic biodegradable load that has a five-day biochemical oxygen demand (BOD5) of 60 g of oxygen per day, or in more popular terms – the organic biodegradable load generated by one person per day.

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

The main objective of the Urban Waste Water Treatment Directive (91/271/EEC) and national legislation for non-EU countries is to protect surface waters from the adverse effects of wastewater discharges. This is achieved through the requirement for collection and treatment of wastewater in all settlements (agglomerations) and areas of economic activity with a population equivalent (p.e.) larger than 2000. The connection rate in Central European countries is even higher, and exceeds 90%. About 80% of the population is connected to waste water treatment in Northern and Southern European countries. On the basis of data reported in 2010, about 67% of the total population is connected to wastewater treatment in the countries of Eastern Europe. Advanced (secondary or tertiary) treatments for nutrient and organic matter removal are required for populations larger than 10 000 p.e. The current population connected to plants with tertiary treatment is in the order of 70% in
Northern and Central Europe, and about 50% for Southern and Eastern Europe (European Environment Agency, 2013b). Average connection rate in South-East Europe (Turkey, Bulgaria and Romania) is about 40%. About a quarter of the population in South Eastern countries is connected to collecting systems without treatment (European Environment Agency, 2013b).

Table 5 shows the generated organic pollution load that agglomerations discharge as a function of size class (European Commission, 2017b).

**Table 5: The generated organic pollution load of urban waste water treatment agglomerations as a function of size class (European Commission, 2017b)**

<table>
<thead>
<tr>
<th>Total load discharged from agglomerations</th>
<th>Total load discharged from agglomerations</th>
<th>Total load discharged from big cities discharging &gt;150 000 pe (million p.e.)</th>
</tr>
</thead>
<tbody>
<tr>
<td>EU15</td>
<td>51</td>
<td>457</td>
</tr>
<tr>
<td>EU13</td>
<td>17</td>
<td>62</td>
</tr>
<tr>
<td>EU28</td>
<td>68</td>
<td>519</td>
</tr>
</tbody>
</table>

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

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

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

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

Table 6: Configurations used in enhanced phosphorus removal methods used in selected EU Member States (adopted from Wilfert et al., 2015)

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

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

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

The current fate of treated sludge includes disposal in landfills, application to land, incineration, or composting (Figure 5). 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.
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 5). 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 5). 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 5).

Although the Sewage Sludge Directive (91/271/EEC) has established the conditions for safe use of sludge on agricultural land, but concerns exist, notably as regards the maximum limit values for cadmium and other contaminants are considered to be too high (European Commission, 2013a). Sixteen Member States have adopted more stringent standards than those set out in the Directive. Therefore, direct sewage sludge application on agricultural land is progressively decreasing in Europe. Harmonisation of higher quality standards would encourage greater confidence amongst farmers and consumers on the safe use of sludge in the EU. In order to encourage more efficient resource use in the future, these issues will need to be addressed so that standards for sewage sludge-derived products inspire confidence right across the chain of end users: namely farmers, retailers and ultimately consumers.
A detailed description of the sewage sludge mono-incineration process is given in Donatello and Cheeseman (2013). Sludge and hot compressed air (ca. 500–600°C) are fed to the combustion chamber. The sand bed temperature is typically 750°C and the overhead freeboard zone at 800–900°C. Temperatures can be finely controlled by the injection of water or liquefied gas oil. The sand bed acts as a “thermal fly wheel” and helps stabilise temperature fluctuations in the incinerator. Particle residence times in the combustion chamber are typically only 1–2 s and during this time water is evaporated, volatile metals vapourise and organic compounds are combusted completely to gases, either directly or via the formation of an intermediate char. The remaining inorganic material is carried out of the chamber as fine particulates with the exhaust gases. During incineration, most of the N is released into the atmosphere while much of the P and K, and heavy metals are retained in the fly ash (Deydier et al., 2005; Zheng et al., 2013). Sulphur (S) is retained in the air pollution system, and can possibly be recycled as Na₂SO₄ from the alkaline scrubber. The fly ash is generally removed by bag filters, electrostatic precipitators or cyclones after passing through a heat exchanger. The flue gas is then treated using a wet scrubber with acid, alkali and possibly activated carbon dosing to comply with emission limits, as required by Industrial Emissions Directive (2010/75/EU). The scrubbing process produces an additional waste sludge, which is dewatered and normally disposed of in hazardous waste landfill. Mono-incineration produces fly ash with high P contents (2-12%). The average P content in sewage sludge ashes is 10.8 % in the Netherlands (CBS Statistics Netherlands, 2015) and 8.9% in Germany (Krüger and Adam, 2015). In Germany, about half of the generated municipal sewage sludge is currently already processed within mono-incineration plants (Adam et al., 2015).

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

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

STRUBIAS materials can be produced from (processed) wastewaters and sludges at municipal waste water treatments plants, as well from the incinerated sewage sludges (see section 4).
3.6 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 recovered phosphate salts 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 recovered phosphate salts, eventually after anaerobic digestion of the residues.

3.6.1 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$_2$H$_2$P$_2$O$_7$) is used to complex iron (Fe$^{2+}$). In this way sodium acid pyrophosphate prevents that iron in the potato reacts with chlorogenic acid during the heating processes (Rossell, 2001). The oxidation of the Fe$^{2+}$-chlorogenic acid complex by oxygen from the air would otherwise result into a grayish-colored substance that causes the after-cooking gray discoloration (Rossell, 2001). This is a very significant P-source in the waste water from potato processing installations. The waste water have a typical PO$_4^{3-}$ 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 Belgium and the Netherlands (Dewaele, 2015).

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

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

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

Typical **cleaning agents** used in the food-processing industry sector are (European commission, 2006b):

- 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 $\text{PO}_4^{3-}$ concentration in the waste water varies between 20 and 200 mg L$^{-1}$ (European commission, 2006b). 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 ($\text{PO}_4^{3-}$ P concentration in the waste water: 60 – 65 mg P L$^{-1}$; P-recovery efficiency of about 75%).

In the **brewery**, waste water from the anaerobic reactor is driven to a reservoir where it is mixed with aerobic sludge (from the second sedimentation) and with untreated neutralised waste water (taken before anaerobic digestion). The $\text{PO}_4^{3-}$ concentration in the waste water varies between 0 and 200 mg L$^{-1}$ (Gendebien et al., 2001). All these flows are recirculating and, in these conditions, the aerobic sludge encourages the growth of P-assimilating bacteria. Phosphorus could then be recovered after the bacterial release of orthophosphates.

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

### 3.7 Chemical industry waste waters

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

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

At present, Genzyme bvba makes use of a struvite reactor for P-recovery in the form of recovered phosphate salts 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).

3.8 Iron and steel industry residues

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 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. The slags are often referred to by the type of furnace: blast furnace slag, blast oxygen furnace slags and electric-furnace slags. 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 (electric arc furnace steelmaking) to 400 kg (basic oxygen steelmaking) of residues. These include slags, dusts, sludges and other materials.

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

Slags that have undergone a thermal oxidation at high temperatures could be considered as ash-based materials in the STRUBIAS project. 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.

3.9 Forest-based industry residues and green waste

3.9.1 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 incinerated without energy recovery.

- Forests and other wooded land occupy over 44% of the EU's surface and represent 5% of the world's forests. In the last 50 years, both their area and the standing timber volume (growing stock) have continued to grow. Nowadays, they gain almost 700,000 ha annually. According to Searle and Malins (2013), it was estimated that the total production of forestry residues in the EU was 80.7 million tonnes in 2011. Some forestry residues are currently collected, but according to ECF (European Climate Foundation, 2013), the current usage of forestry residues in the EU is only about 3%, with activities mainly occurring in Scandinavia. Similar to crop residues, a share of the forestry residues should remain on land to protect soil carbon and sustainable
ecosystem functioning. To be conservative and to avoid other unintended consequences, Searle and Malins (2013) assumed that 50% removal of forestry residues may be sustainable if combined with good management practices. Assuming these values, a total of about 40 million tonnes of uncollected forestry residues might potentially be available for nutrient recovery.

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

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

Bark and wood residues from wood handling is normally incinerated 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 it is not contaminated, e.g. by metals like Hg, Cd and Pb. 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 (mainly bottom ashes) 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 materials that are used as soil improver (EBC, 2012).
For a complete overview of the processing of woody material and recovered materials, it is referred to the Best Available Techniques (BAT) Reference Document for the Production of Pulp, Paper and Board (European Commission, 2015a).

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

All pulp and paper sludge comprises a mixture of cellulose fibre (40 to 60% of dry solids), 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 because legislation that constrains the metal/metalloid content of the ink has significantly. 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 producing processes (Kraft pulping process, sulphite pulping process, mechanical and chemimechanical pulping process, and processing of paper from recycling) is a commonly applied process in the sector. The use of auxiliary fuel may be necessary to maintain good burning conditions unless the sludge is mixed with bark and other wood waste material. Burning reduces the volume of waste and the inorganic content remains as ash. It is noted that specific effluents cannot be sent for incineration as a consequence of the use of certain chemicals (e.g. chlorine dioxide).

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

3.9.3 Emissions to water

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

3.9.4 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.

3.10 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).

3.11 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 subgroup did not identify waste materials, industrial residues or biological materials other than those mentioned in sections 3.2 - 3.9.

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.

3.12 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 waste streams as input material for fertilisers derived from secondary raw materials. The overview presented in this document builds further upon the information presented by the scientific community, multi-stakeholder research platforms, think tanks and international and non-governmental organizations (Sutton et al., 2011; Leip et al., 2014; Buckwell and Nadeu, 2016; van Dijk et al., 2016), complemented by own estimates as described in each of the input material sections.

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

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

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, poultry litter ashes, forest-based industry residues, iron and steel industry residues.

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

4.1 Recovered phosphate salts

4.1.1 Waste water treatment plants

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

Phosphate salts can be recovered from sludge liquor and from digested sludge when the PO$_4^{3-}$ precipitates together with Mg$^{2+}$ or Ca$^{2+}$, possibly also trapping NH$_4^+$ and/or K$^+$ in the molecular structure. Struvite, the most commonly recovered phosphate salt, forms from equimolar quantities of Mg$^{2+}$, PO$_4^{3-}$ and NH$_4^+$ implying that the efficiency of NH$_4^+$ removal is relatively low and the excess N remains in soluble form. In most sewage treatment applications Mg is the limiting element, for which it is added to the process as MgCl$_2$ or MgO. The formation of precipitates is strongly influenced by pH, hence if the feed stream does not have sufficient alkalinity, NaOH is added and/or CO$_2$ is stripped from the solution. More detailed information on the crystallization dynamics and kinetics for the struvite crystallization process is given in Le Corre et al. (2009).

Phosphorus recovery from sludge liquor and from digested sludge is limited to the amount of soluble PO$_4^{3-}$. For most waste water treatment plants, the latter is in the range of 5-20% of total P load of the sludge under normal pH conditions (Jossa and Remy, 2015). P content of the solid phase can be substantially mobilized into the liquid phase by acidification (addition of acids or through the generation of carbonic acid from pressurised CO$_2$) or thermal hydrolysis so that total P recovery rates of up to 50% seem feasible in both pathways.

Recovery in the form of phosphate salts without acidification treatment 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 an acidification step are in principal able to deal with sludge that had been subjected to chemical removal and coagulation with Al and Fe salts. Nevertheless, when ChemP sludge is used as input material, the P-recovery rates are...
reduced or require substantially larger additions of chemicals used in the process (Kabbe et al., 2015).

Four types of recovered phosphate forming processes are considered: (I) from the sludge liquor, (II) from the digested sludge, (III) downstream from the digested sludge after a pH regulating treatment, (IV) upstream from the digested sludge with thermal hydrolysis. The 4 types differ in their P-recovery rate, sludge input materials that be used, and energy and chemical demand (Table 7). For phosphate salt formation from the liquor (type I), the concentrated side streams after the anaerobic treatment or the dewatering unit after anaerobic digestion are the best options for P recovery. The implementation of a P-recovery system before the anaerobic digester (type II and type IV) reduces uncontrolled P precipitation in the anaerobic digester or post-digestion processes and enhances P recovery (Martì 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 7: Overview of the principles and properties of recovered phosphate salt producing processes from waste water treatment plants (adapted from Jossa and Remy (2015)).

<table>
<thead>
<tr>
<th></th>
<th>type I: liquor precipitation</th>
<th>type II: sludge precipitation</th>
<th>type III: downstream sludge</th>
<th>type IV: upstream sludge</th>
</tr>
</thead>
<tbody>
<tr>
<td>liquor precipitation</td>
<td>mostly operating, some piloting</td>
<td>operating (Seaborne), piloting (Stuttgarter)</td>
<td>operating</td>
<td>operating</td>
</tr>
<tr>
<td>sludge precipitation</td>
<td>low to moderate (- 11-12%)</td>
<td>low (- 7%)</td>
<td>high (- 45-49%)</td>
<td>moderate to high (no exact data)</td>
</tr>
<tr>
<td>input material</td>
<td>sludge liquor from EBPR</td>
<td>digested sludge from EBPR</td>
<td>digested sludge from EBPR and ChemP</td>
<td>digested sludge from EBPR</td>
</tr>
<tr>
<td>phosphorus recovery</td>
<td>low to moderate (- 11-12%)</td>
<td>low (- 7%)</td>
<td>high (- 45-49%)</td>
<td>moderate to high (no exact data)</td>
</tr>
<tr>
<td>chemical demand</td>
<td>low</td>
<td>moderate</td>
<td>high</td>
<td>high</td>
</tr>
<tr>
<td>energy demand</td>
<td>low</td>
<td>yes</td>
<td>high</td>
<td>high</td>
</tr>
<tr>
<td>increased sludge dewaterability</td>
<td>no</td>
<td>yes</td>
<td>no</td>
<td>yes</td>
</tr>
</tbody>
</table>

Type I: liquor precipitation. Recovered phosphates, mostly struvites, can be formed from the sludge liquor in mixed stirred tanks. The PHOSPAQ® and ANPHOS® processes operate in a single and two separate stirred tank reactors. An increase in pH (CO₂ 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 (MgCl₂) 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 (NH₄)₂SO₄ 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 I**: sludge precipitation. The AirPrex® process is different from the abovementioned techniques as crystallization of the recovered phosphate salt occurs directly from the digested sludge. A major advantage is the improvement of sludge dewatering. 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). Also the Ostara’s Pearl process can be combined with the Waste Activated Sludge STRIPping process WASSTRIP (Baur, 2009). Here, the digested sludge is sent to the anaerobic reactor where P and Mg are released (stripped) by the micro-organisms as a consequence of endogenous respiration and fermentation, after which the resultant P-rich liquid is sent to the precipitation reactor.

- **Type III**: downstream sludge leaching after acidification. A wet-chemical extraction process to process digested sludge from waste water treatment plants. These processes can use sludges produced in biological (EBPR) and chemical (precipitation with metal salts) waste water treatment processes, although the chemical 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 H₂SO₄ 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 Na₂S has been introduced in order to avoid that Fe phosphate compounds with low plant availability are transferred to the nutrient product. 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 Seabome process, the P as struvite, the N for just under a third in struvite and the remainder in (NH₄)₂SO₄ (Günther et al., 2007).

The Stuttgarter® process relies on the same principle of wet chemical treatment but differs from the Seabome 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).

The 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.

- Type IV: upstream sludge hydrolysis. Instead of releasing phosphorus by acidification from digested sludge by applying chemicals, thermal hydrolysis at temperatures between 150-200°C could be applied on secondary sludge from waste water treatment plants applying EBPR. This releases P to a soluble form so that higher recovery rates can be achieved by phosphate precipitation, and also improves CH₄ production. The digested sludge firstly is thickened and then subjected to hydrolysis and digestion. Digested sludge has a very high water
absorbing capacity, greatly reducing the degree of dewatering in proportion to its share. Through thermal hydrolysis in a pre-treatment step, poorly degradable substrates such as proteins and polysaccharides are modified such that microorganisms can easily degrade them. Hence, by deploying this procedure prior to primary sedimentation, the good degradability and dewaterability rates of the sludge liquor is increased (Ewert et al., 2014). Processes include thermal hydrolysis (e.g. Cambi, Eliquo Stulz (Lysogest), Exelys) or thermo-chemical hydrolysis (e.g. Pondus). Thermal hydrolysis of the digested sludge upstream of primary sedimentation is then followed by P recovery from the fully digested sludge (e.g. AirPrex® procedure) (Ewert et al., 2014).

There are specific processes (PASCH®, P-bac (INOCRE)®) that produce struvite from ashes as input material. These products will be described in section 4.2 (ash-based products).

4.1.2 Precipitation from other input materials

Most of the techniques that recover P in the form of phosphate salts (struvite, dicalcium phosphates, or a mixture of Ca- and Mg-salts) are developed for municipal wastewater (Desmidt et al., 2015). P-recovery techniques based on precipitation techniques can apply in principle to all phosphate rich liquids or slurries. Therefore, the techniques can also be applied on phosphate-rich industrial waste water (e.g. potato industry, dairy industry, type I processes) and anaerobically digested biowaste and manure fractions (mostly type I applications). Although at present only applied municipal waters, 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).

4.1.3 Deliberateness of the nutrient recovery

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

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

- Phosphate salt producing processes of Type II and IV may increase the dewaterability of the sludge, and thus the associated costs of sludge disposal and chemical demand associated to traditional sludge dewatering options (e.g. addition of flocculation agents, acid and alkaline, etc.). At present, operating costs for sludge dewatering usually account for as high as 25–50% of the total expenses of the whole wastewater treatment processes (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 cation creates bridges between particles whereas monovalent cations tend to deteriorate flock structures. Therefore, an improved dewaterability can be expected if the addition of magnesium divalent cations surpasses the effect of sodium hydroxide dosing. Marchi et al. (2015) indicated the importance of a proper tuning of chemical additions in order to achieve progressive dewatering.

- Waste water treatment costs are also reduced by the lower maintenance costs due to the avoided pipe clogging and abrasion of centrifuges.
- The reduction of the P and N load of the sludge liquor has a direct effect on the treatment capacity of the whole waste water treatment plant as well as a cost factor, since the removal of nutrients from the wastewater requires energy, chemicals and tank volume (Ewert et al., 2014).

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

### 4.2 Ash-based products

Whether ashes as obtained after the incineration 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.

#### 4.2.1 Raw ashes and melting/sintering materials

##### 4.2.1.1 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 incineration 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 950°C, being 850°C a usual bed temperature.

- **Circulating Fluidized Bed (CFB)** technology boilers are normally used in larger applications, being similar in basic concept to the BFB. CFB has enhanced flexibility over BFBs for firing multi-fuels with high moisture content and significantly higher efficiency up to 95%. CFB configuration includes solid separators that separate the entrained particles from the flue gas stream and recycles them to the lower furnace. The collected particles are returned to the furnace via the loop seal. The addition of the solid separators allows CFB technology to reach the higher values regarding efficiency and availability and provides fuel flexibility. The entrainment velocity is the limit point that defines the transition from a BFB to a CFB. The CFB operation range is fixed over that entrainment velocity. Beyond this velocity the bed material becomes entrained and the solids are distributed throughout the furnace with a gradually decreasing density from the bottom to the top of the furnace. Fluidizing velocity is higher than in a BFB and can be between 4.5-6.7 m/s.

- Similar to BFB, **grate boilers** are used in units below 100 MWe and normally for industrial uses. Grate technology can burn a range of fuels wider than a BFB, but worse emissions and efficiency as BFB. Grate boiler provides very good performance burning low moisture and high alkaline content fuels. Grate can burn difficult fuels as straw, little chicken, 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 (Na₂CO₃) 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 PO₄³⁻ ionic group by SiO₂²⁻ or CO₃²⁻) affecting the reactivity of the final product and therefore the plant P availability. Metals/metalloids are partially volatilised (Zn, Cg, 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 incineration ashes to improve the material quality (see section 4.2.1).

**Also steelmaking** processes make use thermal oxidation melting processes in blast furnaces and 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 oxidation 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 mineral ores 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 4.2.2).

4.2.1.2 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, 2006a). 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, 2006a). The FGT technology impacts upon the quality of the combustion residues.
4.2.2 Ash derivates

The use of **raw ashes** in agriculture can be associated to two major issues (Chandrajith and Dissanayake, 2009; Herzel et al., 2016). At first, the conditions during incineration induce a wide range of structural modifications that **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 8). 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

  o An almost complete **acidic dissolution of P at pH-values below 2** through the addition of chemicals is the principle of action to recover P 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\(_2\)SO\(_4\) or HCl). Thus, after acidic leaching the toxic inorganic contaminants (e.g. Pb, Cd, Hg, etc.) have to be separated from the dissolved P in order to create a valuable P-recovery product. Additionally, it is desirable to separate especially Al and Fe as well, as these elements impair the quality 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 8).

  ▪ 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 phosphoric acid that can then be used for the production of traditional P-fertilisers (e.g. DAP, MAP, TSP, nitrophosphate, etc.). The P-rich ashes should be consistent and the Fe/Al content should be relatively low in order to enable the partial substitution of phosphate rock by ashes in the process (Langeveld and Ten Wolde, 2013).

  ▪ The basis of the **SEPHOS process** is the **sequential precipitation** of P complexes with an alkaline treatment (Takahashi et al., 2001; Schaum, 2007). The separation of dissolved P from heavy metals is achieved by raising the pH-value in the acidic leachate to induce the precipitation of Al-P while most heavy metals remain in solution (Takahashi et al., 2001). The heavy metal content of the Al-P product is then further decreased by precipitating heavy metals with sulphide (Schaum, 2007). Since the entire P has to be precipitated as Al-P, this process is especially suitable for Al-rich ashes coming from waste water treatment plants that employ chemical P-removal by addition of Al-salts. Since Al-P cannot be directly reused as fertiliser, the precipitated Al-P may be dissolved by alkaline treatment followed by precipitation as Ca-P. Altogether, this type of wet chemical P-recovery process results in a total chemical demand (at least 600 g H\(_2\)SO\(_4\) /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 H₂SO₄ that dissolves about 70-90 % of the P in the ashes, depending on acid concentration and reaction time. The leaching is followed by a solid/liquid separation step carried out on a vacuum belt filter or in a filter press. The leached sewage sludge ash filter cake is withdrawn from the process and must be disposed. The P containing liquid is pumped into a second stirred reactor, where dissolved P is precipitated by dosing of lime (CaO) or caustic soda (NaOH). A product with relatively high P content (13% P), considerable metal depletion and sufficient dewaterability is thereby produced (Adam et al., 2015). Depending on the precipitation agent, P is present in different mineral phases. If precipitated mainly with lime, P is present in the form of calcium phosphate next to aluminium phosphate. After precipitation and separation of the phosphorus product, the liquid waste stream requires additional treatment. Treatment consists of pH elevation to a pH of 9 by dosing of additional lime and of sulphidic precipitation of metals by an organosulphide precipitation agent (Adam et al., 2015). This is carried out in a third reactor followed by an additional solid/liquid separation step by a filter press. Thus the metals in the wastewater are removed almost completely. Thereafter, the pH in the waste water is adjusted to a pH of 7 and is discharged either to a waste water treatment plant or directly to a receiving water body (Adam et al., 2015).

- The PASCH® (Phosphorus recovery from Ash, developed at Aachen University) process utilizes liquid-liquid extraction for heavy metal and iron separation (Nieminen, 2010; Pinnekamp et al., 2010). Different acids were tested for P-dissolution by Montag and Pinnekamp (2009) with results of 25%, 50%, 80% and 90%, for NaOH, H₃PO₃, H₂SO₄ 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$_3$PO$_4$ 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$_2$SO$_4$ 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$_2$SO$_4$ 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$_3$PO$_4$ (Lundin et al., 2004). It should be emphasized that H$_3$PO$_4$ 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$_3$PO$_4$ 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$_3$PO$_4$ for fertiliser or food and feed industry is produced. At the same time most of the produced residues are sellable products as CaCl$_2$, gypsum, silicate as well as iron- and aluminum chlorides. With the TetraPhos® process, Remondis developed and implemented a similar approach in pilot scale (Hamburg) using H$_3$PO$_4$ instead of HCl to leach the ash, and ending up with H$_3$PO$_4$ as a final product (Remondis Aqua, 2016). 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$_3$PO$_4$. 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$_3$PO$_4$.

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 biodegradation 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 “biodegradation 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$_2$SO$_4$**, 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 8). 
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$_2$O$_5$ is 
converted to plant-available Ca-Si-phosphate, similar to the phosphate in 
Thomas ground basic slag. The oxidation of residual metallic iron and 
bivalent iron in the steel slag produces the energy to maintain the
necessary process temperatures of around 1500°C. The process has been tested in laboratory and in industrial scale.

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

The ASH DEC process (Outotec) treats mono-incinerated sewage sludge ashes by a sodium sulphate dosage and thermal treatment below the melting point of sewage sludge ash in order to remove heavy metals making the product suitable for agricultural use by increasing the plant availability of P. An ASH DEC plant could stand alone and being operated as greenfield facility. For economic and ecological reasons it is planned to combine the ASH DEC plant with mono-incineration. The main advantage of the combination is the possibility of feeding hot ash directly from the mono-incineration plant to the ASH DEC facility, thus saving energy and equipment (Adam et al., 2015). In the first step, the ash is mixed with Na₂SO₄ in the thermally treated ash. Fresh Na₂SO₄ input could be partly replaced by recycled Na₂SO₄ from the mono-incineration. Alternatively (older process), MgCl₂ 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 Na₂SO₄ 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. 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 MgCl₂. 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 partly to a vast 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 Woehler 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 P₂O₅ or converted into H₃PO₄. The process enables the use of waste materials as heat sources, reducing agents or additives, including dried sewage sludge, foundry ash, waste salts or meat and bone meal, and low grade phosphate rock, with as advantage that in the RecoPhos process no pre-agglomeration of powder feedstock is needed and no dioxin emissions takes place. The Recophos process also claims to be able to recover P from raw materials containing significant levels of Fe from ChemP waste water treatment plants. The end product P₄ can then be used for production of flame retardants or lubrication additives while H₃PO₄ 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 8: Overview of the principles and properties of P-recovery processes from ashes.

<table>
<thead>
<tr>
<th>process name</th>
<th>status</th>
<th>route</th>
<th>P-separation</th>
<th>products</th>
<th>P-recovery</th>
<th>ref.</th>
</tr>
</thead>
<tbody>
<tr>
<td>SEPHOS</td>
<td>unknown</td>
<td>wet-chemical</td>
<td>sequential precipitation (strong acid, alkalines)</td>
<td>calcium phosphates</td>
<td>90%</td>
<td>(a)</td>
</tr>
<tr>
<td>SESAL-Phos</td>
<td>piloting at laboratory scale</td>
<td>wet-chemical</td>
<td>sequential precipitation (mild acid, alkalines)</td>
<td>calcium phosphates</td>
<td>74-78%</td>
<td>(b)</td>
</tr>
<tr>
<td>LeachPhos</td>
<td>piloting/planned</td>
<td>wet-chemical</td>
<td>sequential precipitation (strong acid, alkalines)</td>
<td>calcium phosphate, aluminium phosphate</td>
<td>70-90%</td>
<td>(c)</td>
</tr>
<tr>
<td>PASCH</td>
<td>unknown</td>
<td>wet-chemical</td>
<td>liquid-liquid extraction</td>
<td>calcium phosphate, struvite</td>
<td>90%</td>
<td>(d)</td>
</tr>
<tr>
<td>BioCon</td>
<td>unknown</td>
<td>wet-chemical</td>
<td>ion exchange</td>
<td>H$_3$PO$_4$</td>
<td>60%</td>
<td>(e)</td>
</tr>
<tr>
<td>EcoPhos</td>
<td>piloting/constructing/operating</td>
<td>wet-chemical</td>
<td>confidential, including ion exchange</td>
<td>H$_3$PO$_4$, dicalciumphosphate</td>
<td>97%</td>
<td>(f)</td>
</tr>
<tr>
<td>TetraPhos</td>
<td>piloting</td>
<td>wet-chemical</td>
<td>confidential</td>
<td>H$_3$PO$_4$</td>
<td>unknown</td>
<td>(g)</td>
</tr>
<tr>
<td>RecoPhos (P 38)</td>
<td>operating</td>
<td>wet-chemical</td>
<td>confidential</td>
<td>mostly calcium and magnesium phosphates</td>
<td>90%</td>
<td>(h)</td>
</tr>
<tr>
<td>Edask</td>
<td>piloting</td>
<td>wet-chemical</td>
<td>ion exchange (electrolysis)</td>
<td>H$_3$PO$_4$</td>
<td>unknown</td>
<td>(i)</td>
</tr>
<tr>
<td>P-bac</td>
<td>piloting</td>
<td>wet-chemical</td>
<td>bioleaching</td>
<td>struvite</td>
<td>90%</td>
<td>(j)</td>
</tr>
<tr>
<td>Mehprec</td>
<td>piloting/constructing/planned</td>
<td>thermal</td>
<td>not applicable</td>
<td>P-rich slag (5-10% P)</td>
<td>80%</td>
<td>(k)</td>
</tr>
<tr>
<td>Kubota</td>
<td>operating (Japan)</td>
<td>thermal</td>
<td>not applicable</td>
<td>P-rich slag (~13% P)</td>
<td>&gt; 80%</td>
<td>(l)</td>
</tr>
<tr>
<td>Ash Dec</td>
<td>piloting/planned</td>
<td>thermal</td>
<td>not applicable</td>
<td>P-rich ashes (5-10% P)</td>
<td>98%</td>
<td>(m)</td>
</tr>
<tr>
<td>RecoPhos (ICL)</td>
<td>piloting/planned</td>
<td>thermal</td>
<td>not applicable</td>
<td>elemental P, P$_2$O$_5$ and H$_3$PO$_4$</td>
<td>89%</td>
<td>(n)</td>
</tr>
</tbody>
</table>

4.2.3 Deliberateness of the nutrient recovery

Thermal oxidation processes may be performed for sanitisation and volume reduction of waste-based materials. Also synergies with energy recovery is possible for organic materials, but the energy recovery potential in mono-incineration plants is largely dependent on the moisture content of the input materials. The chemical or thermal post-processing described above are mainly designed with the specific aim of P recovery and/or to improve material properties, and rely on the input of ashes from mono-incineration facilities. Finally, P-slags from the steel industry are produced as a residue from the steelmaking industry.

Ashes produced through co-incineration (incineration together with municipal solid waste or industrial waste) typically have lower P contents that pose increasing difficulties for nutrient recovery using current techniques. Such plants are targeted towards energy recovery and carbon abatement. The Mehprec and Ash Dec processes are currently the only pathways that integrate nutrient and energy recovery when treating sewage sludge material. Kalmykova and Fedje (2013) performed a pilot using ashes with a low P-content originating from municipal solid waste incineration using acidic leaching/precipitation and acidic/alkaline leaching. The processes indicated low P-recovery potentials due to the high Ca content of the ash (acidic/alkaline leaching method), meanwhile also the trace metal content of the obtained end product could limit the direct application of this product on agricultural land (acidic leaching/precipitation method).

4.3 Pyrolysis materials

4.3.1 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 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 bar over an aqueous solution of biomass for several hours, 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 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).

4.3.2 Spectrum of pyrolysis 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 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%. Mineral-rich input materials such as manure, and feedstock containing large amounts of animal bones, are much lower in organic C. Therefore, the European Biochar Certificate refers to pyrolysed organic matter with a C content lower than 50% as pyrogenic carbonaceous materials, instead of biochar. In the STRUBIAS framework, the name pyrolysis 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 materials.

4.3.3 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 volume 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 mono-
incineration plants is largely dependent on the moisture content of the input materials.

Pyrolysis 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 fertiliser, compost, plant protection);
- Use in the building sector (insulation, air decontamination, humidity regulation);
- The treatment of waste water (active carbon filter, pre-rinsing additive);
- The treatment of drinking water (micro filters);
- Other uses (exhaust filters, carbon fibers, semiconductors, etc.).
5 Agronomic efficiency

5.1 Introduction

Knowledge of the agronomic efficiency of STRUBIAS materials is the key starting point in any assessment of the following impacts:

- environment & human health impacts: the relative amount of STRUBIAS fertilisers needed to achieve the same agronomic objectives as with an alternative fertilising material will determine what will be the relative impact on the environment and human health from the production and use phases of any substituting STRUBIAS fertilisers;
- 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 was 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 establishing whether the scientific findings are consistent and generalisable across settings and facilitate understanding the reasons why some studies differ in their results. For these reasons, a meta-analysis of similar, well-conducted, randomized, controlled trials has been considered one of the highest levels of evidence (Garg et al., 2008).

5.2 P-fertilisers containing STRUBIAS materials

5.2.1 Meta-analysis approach

In the proposal for the Revised EU Fertiliser Regulation, mined and synthetic inorganic fertilisers are considered in the 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$_2$O$_5$ lower limit value of 12% is considered for the first category, whereas the second category requires a minimum P$_2$O$_5$ content of 3% P$_2$O$_5$ plus the presence of one of the other considered plant macronutrients (K$_2$O, MgO, N, CaO, SO$_3$, or Na$_2$O). Solid organic (> 15% organic C) and organo-mineral (> 7.5% organic C) P-fertilisers require a minimum P$_2$O$_5$ content of 2% (0.9% P). Therefore, this assessment focuses on STRUBIAS materials that have a minimum P$_2$O$_5$ content of > 2%.

The agronomic efficiency of fertilisers was assessed using two different plant response variables (Figure 7):

i. The plant dry matter yield (DMY): This is the most common response parameter documented in studies. Comparing the absolute values for DMY from $F_{\text{prim}}$ and $F_{\text{sec}}$
(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$_{applied}$):

\[
PUE = \frac{(PU_F - PU_C)}{P_{applied}} = \frac{\Delta PU}{P_{applied}}
\]

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 results of studies that documented a significant increase in plant uptake relative to control for P-fertilisers derived from phosphate rock were taken into consideration (see box 1).

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 7):

\[
RAE_{DMY} = \frac{DMY_{F_{sec}}}{DMY_{F_{prim}}}
\]

and

\[
RAE_{PUE} = \frac{PUE_{F_{sec}}}{PUE_{F_{prim}}} = \frac{\Delta PU_{F_{sec}}}{\Delta PU_{F_{prim}}}
\]

Note that the P application rate (P$_{applied}$) is levelled by dividing PUE$_{F_{sec}}$ by PUE$_{F_{prim}}$ to calculate RAE$_{PUE}$. 

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Figure 7: Schematic outline of the plant response variables used to calculate the relative agronomic efficiencies $\text{RAE}_{\text{DMY}}$ and $\text{RAE}_{\text{PUE}}$ following treatment with fertilisers from primary materials ($F_{\text{prim}}$) or secondary materials ($F_{\text{sec}}$).

A relative agronomic efficiency value below 1 indicates 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 fertilising effectiveness of $F_{\text{sec}}$ as a function of soil type, plant group, feedstock used for $F_{\text{sec}}$ production, and variables related to the experimental design of the study (e.g. fertiliser regime, pot versus field trial, etc.). These parameters that discern groups were referred to as grouping variables. The relative agronomic efficiency for both response variables ($\text{RAE}_{\text{DMY}}$ and $\text{RAE}_{\text{PUE}}$) 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, etc.) 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 sub-group and from scientific literature. The number of studies and "cases" for recovered phosphate salts, ash-based materials and pyrolysis materials is indicated in Table 9. A significant number of studies were available for recovered phosphate salts and ash-based materials, whereas data coverage for pyrolysis materials was poor. Therefore, no hard conclusions on the agronomic efficiency of pyrolysis materials could be made. Hence, the results for pyrolysis 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 is given in Box 1.

**Table 9: Number of studies and cases included for the meta-analyses on the relative agronomic efficiency of P-fertilisers derived from recovered phosphate salts, ash-based materials and pyrolysis materials.**

<table>
<thead>
<tr>
<th></th>
<th>recovered phosphate salts</th>
<th>ash-based materials</th>
<th>pyrolysis materials</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>RAE&lt;sub&gt;DMY&lt;/sub&gt;</strong></td>
<td>26</td>
<td>17</td>
<td>8</td>
</tr>
<tr>
<td><strong>RAE&lt;sub&gt;PUE&lt;/sub&gt;</strong></td>
<td>19</td>
<td>14</td>
<td>6</td>
</tr>
<tr>
<td><strong>cases</strong></td>
<td>173</td>
<td>117</td>
<td>31</td>
</tr>
<tr>
<td></td>
<td>104</td>
<td>94</td>
<td>16</td>
</tr>
</tbody>
</table>

**Box 1: Meta-analysis methodology**

**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-publicly 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 coordinates > 35°N/S). F<sub>prim</sub> 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:
Recovered phosphate salts: (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., 2014; Uysal et al., 2014; Bonvin et al., 2015; Cerrillo et al., 2015; Vogel et al., 2015; Wragge, 2015; Degryse et al., 2016; Hilt et al., 2016; Katanda et al., 2016; Liu et al., 2016; Sigurnjak et al., 2016; STOWA, 2016; Talboys et al., 2016; Vaneeckhaute et al., 2016).

Ash-based materials: (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 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).

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):

$$\ln R = \ln \text{RAE}$$

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.

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 ash-based materials, 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 materials as F\(_{sec}\), but no addition of N or K in P\(_{prim}\)).

Results 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 do not cross the vertical 1 line indicate that F\(_{sec}\) is not significantly different from F\(_{prim}\).

5.2.2 Recovered phosphate salts

The overall results indicated a similar agronomic efficiency for recovered phosphate salts to mined and synthetic P-fertilisers. The mean values of RAE\(_{DMY}\) and RAE\(_{PUE}\) equal 0.99 and 1.05, respectively (Figure 8), 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 recovered phosphate salts 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 \( \text{RAE}_{\text{DMY}} \) of calcium phosphates (grouping variable fertiliser) extended to a value marginally below 1 (0.995; Figure 8). No significant differences across selected groups were observed at the 95% level, albeit the effect of plant type was marginally significant (P: 0.06; data not shown) for \( \text{RAE}_{\text{DMY}} \).

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 crystallization 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.** Talboys et al. (2016) indicated that a complete dissolution of struvite, the most common precipitated phosphate salt, occurs in a relatively short term (<42 days) across a wider soil pH range of 5.0 – 8.0, a range relevant for most European soils. The rapid dissolution could also explain the observed non-significant effect of assessment time on RAE; the supplementary long-term effect of precipitated phosphate salts is thus not confirmed in this meta-analysis.
**Figure 8:** The relative agronomic efficiency of recovered phosphate salts for the plant response variables DMY (dry matter yield) and PUE (phosphorus use efficiency) as a function of grouping variables. Results are presented as weighted mean (square) and 95% confidence intervals (error bars).
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 marginally higher RAE_DMY observed for grasses than for cereal and oilseeds.

In line with the observation that feedstock does not have a major impact on the chemical composition of the recovered phosphate salts, no input material-specific impacts on RAE were observed.

5.2.3 Ash-based materials

The overall mean effects for ash-based materials were 0.92 and 0.81 for RAE_DMY and RAE_PUE, respectively (Figure 9). The 95% confidence intervals for both response variables indicated that the agronomic efficiency for ash-based materials was overall lower than for mined and synthetic fertilisers (Figure 9; 95% confidence intervals do not cross 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). Ash-based materials derived from sewage sludge showed a significantly lower RAE_DMY and RAE_PUE than ash-based materials derived from crop residues and poultry litter (Figure 9), but it should be deliberated that sewage sludge ash-based materials 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 9). The RAE_DMY values were 1.03 and 0.93 for materials subjected to wet-digestion and thermal post-processing steps (Figure 9). The RAE of F_sec derived from crop residues, poultry litter and pig manure did not differ from F_prim (Figure 9). Ash-based materials derived from wood and steel slags showed a low RAE_DMY and RAE_PUE, but the results should be interpreted with precaution because of the low number of cases (Figure 9). A significant effect of assessment time on RAE_DMY and RAE_PUE was observed (P < 0.001; Figure 9), 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 9). No significant effects of soil pH, soil texture, plant type and soil P status were observed, albeit the effect of soil pH on RAE_PUE was marginally significant (P: 0.08) (Figure 9).

Significant differences in the relative agronomic efficiency of ash-based materials were observed, primarily dependent on the feedstock applied and the possible post-processing
steps that were performed. Ash-based materials 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)) of equal chemical composition to 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 validate that ash-based materials can deliver an effective alternative for mined and synthetic P-fertilisers in the European agriculture, but that the RAE is dependent on the properties of the produced ash-based end-material.

It is observed that the effectiveness of ash-based materials in European soils is not affected by soil pH. The impact of pH on the P-dissolution depends on the elemental composition of the materials because P is strongly bond to Ca at high pH and to Fe and Al at low pH (Hinsinger, 2001; Tóth et al., 2014). Therefore, ash-based materials rich in Al/Fe could possibly be more effective in basic soils, whereas Ca-rich constituents could be more effective in acid soils. However, the high basic cation contents of some ash-based materials might buffer the acidity effect of the soil micro-environment, further obscuring the effect of the soil pH. Hence, no overall consistent effect of soil pH on RAE was observed due to the specific interactions between soil pH and ash-based fertiliser properties on P-release. Also, no differences were observed in RAE across plant types, indicating that possible differences in root exudation patterns of organic acids are not impacting the P-release patterns from ash-based materials.

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 materials resulting from wet-digestion post-processing, especially for these that have an equal chemical composition to that of mined and synthetic 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 as Thomasphosphate and Rhenaniaphosphate, 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 3). The plant-availability of the P in P-fertilisers is likely controlled by the coordinated cations of Ca, Mg Al and Fe to which $\text{PO}_4^{3-}$ is bound. All these different ions are abundantly present in ash-based materials, 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 ash-based materials. 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 sorbed 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 and synthetic P-fertilisers compared to ash-based materials rich in trivalent cations.

Studies that supply primary and secondary macronutrients together with mined and synthetic P-fertilisers to ensure the 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 ash-based materials 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 ash-based materials as P-fertilisers could provide the complementary benefit of supplying secondary macronutrients and micronutrients to enhance agronomic yields.
Figure 9: The relative agronomic efficiency of ash-based materials for the plant response variables DMY (dry matter yield) and PUE (phosphorus use efficiency) as a function of grouping variables. Results are presented as weighted mean (square) and 95% confidence intervals (error bars).
5.2.4 Pyrolysis materials

The overall results suggested a significantly lower agronomic efficiency for pyrolysis materials than for mined and synthetic P-fertilisers (Figure 10). **The overall mean effects for pyrolysis materials were 0.87 and 0.46 for RAE_{DMY} and RAE_{PUE}, respectively** (Figure 10). The analyses for each of the grouping variables was constrained by the number of studies available; only the RAE values for neutral and basic soils and for pyrolysis 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 (RAE_{DMY} and RAE_{PUE}; Figure 10). For these groups, the RAE_{DMY} and RAE_{PUE} values pointed towards a significantly lower agronomic efficiency than for mined and synthetic P-fertilisers. The significant differences of specific groups such as those varying in soil texture (RAE_{DMY}), feedstock (RAE_{PUE}), application form (RAE_{PUE}), plant type (RAE_{DMY} and RAE_{PUE}), experimental design and setting (RAE_{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 materials applied to different soil types, feedstocks, application form and plant types. Figure 10 enables, nevertheless, a standardised visual assessment of the RAE ranges observed across selected studies.

Given the small sample size for pyrolysis materials, it is not pertinent to draw overarching conclusions for pyrolysis materials from the available data. The properties of pyrolysis materials can vary widely, depending on the interactive effects between production process conditions and feedstock applied. Many groups, including pyrolysis 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 materials that have been applied in granulated form. For these groups, a lower agronomic efficiency than for mined and synthetic P-fertilisers is observed. Potentially, some of the documented high agronomic efficiencies after PY 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 that 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 materials in the same physical form as it will be applied under actual settings in agriculture. It is concluded that the current available data does not enable a comprehensive assessment of the agricultural efficiency of P-rich pyrolysis materials in relevant European agricultural settings, and that plant responses for P-rich pyrolysis materials can vary widely depending on the feedstock and production conditions of the pyrolysis materials, as well as on the soil and plant type under fertilisation.
Figure 10: The relative agronomic efficiency of pyrolysis materials for the plant response variables DMY (dry matter yield) and PUE (phosphorus use efficiency) as a function of grouping variables. Results are presented as weighted mean (square) and 95% confidence intervals (error bars).

5.2.5 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. Especially the effect of the north–south position (i.e. latitude of the geographic coordinates) is relevant to consider,
given that climate 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 $\text{RAE}_{\text{PUE}}$ was indicated ($p: 0.02$), with greater $\text{RAE}_{\text{PUE}}$ values observed in sites of lower latitudes than in higher latitudes (Figure 11). Latitude explained, however, only a minor share of the total variance observed ($R^2_{\text{adj}}: 0.14$).

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 $\text{RAE}_{\text{PUE}}$ correlated negatively to latitude (Figure 11). Therefore, the effectiveness of $F_{\text{sec}}$ for semi-arid and Mediterranean European regions is supported. 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 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 and synthetic 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.

**Figure 11:** Bubble plot indicating the relationship between $\text{RAE}_{\text{PUE}}$ and latitude. The size of the bubbles represents the number of cases and relative weight for each data pair.
5.3 C-rich pyrolysis materials

A recent meta-analysis for C-rich pyrolysis materials indicated that, on average, C-rich pyrolysis materials did not increase plant yields relative to unfertilised control sites (Simon 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\(^{-1}\). 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 materials. Therefore, positive plant growth responses can mainly be expected in soils of lower fertility.
6 Environmental and human health impacts

6.1 Production phase
[to be completed for the final report]

6.2 Use phase
[to be completed for the final report]
7 Economic impacts

7.1 Sales prices of P-fertilisers

7.1.1 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 price of fertilisers will be equal to the marginal yield gain expressed in monetary terms:

\[ \text{Price}_{\text{fert}}^i = \text{Price}_{\text{crop}}^i \times \text{Area}_{\text{crop}}^i \times (\text{Yield}_{\text{fert,crop}}^i - \text{Yield}_{\text{nofert,crop}}^i) \]

With \( \text{Price}_{\text{crop}}^i \): the sales price for the crop as received by the farmer (Euro per tonne yield), \( \text{Area}_{\text{crop}}^i \): the area of the cultivated crop (ha); \( \text{Yield}_{\text{fert,crop}}^i \): the crop yield on fertilised areas (tonne yield per ha) and \( \text{Yield}_{\text{nofert,crop}}^i \): the crop yield on fertilised 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 12. It is important to notice that all the agents involved are sometimes 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.
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 identifying prices more accurately, as these are observed at a consistent point. Moreover, focussing 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 as 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 practiced, 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 a “P₂O₅ 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+P₂O₅+K₂O) across commoditised products, meaning that it minimises freight costs per tonne 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 logistic cost associated to the transport of SSP (low P-content; 16-22% P₂O₅ versus 46% P₂O₅ 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.

7.1.2 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 to higher sales prices, but such price settings are not covered in this document.

Through the 1990s and in to 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 in the 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 period 2002 to 2007 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 $235/t (189 Euro/tonne), in 2007 it was $425/t (310 Euro/tonne) and in 2009 it was $1090/t (782 Euro/tonne). With the economic crash in late 2008 reality re-asserted 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 13 shows the price evolution for DAP and rock on the basis of FOB Morocco / North Africa. The breaks in the prices were effectively when there was no market; producers wanted prices buyers were unprepared to pay.

![Price evolution for diammonium phosphate (DAP) and phosphate Rock (Euro/tonne product; assuming an average exchange rate of 1.2 Euro/US dollar; 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 off-set 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 14).

The price per tonne P in mined and synthetic 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 (381 US dollars per tonne ammonia-N, or 69 US dollars for the 18% N in DAP). Also, a constant 1.2 Euro/US dollar exchange rate is used in these calculations. A price of 988 Euro per tonne P FOB is indicated for the year 2017.
Figure 14: Fertecon DAP Price Forecast to 2025 expressed per tonne of DAP and per tonne P (FOB Morocco; exchange rate of 1.2 Euro/US dollar 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 period 1990 to 2005, the current price level might be better described as ‘normal’.

7.1.3 Price setting for STRUBIAS materials

Given the lack of legal outlets for many STRUBIAS materials in many EU Member States, these materials are at present not valorised within 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 challenging 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.

This section aims at estimating the potential value of STRUBIAS materials and incineration ashes that can be used as raw materials by the fertiliser industry. Rather than giving an overview of the expected sales prices, this section provides a benchmark for the economic valuation of STRUBIAS materials, and focusses on the drivers that can be expected to affect sales prices for use in conventional agriculture.

It is also important to note that the values are presented for Cost and Freight (CFR) agreements to downstream users, and thus include the shipment costs to the downstream processor or user. 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 valuation (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 P2O5 can be expected than for finished fertiliser products as various additional costs need to be 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 that 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 P2O5 can be expected compared to raw material prices; the expected sales price includes the manufacturing costs already occurred throughout the value chain, plus some profit margin of all actors involved within 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 somewhat in the middle between the previous two categories. Some producers purchase products (e.g. DAP), grind it 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 by far and large 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 often are able to obtain a relatively lower price per tonne P2O5 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 setting for the P contained in the STRUBIAS materials is likely to occur 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 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 \( (P_{\text{intermediate}} \text{ in Table 11}) \), 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 \( (P_{\text{final}} \text{ in Table 11}) \). 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 10: Value of P present in DAP and phosphate rock from Morocco (Euro per tonne P, CFR Morocco) assessed for different EU Member States for the year 2017 based on import prices (Source: Fertecon; exchange rate of 1.2 Euro/US dollar applied by JRC).

<table>
<thead>
<tr>
<th></th>
<th>France</th>
<th>Germany</th>
<th>Italy</th>
<th>UK</th>
<th>Poland</th>
<th>Romania</th>
<th>Average</th>
</tr>
</thead>
<tbody>
<tr>
<td>DAP</td>
<td>1213</td>
<td>1397</td>
<td>1213</td>
<td>1234</td>
<td>1198</td>
<td>1256</td>
<td>1252</td>
</tr>
<tr>
<td>phosphate rock</td>
<td>558</td>
<td>685</td>
<td>594</td>
<td>852</td>
<td>539</td>
<td>665</td>
<td>649</td>
</tr>
</tbody>
</table>

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 recovered phosphate salts. The use of recovered phosphate salts in fertiliser blends could effectively the Cd content in P-fertilisers, and avoid as such the need for decadmiation (a cost estimated of about 27 – 68 Euro 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 130 Euro per tonne Mg. An additional Mg value for STRUBIAS materials containing Mg (e.g. struvite) will be added. **Nitrogen** is valued at around 320 Euro per tonne across EU Member States. The value of N is based on 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 and synthetic 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 its use in fertiliser blends, and v) are generally associated to 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 11.

**Table 11: Benchmark CFR prices for the valuation of incineration ashes and STRUBIAS materials based on the P contained and other relevant price setting factors (Euro per tonne material).**

<table>
<thead>
<tr>
<th>material</th>
<th>basic price (P_{\text{intermediate}})</th>
<th>price premium</th>
<th>price discount</th>
</tr>
</thead>
<tbody>
<tr>
<td>granulated struvite (12.6% P, 9.9% Mg, 5.7% N)</td>
<td>- 158</td>
<td>low in metals, N &amp; Mg value of 28 Euro/t, granulated material</td>
<td>not applicable</td>
</tr>
<tr>
<td>powdered struvite (12.6% P, 9.9% Mg, 5.7% N)</td>
<td>- 158</td>
<td>low in metals, N &amp; Mg value of 28 Euro/t</td>
<td>not granulated</td>
</tr>
<tr>
<td>sewage sludge ashes (9% P, low in Al/Fe content)</td>
<td>58 -</td>
<td>not applicable</td>
<td>not applicable</td>
</tr>
<tr>
<td>poultry litter ashes (5.5% P, )</td>
<td>- 69</td>
<td>PK fertiliser</td>
<td>low nutrient density, agronomic efficiency tested on limited soil and plant types</td>
</tr>
<tr>
<td>TSP derived from sewage sludge ash (20% P)</td>
<td>- 250</td>
<td>low in metals, granulated material</td>
<td>not applicable</td>
</tr>
<tr>
<td>Thomasphosphate-like ashes (e.g. AshDec, 7.5% P)</td>
<td>- 94</td>
<td>low in Cd</td>
<td>low nutrient density, agronomic efficiency tested on limited soil and plant types</td>
</tr>
</tbody>
</table>

For some STRUBIAS materials, the target market for pyrolysis materials may not be the conventional agricultural market. Specific materials of all SRUBIAS material groups could possibly make a market entry niche markets where fertilising products are traded at higher prices (e.g. organic farming, horticulture, and growing media). In organic farming, the absence of competition from mined and synthetic 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 7.1.2. For these materials, a negligible increase for the period 2016 - 2025 is expected. The future prices for STRUBIAS materials are therefore expected to remain stable. Nonetheless, the consumers’ confidence in STRUBIAS materials and the field verification of the presumed agronomic efficiency in the relevant agricultural sectors could further impact the future prices for 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 target markets.

Question 1 to STRUBIAS sub-group:

Please provide an indication of:

- the sales prices of STRUBIAS materials and mono-incineration ashes on the market (Euro per tonne material and P concentration, or Euro per tonne P; indicate if prices are "Free on Board (FOB)" or “CFR (Cost and Freight)” with an indication of the price for transport).
- Indicate also the physical form of the material (powder, granules, other relevant physical parameters), and to whom materials are sold (blending companies, retailers, end users, etc.).
- If available, please provide an evolution of the average sales prices in the last decade in order to determine the main factors affecting fertiliser price and their relative importance.

7.2 Production cost for STRUBIAS materials
[to be completed for the final report]

7.3 Compliance costs
[to be completed for the final report]

Question 2 to STRUBIAS sub-group:

Please provide information on the following elements that form part of the compliance costs:

- Cost for REACH registration for fertiliser end-material that will be brought on the market.
- Cost for compliance under already existing national end-of-waste or similar regimes that enable a market entry for fertilising products derived from STRUBIAS materials.
• Estimated cost for compliance for P-fertilisers derived from primary raw materials.

• For facilities that process waste-based materials, information on the cost associated to acquiring waste permits in different EU Member states for non-hazardous and hazardous waste materials. Notably, the costs associated to complying with the obligation for an establishment or undertaking carrying out waste management operations to have a permit or to be registered in accordance with Article 23-26 of the Waste Framework Directive 2008/98/EC.

• Cost of sampling and analysis through accredited laboratories:

  Recovered phosphate salts:
  - Nutrients: P, Ca, Mg, citric-acid P
  - Metals and metalloids: As, Cd, Cr (total), Cr (VI), Cu, Hg, Ni, Pb, Zn
  - Persistent organic pollutants: PAH$_{16}$
  - Biological pathogens: E. coli or Enterococcaceae & Salmonella spp.
  - Others: macroscopic impurities, dry matter content, particulate matter < 100 µm.

  Ash-based materials:
  - Nutrients: P, K, Ca, Mg, S, citric-acid P
  - Metals and metalloids: As, Cd, Cr (total), Cr (VI), Cu, Hg, Ni, Pb, Zn, B, Ba, Co, Mn, Mo, Sb, V
  - Persistent organic pollutants: PAH$_{16}$, PCB, PCDD/F
  - Others: pH and neutralising value

  Pyrolysis materials:
  - Major elements: C, Corg, P, K, Ca, Mg, S
  - Metals and metalloids: As, Cd, Cr (total), Cr (VI), Cu, Hg, Ni, Pb, Zn, Ba, Co, Mo, Sb, V
  - Persistent organic pollutants: PAH$_{16}$, PCB, PCDD/F
  - Biological pathogens: E. coli or Enterococcaceae & Salmonella spp.
  - Other: pH, neutralising value, macroscopic impurities, particulate matter < 100 µm, particle density, volatile organic matter, specific surface area earthworm avoidance test (ISO 17512)

Where analysis packages are available (e.g. sampling + analysis of a series of metals, PAH$_{16}$, PCB and PCDD/F), please clearly state what the package contains and its cost.

• Measurement standards currently applied (national standards, ISO/EN standards, etc.)

7.4 Possible economic benefits and drawbacks of producing fertilising products containing STRUBIAS materials

Question 3 to STRUBIAS sub-group:

Please provide information, preferably in a quantitative manner, on following possible economic benefits and drawbacks of producing fertilising products containing STRUBIAS materials compared to equivalent mined and synthetic inorganic fertilising products.
• reduced waste compliance costs (e.g. changes in the economic valuation of sewage sludge ashes, etc.);
• reduced externalities (e.g. avoided costs due to eutrophication, positive effects on human health due to reduced contaminant levels, etc.);
• potential job creation in production and downstream fertiliser distribution and farmer's cooperatives; please relate expected STRUBIAS production volumes to number of persons employed.
• impacts on the rural economy;
• benefits of restoring soil organic carbon for soil fertility;
• cost associated to new logistics for recovered nutrient products;
• implications for the restructuring the production and distribution of fertilising products;
• agricultural equipment adaptations.
Market aspects and outlook for the year 2030

In the proposal for the Revised EU Fertiliser Regulation, 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 sub-sections that cover uses of such materials in P-fertilisers (section 8.1 and 8.2) or in liming materials, soils improvers, growing media and plant biostimulants (section 8.3). The sections 8.1 and 8.2 on P-fertilisers cover materials with a minimum P-content of 3% and 2% P₂O₅, 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 8.3.

8.1 Market aspects and outlook for P-fertilisers

8.1.1 Phosphate rock as the major source for P-fertilisers

The current market of P-fertilisers is dominated by synthetic inorganic 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 quantities applied and agricultural sectors covered. The future outlook of P-fertilisers is based on the evolution of supply and demand of mined of synthetic and mined fertilising products that currently dominate the market of P-fertilisers.

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 sulfuric 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% P₂O₅ and 0% K₂O), 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 NPKs. 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. The main benefit of this route is that it includes an intermediate stage where phosphoric acid is formed, which currently is the only potential stage where decadmiation technologies could possibly be applied to remove cadmium.

- The second route also uses sulfuric 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 (“Odda” process or 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 a chemical reaction in a chemical plant. For example, an 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 e.g. steam granulation. This ensures that the desired N, P and K proportions are present in each granule, but without 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 e.g. AN, or MAP, or KCl. Blending plants are the least expensive, requiring only storage, physical mixing and bagging operations, and have the highest flexibility in terms of products offered.

![Figure 15: Production routes for P-fertilisers that are derived from mined phosphate rock](source: European Phosphate Fertilizer Alliance (2017))
8.1.2 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 plantings 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-demand by crop was given for Europe (Figure 16). Cereals consumed >50% of all P-fertilisers in 2010-11.

Figure 16: P-fertiliser use by crop (%) in Europe for the year 2010/2011 (Source: IFA)

The main driver of the reduction in apparent P-fertiliser consumption in Europe has been the significant change 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 has caused that production and fertiliser use declined, or at best stabilised, in the period to 2008. For example, maize production in the EU-28 dropped from 73 Mt in 2004 to 49 Mt in 2007, and wheat dropped from 150 Mt to 121 Mt over the same timeframe. 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% p.a. through 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 plantings, crop prices, the weather, government policy and fertiliser prices will all influence how demand develops.

8.1.3 Production and consumption volumes

8.1.3.1 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 economic deposit of phosphate rock occur where there are one or more seams of rock containing generally more than 15% P₂O₅, which have 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 17.
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, 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 of 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 (SSP), single superphosphate (SSP). The main phosphate rock importing countries in the EU are Lithuania, Poland, Belgium, Bulgaria, Romania and Spain. Most of EU imports come from Morocco (1.6 Mt), Russia (1.4 Mt), Algeria (702 kt), Israel (506 kt), and South Africa (429 kt).

**Imports of rock** into Europe have 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% P₂O₅ (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 is 7.9% p.a., whereas in the EU-13 it
is a much more modest 1.2% p.a. 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 equal ~936 kt P (6.7 Mt material x 0.32 kg
P₂O₅ / kg phosphate rock x 0.437 kg P/ kg P₂O₅).

Figure 18: 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% p.a., compared with a more modest 1.4% p.a. in the EU-13 countries (Figure 18). Since
2010 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 5.0M
tonnes product to levels below 3.5M tonnes product (Figure 18).

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% P₂O₅, and purified phosphoric
acid (PPA, otherwise known as white acid), typically containing around 61% P₂O₅. 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 equal ~ 524 kt P (Figure 19). The EU-15 accounts for over 95% of all
phosphoric acid imports in the EU (Figure 19). This might overstate reality, depending on the
accuracy of data on re-exports from key import-ports in e.g. 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 have declined by 1.3% p.a. in the 2005 – 2015 timeframe, whereas imports have grown at 1.8% p.a. in the EU-13 region.

**Figure 19:** Imports of phosphoric acid into Europe by region (Source: GTIS, IFA)

### 8.1.3.2 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 trustworthy. As indicated by Eurostat itself, the quality of 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 harmonized data source. Data are available at NUTS0 for EU-27 (i.e. does not include Croatia). Data from Fertilizers Europe relate to crop years (t-1/t) which are reported under year t-1. These data have been used in this assessment, also because they are in between the Member State data documented by Eurostat and the FAO.

Fertilizers Europe assessed the apparent P-consumption for fertilisers in the EU-28 at **1090 kt P in 2015** (Figure 20). Eurostat estimated the total P-consumption as mineral
fertilisers at 1133 kt P for the year 2015, whereas FAOSTAT documents 1070 kt P for the EU-28 in 2015. The data from Fertilizers Europe indicate that mineral P-fertiliser consumption was larger in the EU-13 (70%, 765 kt P yr\(^{-1}\)) than in the EU-13 (30%, 325 P yr\(^{-1}\)) (Figure 20).

Looking forward, Fertecon forecasts phosphate consumption for fertilisers to grow across the EU-28 at a CAGR (Compound Annual Growth Rate) of 0.8% p.a. for the next decade. Growth in the EU-15 is stable within any reasonable margin of error (0.09% p.a.). Fertecon, however, forecasts a growth in the EU-13 countries at a CAGR of 2.2% (Figure 20). Future estimates on mineral P-fertiliser consumption were obtained by projecting these CAPR growth rates, resulting in a 2030 estimate of 1220 kt P yr\(^{-1}\). The EU-13 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\(^{-1}\)).

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. (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 21. The analysis shows that in terms of product types, NPKs accounted for 55% of P-containing tonnes consumed, followed by DAP (18%) and SSP (8%) (Figure 21). In terms of
actual P delivered however, because NPKs contain less P than as MAP, DAP or TSP, the proportions change. DAP has the largest share at 32%, followed by NPKs at 31%, and MAP with 12% (Figure 21).

Figure 21: Apparent consumption of P-fertilisers in the EU-28 for the year 2015. The left Figure indicates the actual tonnes of material; the right Figure indicates the same data expressed on a P-basis (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 7 markets accounted for just over 76% of apparent consumption of phosphate for fertilisers, with the balance spread over the remaining EU countries (Figure 22).
Figure 22: EU-28 Top 10 markets for apparent consumption of mineral phosphate fertilisers for the year 2015 (Source: Fertilizers Europe)

Box 2: Production of DAP, MAP, TSP, SSP and NPK in the EU (AEEP, 2017; Fertecon):

- **Diammonium phosphate (DAP, (NH₄)₂HPO₄):** DAP is typically 18-46-0 (i.e. contains 18% N, 46% P₂O₅ and 0% K₂O). It was one of the first fertilisers to have a standardised content, which in part explains why it is the largest 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, NH₄H₂PO₄):** 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 as the main producers.

- **Single Superphosphate (SSP; Ca(H₂PO₄)₂ + CaSO₄):** 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(H₂PO₄)₂•H₂O):** TSP is the highest analysis 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.

8.2 Market outlook for P fertilisers derived from STRUBIAS materials for the year 2030

8.2.1 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 so-called "relative agronomic efficiency" (see section 5). A relative agronomic efficiency value below 100% indicates 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.

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 23). Legislative and policy impacts due to beneficial impacts of STRUBIAS materials relative to alternative fertilising products that are available on the internal market are considered. As matter of fact, given that the production cost for STRUBIAS might be higher than for mined fertilising products (Egle et al., 2016), legislative aspects and policies may prove to be important drivers for STRUBIAS materials.

Figure 23: 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, etc.), 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 case STRUBIAS
materials are associated to 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 8.2.10.2).

The sections 8.2.4 - 8.2.8 discuss market estimates under the anticipatable EU and national legislation for all process pathways of high technological readiness level (TRL 7-9). A broad spectrum of new STRUBIAS production processes are emerging and developing, some of them being present at 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 TRL 6 processes are discussed together with possible legislative drivers in section 8.2.10.2.

8.2.2 Market and consumer readiness

8.2.2.1 Conventional agriculture

STRUBIAS materials are used to produce a variety of fertilising products (see section 4). On the one hand, STRUBIAS materials can be used for the production of traditional P-fertilisers of 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 of a different chemical composition. Several of these "new" P-fertilisers are marketed as "slow-release" or "controlled-release" because the P is not water-soluble. Development of new fertiliser products requires determining 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 machineries that apply fertilisers in a time and energy-efficient manner, for which reason the physical form of the fertilisers should be compatible with existing machineries. This implies, for instance, that fertilisers that will be broadcasted 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 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. Alternatively, materials could be applied using lime spreaders that homogeneously distribute fertiliser materials on the field, albeit in a more labour-intensive manner.

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 recovered phosphate salts is already in plant-available form, there is no need for acidulation; ICL tests indicated that in quantities <20%, recovered phosphate salts can be placed directly in a granulator with acidulated phosphate rocks (Six et al., 2014). In such a process, struvite is physically mixed 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 as for traditional P-fertilisers. Some fertiliser blending companies formulate poultry litter ashes as powder with KCl or TSP. The above assessment indicated that a market entry for recovered P-fertilisers would only be based on proven agronomic benefits and equal fertiliser efficiencies relative to traditional P-fertilisers expressed on a monetary basis. Such proof would not be available until 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 than those products currently dominating the market (DAP, MAP, TSP, SSP, nitrophosphate, etc.). There is an increased acceptance for controlled release fertilisers within the European agricultural sector, but the agronomic value of the marketed product should be extensively demonstrated before a market breakthrough for new fertilising materials can be expected.

In conventional European agriculture, the P-fertilisers MAP and DAP (with a P$_2$O$_5$-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 8.1.3.2). The straight P-fertiliser TSP (45% P$_2$O$_5$) and SSP (16-20% P$_2$O$_5$) only make up 18% of the total share of all P-fertilisers, expressed on a P-basis (see section 8.1.3.2). 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, a cost which exceeds in some cases the cost of fertiliser purchase. 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.

Assuming a similar price setting as for 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 synthetic and mined 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 of chemical composition that are already available on the market and slow-release fertilisers that are well-advanced in the process of agronomic efficiency testing and are compatible for blending are associated to 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 market niches or be traded at a lower price.

8.2.2.2  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 also natural or naturally-derived substances and low-solubility mineral fertilisers can 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. P-fertilisers derived from STRUBIAS materials could provide an alternative for meat and bone meal, meat and bone meal ashes and low concentrated P-fertilising products such as manure...
and compost that are currently the major P-sources in organic farming (Nelson and Janke, 2007). Because of the low N:P ratios of many organic inputs, organic farmers often over-fertilise for P while trying to match crop N requirements; evidence on the potential uptake of STRUBIAS materials remains indeterminate, except in organic systems that rely on biological N fixation for N inputs (Seufert and Ramankutty, 2017). 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 evaluated positively two dossiers proposing authorisation of recycled phosphate products as fertilisers in organic agriculture (under 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. EGTOP concluded that struvite recovery is conform 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 is conform 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 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 revised Fertiliser Regulation 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 synthetic and mined P-fertilisers is negligible in this agricultural sector. Regardless of their 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 synthetic and mined P-fertilisers unless a significant expansion of organic farming occurs prior to the year 2030 at the expense of traditional farming.
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 of 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 8.2.10.2 whereas TRL 7-9 processes are split up as a function of eligible input material (see section 8.2.4 - 8.2.8).

The underlying justification for excluding TRL 1-5 processes is the unlikeliness of technologies of a lower technological readiness level to

- surpass TRLs 6-9;
- comply with administrative arrangements (e.g. obtain waste treatment permit, REACH registration of CE fertilising product containing STRUBIAS materials, arrangements with downstream distributors and retailers, etc.);
- build the necessary infrastructure; and
- perform extensive product testing on agronomic efficiency.

in order to ensure market and consumer confidence (see section 8.2.2) prior to the year 2030. Although TRL 1-5 STRUBIAS materials are not covered in this section, the possibility of such emerging process pathways to contribute to the substitution effect for mined and synthetic 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. Given that the total production cost of STRUBIAS materials from some input materials is higher than for P-fertilisers derived from phosphate rock, the substitution potential will be partially dependent on the extent of existing pre-treatment processes for primary input materials. Often, such pre-treatment has other objectives than the recovery of nutrients: energy recovery from manure, steel products from ores, etc. An additional aspect related to infrastructure that is especially relevant to consider for ash-based materials 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., 2013). In the interest of promoting more efficient recycling of the P in sewage sludge, the mono-incineration capacity should therefore be increased in the EU.

A major challenge also lies in increasing the production volumes of P-fertilisers derived from secondary raw materials. With the exception of struvite producing 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 and Budenheim, will start to build full-scale installations in Europe, of which the first will most likely be operational in 2018. It should be evaluated to what extent current P-fertiliser production processes through the acidulation route could be adapted to use other P-sources than phosphate rock (for instance P-rich ashes). 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.

### 8.2.4 STRUBIAS materials from crop residues, manure, and bio-waste

#### 8.2.4.1 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, nut shells. 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 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. Together 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 3.2). 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 pre-treatment, 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 pre-treatment) or a thermochemical process (based on hydrogenation, gasification or pyrolysis). The review of a wide range of life cycle assessments (LCA) 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 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 utilization and (iii) prohibited periods for land spreading (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 largest 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 of high moisture content (Foget et al., 2011). Several EU Member States have no other type of manure processing than 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 pre-treatment technologies that can enhance the biogas production, or as post-treatments, which can help to convert the digestate into products with envisaged properties (Foget et al., 2011). Solid manure fractions, such as poultry manure, can also be used for direct incineration leading the manufacturing of ash-based materials, 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 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 3.6).

Co-digestion of animal manure with solid or slurried bio-waste of high methane potential such as oily residues and by-products, alcohol residues, digestible organic wastes from agri-processing and 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 utilise 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 500-2000 mg L⁻¹ PO₄³⁻P, whereas liquid digestate fraction produced from dominantly plant-based materials contain much lower phosphate contents (~0 - 500 mg L⁻¹ PO₄³⁻P) (Akhiar et al., 2017).

### 8.2.4.2 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 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 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 the 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, as appropriate, and in accordance with Articles 4 and 13 of the 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 an insufficiency 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 is 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 is avoided at the same time. The monitoring of waters for nitrate 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 2000/7/2006), 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-environment 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 Directive 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 (Art. 9(3)) in order to continuously prevent and reduce pollution. The purpose of the IED Directive was to achieve integrated prevention and control of pollution arising from several categories of industrial activities. The indicative list of 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 environmental-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 set 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, 2015b) 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 Waste Water Treatment Directive. This is much 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 Water Framework Directives 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 nitrates emissions outside nitrate vulnerable zones established under the Nitrates Directive.

Supplementary measures reported in agriculture are largely voluntary, including advice schemes and agri-environment 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 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, 2013a).

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 recovered phosphate salts, ash-based materials and pyrolysis 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 produced digestate 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 high livestock density, make recovery and recycling from 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 characterized 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 N–K 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 are 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, 2013a). Wastes and crop residues are currently underused and can contribute significantly to reaching EU bioenergy targets (European Environment Agency, 2013a). 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 to 13.5 Mtoe, with the main producers being Germany, 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 were 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 generated electricity, 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 the Slovak Republic and Hungary. Slovak energy distribution companies announced a blanket stop on connecting new electricity producing 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 into biogas facilities economically unattractive. In the Netherlands, the low electricity prices have even led to a decline in biogas production.
Landfilling of biodegradable waste


8.2.4.3 Anaerobic digestion followed by P-precipitation

The European Environment Agency that 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 content of phosphate 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 3: Recovered phosphate salts 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, 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 biowaste 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 of 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 of P 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, was 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
112

million tonnes of manure would be processed through anaerobic digestion, with an estimated P content of 306 kt P. 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 tear 2005). These P-estimates are assumed to at steady state for the year 2030.

5) It is assumed that recovered phosphate salts 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 processes in Europe that processes P-rich digestate liquids that complies with the draft proposals of the STRUBIAS nutrient recovery rules. In this process, K-struvite is precipitated from the liquid digestate fraction. The cost assessment of the process indicates a solid business case for the plant in the Netherlands where manure treatment is associated with a gate fee, even without considering the revenues from the sales of the K-struvite. Given the estimate pricing of recovered phosphates in the future (see section 7.1), the return on investment of similar P-recovery systems is high. Therefore, it can be 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 10-30% 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 recovered phosphate salts. 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) of recovered phosphate salts relative to synthetic and mined P-fertilisers is assumed to be 1.05 (see section 5.2.2).

7) Considering 1) - 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 on 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, etc.) is unlikely.
8.2.4.4 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), IE (BHSL), the Netherlands (BMC Moerdijk), and some Scandinavian Member States produce 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 (De Graaff et al., 2017). The study concluded that the production of electricity through thermal oxidation is the most attractive and sustainable manner to process poultry litter from an environmental perspective.

In section 5.2.3, a relative agronomic efficiency for the response variable phosphorus use efficiency (RAE\(_{\text{PLUE}}\)) of 1.49 was observed. The accuracy of this value is, however, uncertain, because this value was based on only 4 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 5.2.3 due to lack of detailed results. Therefore, the lower end estimate of 0.90 for RAE\(_{\text{PLUE}}\) for poultry litter ashes was retained for further calculations; this value corresponds roughly to the weighted mean of the data from section 5.2.3 and the results given in De Graaff et al. (2017).

Box 4: Ash-based materials from solid manure fractions
To derive the substitution potential of P-fertilisers containing STRUBIAS materials from solid manure fractions from the agricultural sector, following assumptions and calculations were made:

1) Only poultry litter is considered as input material for ash-based STRUBIAS materials 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 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) – 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 at a relative small scale, enabling the establishment of many small-scale facilities to process high-volume, geographically dispersed materials, such as excess manure, without excessive transport costs. Moreover, the syngas generated during the pyrolysis process can be used as an energy source to dry the manure input material, although the drying takes place to the detriment of renewable energy production. 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 be principally performed as a measure for pollution control.

Although no commercial thermochemical technologies using biomass at the time of writing have been identified in Europe, 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, as obtained after solid-liquid separation treatment. This issue is particularly relevant given that the highly diluted pig
manure is associated with high transport costs, and most farmers have only a small surface
area to dispose the slurry produced (Foget et al., 2011).

The EU funded BioEcoSim project (https://www.bioecosim.eu/) projects to process
2.000.000 m³ of pig manure per year, resulting in a production of 62 kt of pyrolysis materials
derived from pig manure (3 kt P, assuming a 5% P content in the resulting output material).
The Japanese company Hitachi-Zosen Limited also aims to turn pig manure from Dutch and
Belgian pig farmers into P-rich pyrolysis materials. The latter company has gained
experience with the process from their operations in Japan. Production costs for P-rich
pyrolysis materials are estimated at about 4 Euros per tonne of pig manure slurry (Smit et al.,
2012), or about 4000 Euro per tonne P. It should be noted, however, that the material also
contains other elements of interest that can be returned to the agricultural field (see below).
The pyrolysis process reduces the mass of the digestate by 70%, significantly reducing
transport costs to agricultural land.

The market outlook for this STRUBIAS pathway is associated to a large degree of
uncertainty as the resulting pyrolysis material is the only output material that will be brought
on the market. The economic valuation of pyrolysis materials is at present, however,
largely unknown, as well as the degree of consumer confidence associated to the end-
material. Pyrolysis 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.
Expressed on a P-basis, the producers of pyrolysis materials claim a higher sales price for
pyrolysis materials than for traditional P-fertilisers, based on allegations of high agronomic
efficiencies (> 100%, expressed on a P-basis relative to synthetic and mined fertilisers) and
additional benefits from increased soil fertility (see section 7.1.3). Nonetheless, at present,
scientific and experimental evidence is lacking that demonstrates the superiority of pyrolysis
material compared to traditional P-fertilisers in terms of fertiliser efficiency (see section
5.2.4).

The production cost for P-rich pyrolysis materials derived from manure is higher than
for synthetic and mined P-fertilisers. Moreover, the nutrient density of the resulting
pyrolysis material is much lower, resulting in a higher land application cost for pyrolysis
materials relative to synthetic and mined P-fertilisers. Unless the high sales prices of the
pyrolysis materials could be realised, it is assumed that the market potential of pyrolysis
material in the conventional agricultural sector is low to negligible in the short-term (<
year 2030). Given the lack of availability of concentrated P-fertilisers for organic farming,
pyrolysis materials could potentially make an entry into 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 this STRUBIAS material might be demanded
and traded in this sector. Here, the higher price setting of P-rich pyrolysis materials relative
to synthetic and mined P-fertilisers could potentially be compensated by the higher sales
prices of organically-grown food products. Manure-derived pyrolysis materials could thus
replace current organic P-inputs such as manure, compost, meat & bone meal and meat &
bone meal ashes in the organic farming sector. It is, however, concluded that STRUBIAS
materials produced through the pyrolysis of manure fractions will not significantly contribute to the substitution of synthetic and mined P-fertilisers as the latter are only marginally used in this agricultural sector.

8.2.5 STRUBIAS materials derived from slaughter residues

8.2.5.1 Introduction

Animal by-products activities covered under this section include the by-products obtained from the treatments of entire bodies or parts of animals at slaughterhouses. Rendering activities include the treatments 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 category 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 produced ashes should be disposed of as waste. The ban on the use of processed animal proteins in feed for animals farmed for food and pet food 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, also having a legal status as 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 3 (11.7 million tonnes) (EFPRA, 2017).

8.2.5.2 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, 2016b). After the recent recovery, 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 next years 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 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 somewhat increased export prospects (European Commission, 2016b).

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 category 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 category 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 of 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). Also some 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 smaller 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 50-70 Euro per tonne, or about 943 – 1320 Euro per tonne P. The bone fraction is even traded at a higher price, expressed on P-basis.
8.2.5.3 Thermochemical conversion processes

Meat and bone meal has a heating value ranging between 13000 and 30000 MJ per tonne (Conesa et al., 2003). Therefore, a 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, 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. Also, the environmental impact of producing fertiliser using these secondary phosphate sources suggests the emissions of phosphate and fluorine is 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 (943 – 1320 Euro per tonne P, see section 8.2.5.2) are more expensive than phosphate rock (on average 649 Euro per tonne P; see section 7.1.3). However, in contrast to phosphate rock, the combustion of meat and bone meal and processed animal proteins will enable energy recovery. Thermochemical P-fertiliser production processes such as the Mephrec process, the FEhS/Salzgitter process, and the thermo-reductive Recophos process (see paragraph 4.2.2) 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 directly be used as a P-fertiliser. The P-recovery rates of these processes are high, with values ranging from 80% to 89% (see paragraph 4.2.2). Moreover, meat and bone meal ashes are a P-rich material (~14%, Moller, 2015) low in Al/Fe oxides, for which reason they could also be used to increase the
The suitability of combustion ashes, either directly as a fertiliser or as an intermediate in P-fertiliser production processes (Vamvuka et al., 2017).

<table>
<thead>
<tr>
<th>Box 5: Ash-based materials from slaughter residues</th>
</tr>
</thead>
<tbody>
<tr>
<td>To derive the substitution potential of P-fertilisers containing STRUBIAS materials through this route, following assumptions and calculations were made:</td>
</tr>
<tr>
<td>1) For the year 2030, the total incoming P in the protein fraction of category 2 and 3 materials contains is assumed <strong>154 kt of P yr⁻¹</strong>. In line with Regulation EC 1069/2009, category 1 animal by-products are not considered in this assessment.</td>
</tr>
<tr>
<td>2) It is assumed that the increased demand of meat and bone meal and processes animal proteins in P-fertilisers will lead to a new equilibrium for the use and fate of category 2 and 3 animal by-products. The rate of adjustment to bring markets to the long run equilibrium depends on various factors, such as the degree of technological change or market conditions, and cannot be clearly determined at the moment. In this preliminary assessment, it is assumed that material use for the production of STRUBIAS P-fertilisers will increase to half of the total fraction, or <strong>77 kt of P yr⁻¹</strong>. The STRUBIAS sub-group is invited to provide techno-scientific data that can further refine this estimate.</td>
</tr>
<tr>
<td>3) The assumed P-recovery rate is on average <strong>85%</strong> of the P present in the protein fraction.</td>
</tr>
<tr>
<td>4) It is assumed that the materials will be used in thermal and wet-chemical manufacturing processes that aim at improving the agronomic efficiency of the materials relative to raw ashes. The relative agronomic efficiency (RAE_{PUE}) of thermal and wet-chemical processes is estimated at 0.85 and 1.00, respectively (see section 5.2.2). <strong>A mean RAE value of 0.92 is assumed.</strong></td>
</tr>
<tr>
<td>5) Considering 1) - 4), the P-recovery from slaughter residues is estimated at <strong>61 kt of P</strong>. It is estimated that these materials will be brought on the market as P-fertilisers that are already available on the market (e.g. MAP, DAP, TSP, etc.) or as P-fertilisers that have chemical composition similar to Thomasphosphates and Rhenaniaphosphates.</td>
</tr>
</tbody>
</table>
Pyrolysis of animal bone materials

Animal by-products of Category 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%), CaCO₃ (7-13%) and carbon (9-11%), with a P content of 13% (30 P₂O₅) (3R AgroCarbon, 2016).

Similar to the market for pyrolysis materials derived from manure, the market outlook for this STRUBIAS pathway is associated to a large degree of uncertainty as the resulting pyrolysis material is the only output material that will be brought on the market. The economic valuation of pyrolysis materials is at present, however, largely unknown, as well as the degree of consumer confidence associated to the end-material. Expressed on a P-basis, the producers of pyrolysis materials claim a higher sales price for pyrolysis materials than for traditional P-fertilisers based on allegations of high agronomic efficiencies (> 100%, expressed on a P-basis relative to synthetic and mined 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\textsubscript{PUE}) that is, at a maximum, close to a value of 1 (thus equivalence to mined and synthetic P-fertilisers; see section 5.2.4).

The production cost for P-rich pyrolysis materials derived from animal bone material is higher than for synthetic and mined P-fertilisers. Unless the high sales prices of the pyrolysis materials could be realised, it is assumed that the market potential of animal bone-derived pyrolysis material in the conventional agricultural sector is low to negligible in the short-term, i.e. before the year 2030. Given the lack of availability of concentrated P-fertilisers for organic farming, pyrolysis materials could potentially make an entry into the organic farming sector. It is re-iterated 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 synthetic and mined P-fertilisers could potentially be compensated by the higher sales prices of organically-grown food products. Here, animal-bone derived pyrolysis 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 materials in the organic farming sector. A market entry in this sector may also enable further long-term product testing under realistic circumstances. It is, however, assumed that STRUBIAS materials produced through the pyrolysis of animal bone materials will not significantly contribute to the substitution of synthetic and mined P-fertilisers as the latter are only marginally used in this agricultural sector.

8.2.6 STRUBIAS materials from municipal wastewaters

8.2.6.1 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 physical-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 to 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 86/278/EEC (see section 8.2.6.2). Therefore, the fraction of sewage sludge that is disposed of 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 a possibility 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 recovered phosphate salts and ash-based materials.
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. These Directives are listed in Appendix 1. 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 Waste Water 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 waste water 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 metals concentration for sewage sludge as well as for soil when sewage sludge is used on land and maximum annual heavy metals 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, re-use, (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 as a priority order 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 agriculture 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. Environmental impacts of transport of the high-volume sludge should also be considered.

Benefits in terms of climate change and greenhouse gases 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, it remains unknown to what extent the long-term effect is scientifically substantiated 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 to 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-incinerated 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 has 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 ash-based STRUBIAS materials will have to comply with contaminant limits in the proposed revision of the Fertiliser Regulation that are more strict 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 126-185 Euro per tonne sludge (dry matter basis), whereas the cost for the mono-incineration of sewage sludge is estimated at 411 Euro 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 (Nättor & 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, 2017a). A total of 43 mono-incinerators are present in Europe, of which the overall share (23) is located in Germany (European Commission, 2017a). For cities and regions where infrastructure with mono-incinerators is available and P-rich ashes are already generated, recovering P from ashes with best available ash-process techniques is associated to little or no additional costs (compared to ash disposal) if revenues from P-fertilisers sales are taken into consideration (Egle & al., 2016). This is exemplified by the agreement between EcoPhos and Dutch mono-incinerating facilities (60.000 tonne of sewage ashes per year), indicating the commercial viability to transform 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 to 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 on the market for STRUBIAS materials derived from sewage sludge.

Germany and Austria have already taken up 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. In Austria, The draft Federal Waste Plan 2017 (Bundes-Abfallwirtschaftsplan) includes a ban of direct land application or composting for sewage sludge generated at Wastewater Treatment Plants with capacities of 20 000 people equivalents or above within a transition phase of 10 years. These wastewater treatment plans will have to recover the P from sludge targeting P contents below 20 g P / kg dry solids in the rest fraction, or have to deliver their sludge to sludge mono-incinerators. The P is then to be recovered from the sewage sludge.
ashes obtained. This regulation will cover 90% of the P contained in the Austrian municipal wastewater. In Germany, the new sewage sludge ordinance will make phosphorus 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 the P if the sludge contains more than 2% phosphorus /DS (dry solids) or have to incinerate the sludge in mono-incinerators. Land application of sludge will only be allowed under strict conditions for wastewater treatments plants < 50,000 p.e. The ~500 plants that are projected as subject to the requirement of P-recovery represent roughly 66% of the total P removed from German wastewater and transferred into the sludge. 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 until 2026. Current studies are on-going to determine the on-the-ground implementation (e.g. minimal recovery rates).

The implementation of the EU directive on waste water 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 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" (art. 1.3.). Italy, Greece and Ireland have not taken up 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 recovered phosphate salts and ash-based material 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 Waste Water 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 waste water treatment plants would be estimated at 360 kt of P (based on the sum of P in communal sewage sludge and urban waste water 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 impossibility 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 off-set by the increased revenues from the valorisation 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 landspreaded sewage sludge, and the possible synergies of incineration with energy recovery could drive shifts in sewage sludge management.

8.2.6.3 Trade-offs between process pathways

Phosphorus can be recovered upstream in the wastewater treatment facility as recovered phosphate salts or downstream from the resulting sludges and mono-incinerated sludge ashes (section 4). Basically, it is relevant to distinguish two relevant configurations:

i. The sludge is recovered downstream from the resulting sewage sludge treatment, after the mono-incineration of the produced sludge. In this case, most (~95%) of the incoming P in the wastewater can be recovered. Note that P-recovery technologies exist for sewage sludge ashes that have been produced using chemical precipitation and EBPR techniques (see section 4.2). P-recovery from the sludge can also occur after upstream P-recovery from the sludge liquor or digested sludge in EBPR treatment facilities with a recovery of 10-15% (first generation struvite production processes; see section 4). In this case, the resulting sludges will still contain sufficient P (>85%) to enable P-recovery in a present-day techno-economic perspective;

ii. Specific processes that recover P from pre-treated (e.g. after thermal hydrolysis) sludges either upstream in the waste water treatment process (e.g. Cambi process; that releases roughly 50% from the sludge matrix into the liquid sludge liquor) or
downstream on the generated sludge (e.g. Budenheim process) show P-recovery rates of about 40-50%. Such processes, however, produce sludges that contain only the remaining 50-60% of the P, making them less suitable for landspreading (low P/contaminant ratios) and economically viable P-recovery in the form of ashes.

With a view on the obligation to recover the dominant P fraction from municipal wastewaters in the Waste Framework Directive, option ii) is associated to a major disadvantage. The legislative proposals in Germany and Austria effectively require P-recovery rates of >50%, and thus restrict the viability of configurations described in point ii). For the estimation of the potential of STRUBIAS materials to substitute synthetic and mined P-fertilisers, only route i) is considered.

8.2.6.4 P-precipitation from sludge liquor and digested sludge at biological wastewater treatment plants

The Urban Waste Water Treatment Directive (91/271/EEC) requires that more advanced treatment (tertiary treatment) is required 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 sensitive area/catchment of sensitive area and therefore requires more stringent treatment. The territory of sensitive areas in EU-28 increased by 4.8% compared to the last Implementation Summary of the year 2001 (i.e. 4.8 % increase/decade). In 2011, tertiary treatment was in place for 89% of the load for EU-15 and for 27% of the generated load for 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 EU-15 and 24% of the total generated load for 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 tertiary treatment in the year 2030 is estimated at 83% and 26% for EU-15 and EU-13, respectively. Given that ~75% of the disposed sludge is generated in the EU-15 (Eurostat, 2017), a weighted average for the EU-28 of the relative wastewater load connected to tertiary treatment is estimated at 69%.

The most popular P removal techniques to remove P from wastewater treatment effluents are enhanced biological phosphorus removal (EBPR) and the more widely used chemical phosphorus removal (Chem-P) using iron or aluminium salts. 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 on sludge produced or people equivalents). The remaining 40% of the plants rely on EBPR or EBPR with Chem-P support, configurations that would enable the P-recovery as recovered P-salts from the digested sludge or the sludge liquor (see section 4). Excluding pre-treatments such as thermal hydrolysis, these processes have a recovery rate of
about maximal 15% and can provide substantial benefits for the functioning, maintenance and sludge handling for EBPR waste water treatment plant (improved sludge dewatering, reduced pipe clogging, etc.). Therefore, these processes are considered economically viable, especially if also the revenues from the sales of the recovered phosphate salt are taken into consideration. This route of P-recovery is technically impossible for wastewater treatment plants that rely exclusively on Chem-P techniques.

Box 6: Recovered phosphate salts from the sludge liquor and digested sludge at EBPR wastewater treatment plants.

To derive the substitution potential of P-fertilisers containing STRUBIAS materials through this route, following assumptions and calculations were made:

1) For the year 2030, the total incoming P in municipal waste water treatment plants would is estimated at 360 kt of P.

2) 69% of the EU-28 urban wastewaters will be processed with tertiary treatment. Struvites are produced at 100% of the EBPR wastewater treatment plants and EBPR plants with Chem-P support in Europe.

3) For the year 2030, the current share of the plants with tertiary treatment that partly rely on EBPR configurations is maintained (40%). At present, there is no clear technological, scientific or economic evidence that justifies a radical change in wastewater treatment plant configurations, for which reason only minor shifts can be expected in sludge that is treated in the year 2030. Given 1) and 2), EBPR wastewater treatment plants will treat about 99 kt P yr⁻¹.

4) The assumed P-recovery rate is 15% of the P present in the incoming P content of the wastewater.

5) The relative agronomic efficiency (RAE_{PUE}) of recovered phosphate salts relative to synthetic and mined P-fertilisers is assumed to be 1.05 (see section 5.2.2).

6) Considering 1) - 5), the P-recovery from the sludge liquor and digested sludge at EBPR wastewater treatment plants is estimated at 16 kt of P. It is estimated that these materials will be brought on 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, etc.) is unlikely.
8.2.6.5 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 such materials, possibly thanks to the simultaneous incorporation of such Fe/Al-rich ashes with more suitable input materials such as phosphate rock in the production process. Processes of high TRL level such as the acidulation process, the Ecophos process and thermo-reductive RecoPhos process (ICL) all produce inorganic intermediates ($\text{H}_3\text{PO}_4$, $\text{P}_4$) that can be used to produce conventional, mineral P-fertilisers (MAP, DAP, DCP, SSP, TSP, nitrophosphate, etc.). Given that these end-materials are associated to 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 to 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 be possible 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 the (cost-intensive) mono-incineration.

As indicated in section 8.2.6.2, it is estimated that 37% of all generated sludge will be treated via mono-incineration followed by P-recovery in the year 2030.

---

Box 7: Ash-based materials from mono-incinerated sewage sludge ashes

To derive the substitution potential of P-fertilisers containing STRUBIAS materials through this route, following assumptions and calculations were made:

1) The calculations are based on the full implementation of the provisions of the Waste Framework Directive (Directive 2008/98/EC) that requires that P is effectively recovered from waste water treatments, either through landspreading of the sludge or through the return of STRUBIAS materials to the agricultural field.

2) For the year 2030, the predicted P-content in sewage sludges corresponds to 95% of the P present in the incoming waste water minus the P contents that are already recovered upstream in the form of recovered P-salts (see section 8.2.6.4). Hence, the maximal total P in municipal sewage sludges is estimated as $(360 \, \text{kt P yr}^{-1} \times 0.95) - 15 \, \text{kt P yr}^{-1} = 327 \, \text{kt P yr}^{-1}$.
3) A total of 37% of the total P present in municipal sewage sludges will be mono-
incinerated, after which on average 95% of the P will be recovered from the ashes.

4) The recovered P will be used to produce water-soluble mineral P-fertilisers that have, per
definition, a 100% relative agronomic efficiency relative to their mined counterparts.

5) Given 1) to 5), a P-substitution potential of 115 kt of P yr\(^{-1}\) through this route is
estimated with ash-derivates as end-material of the production process.

8.2.7 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) 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 "recovered phosphate salts". To the
best of the authors' knowledge, these are currently the only TRL 7-9 processes that
manufacture P-fertilisers meeting the conditions as outlined for PFC 1. Nutrient recovery
processes from other residual waste water streams from food processing industries have not
been documented. The underlying reasons for this observation remain unknown, but may be
associated to the spatial scattering of the food processing facilities (i.e. lack of central
collection possibilities of the wastewaters and sludges), the chemical composition of the
waste water (a phosphate content < 50 mg P L\(^{-1}\) making them unsuitable for precipitation as
Ca- or Mg-phosphates), the transfer to urban waste water treatment plants, or the direct
return to agricultural land of the uncontaminated sludges and waste waters. With the
exceptions indicated for specific food industries, waste waters from other food processing
industries are mostly characterised by rather low phosphate contents, for which reason they
are unsuitable for P-recovery in the form of recovered phosphate salts. Therefore,
production processes from 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
coaagulants 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) will vary with wastewater characteristics
across dairy plants.
Box 8: Recovered phosphate salts from the food processing industry

To derive the substitution potential of P-fertilisers containing STRUBIAS materials through this route, following assumptions and calculations were made:

For the year 2030, there is a potential for P-recovery in the form of recovered phosphate salts from wastewaters from the potato industry and the dairy industry. P-precipitation from 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 was 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. Under 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 valorisation of recovered phosphate salts will provide a competitive advantage over chemical P-removal, at least for the largest facilities that process large amounts of waste waters of 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 realised.

8.2.8 STRUBIAS materials derived from the metallurgic industry
Slags are produced as residues from the metallurgic industry through a variety of techniques (see section 3.8). Due to their high Ca and P contents, slags from the iron and steel industry are currently applied on land as P-fertilisers or liming materials. 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 granulated to granulated blast furnace slag (Jewell and Kimball, 2014). The output mass of blast oxygen furnace slag per tonne of crude steel is 0.10–0.15 tonnes (Jewell and Kimball, 2014). Altogether, slags from the steel industry are produced in Europe in an amount of approximately **25 - 40 million tonnes** (Branca et al., 2014; European Commission, 2016a).

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 percent of the slags, representing > 800 000 tonnes, are used as for the production of **fertilising products** (European Commission, 2016a). 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. Slag can be used as a liming material or P-fertiliser production and micro-nutrients 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 content of Ca (25 – 30%) and some Mg. The relative agronomic efficiency of such materials remains unknown; the RAE$_{DMY}$ for electric arc furnace steel slags was estimated at 0.46 (see section 5.2.3; based on data from Bird and Drizo (2009)). However, the steel industry is currently testing processes that aim to increase the quality of the output materials (e.g. FehS/Salzgitter process; see section 4.2.2). With P contents varying between 4.4% and 9.6% (Gendebien et al., 2001), steel industry slags are thus nowadays an important P source for the European agriculture. Altogether, the use of steel industry slags in European agriculture is **estimated at about 26 kt P yr$^{-1}$** (i.e. 812 500 tonnes material x 7% P x 46% relative agronomic efficiency). This estimate is kept for the year 2030, but more information is requested from the STRUBIAS sub-group to make a more informed estimate in upcoming reports (see below).

The European steel association estimates that the potential use of slags in agriculture could reach 1 500 000 tonnes across the EU. This has to be considered as a low estimation as the UK and Poland, which have still a strong steel industry, have not replied to the survey (European Commission, 2016a). It remains, however, to be evaluated to what extent the P-fertilisers derived from the steel industry will meet the proposed levels of metals/metalloids that will be set at CMC or PFC level.

**Question 4 to the STRUBIAS sub-group:**

Please provide an outlook for slags from the steel industry that are intended for use in agriculture. More specifically, information is requested on following aspects:

- Evolution in supply and demand for steel industry slags that are used as P-fertilisers in agriculture, as well a realistic outlook for the market outlook for the year 2030. Please express numbers in absolute tonnes of material per year, indicating the P concentration, or in kt P per year. Please provide separate datasets for different types of slags (blast furnace slag, basic oxygen slag, etc.).
• Data on the content of metals/metalloids (specifically B, Ba, total Cr, Cr (VI), Co, Cu, Hg, Mn, Mo, Ni, Pb, Sb, and V) in the different types of slag. Please provide full datasets or descriptive statistics indicating the 10th, 25th, 50th, 75th and 90th percentile of the distributions.
• At present, it remains unclear if the production process is adapted in order improve the quality of the resulting slag, or if any supplementary processing is performed on the raw slags obtained to increase their value as a fertilising material. Please provide any relevant information.
• Information on the relative agronomic efficiency of different output STRUBIAS materials.
• Evolution of the average sales prices of steel industry slags in the last decade and future outlook (Euro per tonne material and P concentration, or Euro per tonne P).

8.2.9 Other process pathways and eligible materials not considered

To the best of the authors' knowledge, all P-fertilisers as defined by the minimum P₂O₅ contents for PFC 1 produced through STRUBIAS production processes from eligible input materials are covered in the sections 8.2.4 - 8.2.8. Other STRUBIAS production processes (e.g. thermochemical conversion materials from biowaste, pulp and paper industry sludge, woody biomass, etc.) may lead to marketable fertilising products, but likely under different PFC categories (e.g. liming materials, soil improver, biostimulants, etc.). Therefore, these production processes will be covered in the separate section 8.3.

Question 5 to STRUBIAS sub-group:

Please indicate any additional process pathways that result in the formation of P-fertilisers (pursuant definition and criteria of P-fertilisers as given for PFC 1 in the proposal for the Revised Fertiliser Regulation) and describe their market outlook in term of volumes of P that could be produced by the year 2030. The proposed pathways should meet following conditions:

• Derived from eligible input materials for each of the three STRUBIAS material groups as outlined in JRC Interim Report on nutrient recovery rules as distributed in May 2017 or derived from eligible input materials that were not listed, but are nonetheless in line with environmental and human health safety aspects and agronomic efficiency as indicated by techno-scientific evidence.
• Associated to a technological readiness level of stage 6-9.
• Realistic from an economic point of view based on the current and expected costs/gate fees for input materials, production processes, and revenues of generated output materials.
8.2.10 Conclusion

8.2.10.1 Market estimate for TRL 7-9 processes under the anticipatable 2030 legislative framework

The estimates given in sections 8.2.4 - 8.2.8 are subject to many uncertainties and as a result should only be interpreted with the necessary precaution. 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 waste waters, slaughter residues, the food processing industry, and the metallurgic industry a total of 302 kt of plant-available P could be recovered as recovered phosphate salts and ash-based materials (Table 12). Although there could be a market for pyrolysis materials, the 2030 market potential remains indeterminate due to the lack of comprehensive test results on agronomic efficiency for this STRUBIAS group. Therefore, it remains unclear if 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.

STRUBIAS materials expected to be on the market in 2030 are almost exclusively derived from municipal waste waters (43%), manure (27%) and slaughter residues (20%) (Table 12). 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 heavily determining the STRUBIAS market and trade potential.

It is estimated that ash-based materials will be the dominant STRUBIAS output material group with a total estimated production volume of 236 kt P yr⁻¹ or 78% of all STRUBIAS materials (Table 13). It is estimated that a major share of ash-based materials will be brought on 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 to these products in the conventional agricultural market. It is estimated that also the market for recovered phosphate salts (66 kt P yr⁻¹; 22% of the total plant-available P recovered as STRUBIAS) will grow exponentially in the coming years (Table 13), 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 been 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-dense 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 the 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 organic nature.

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 12). With ash-based materials 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, it should be noted that incineration ashes will also be shipped to facilities in other European regions (e.g. EcoPhos manufacturing site in Bulgaria).
Table 12: 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 (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>
<thead>
<tr>
<th>process pathway</th>
<th>section</th>
<th>recovered STRUBIAS material</th>
<th>recovered total P (kt P yr(^{-1}))</th>
<th>relative agronomic efficiency (%)</th>
<th>recovered bio-available P (kt P yr(^{-1}))</th>
<th>share of total bio-available P recovered (%)</th>
<th>regional distribution in Europe</th>
<th>major market drivers that will stimulate the market</th>
</tr>
</thead>
<tbody>
<tr>
<td>P-precipitation after anaerobic digestion dominantly from manure</td>
<td>8.2.4.3.</td>
<td>recovered phosphate salts</td>
<td>45</td>
<td>105</td>
<td>48</td>
<td>16</td>
<td>livestock dense regions having soils showing P-excess and water bodies subject to P-eutrophication</td>
<td>reducing externalities due to manure excess (water quality), renewable energy targets</td>
</tr>
<tr>
<td>thermal oxidation of solid manure fractions</td>
<td>8.2.4.4.</td>
<td>ash-based materials</td>
<td>39</td>
<td>90</td>
<td>35</td>
<td>12</td>
<td>livestock dense regions having soils showing P-excess and water bodies subject to P-eutrophication</td>
<td>reducing externalities due to manure excess (water quality), renewable energy targets</td>
</tr>
<tr>
<td>thermal oxidation of slaughter residues</td>
<td>8.2.5.3.</td>
<td>ash-based materials</td>
<td>65</td>
<td>92</td>
<td>61</td>
<td>20</td>
<td>livestock dense regions</td>
<td>synergies with energy recovery from animal by-products</td>
</tr>
<tr>
<td>P-precipitation from biological municipal wastewater treatment plants</td>
<td>8.2.6.4.</td>
<td>recovered phosphate salts</td>
<td>15</td>
<td>105</td>
<td>18</td>
<td>5</td>
<td>regions and cities that apply tertiary treatment at water treatment plants</td>
<td>increased share of biological waste water treatment plants</td>
</tr>
<tr>
<td>thermal oxidation of sewage sludge</td>
<td>8.2.6.5.</td>
<td>ash-based materials (derivatives of well-known chemical composition (e.g. DAP, MAP))</td>
<td>115</td>
<td>100</td>
<td>115</td>
<td>38</td>
<td>regions with low acceptance for sewage sludge landspreading and mono-incineration capacity</td>
<td>focus on increased soil protection from metal accumulation and nutrient excess, ban on landfiling of biodegradable waste</td>
</tr>
<tr>
<td>P-precipitation from the food processing industry</td>
<td>8.2.7</td>
<td>recovered phosphate salts</td>
<td>2</td>
<td>105</td>
<td>2</td>
<td>1</td>
<td>regions with dairy and potato processing industries</td>
<td>focus on externalities and costs associated to industry waste water discharges</td>
</tr>
<tr>
<td>thermal oxidation of iron ores</td>
<td>8.2.8.</td>
<td>ash-based materials (slags)</td>
<td>56</td>
<td>46</td>
<td>26</td>
<td>9</td>
<td>regions with high densities of steel making companies</td>
<td>increased costs associated to landfiling</td>
</tr>
<tr>
<td>pyrolysis materials of solid manure fractions</td>
<td>8.2.4.4.</td>
<td>pyrolysis materials</td>
<td>indeterminate</td>
<td>indeterminate</td>
<td>indeterminate</td>
<td>indeterminate</td>
<td>livestock dense regions having soils showing P-excess and water bodies subject to P-eutrophication</td>
<td>reducing externalities due to manure excess (water quality), soil quality improvement, increased consumer and market acceptance</td>
</tr>
<tr>
<td>pyrolysis of slaughter by-products</td>
<td>8.2.5.3.</td>
<td>pyrolysis materials</td>
<td>indeterminate</td>
<td>indeterminate</td>
<td>indeterminate</td>
<td>indeterminate</td>
<td>potentially somewhat more important in livestock dense regions</td>
<td>increased consumer and market acceptance</td>
</tr>
</tbody>
</table>

**OVERALL TOTAL** | **337** | **302** | **100**
Table 13: Market estimate for TRL 7-9 processes under the anticipatable 2030 legislative framework as aggregated per STRUBIAS material group and per input material (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>
<thead>
<tr>
<th>Aggregated per STRUBIAS material group</th>
<th>P content (kt P yr⁻¹)</th>
<th>relative contribution (% of total)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Recovered phosphate salts</td>
<td>66</td>
<td>22</td>
</tr>
<tr>
<td>Ash-based materials</td>
<td>236</td>
<td>78</td>
</tr>
<tr>
<td>Pyrolysis materials</td>
<td>indeterminate</td>
<td>not considered</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Aggregated per input material</th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td>manure</td>
<td>78</td>
<td>27</td>
</tr>
<tr>
<td>municipal wastewater</td>
<td>131</td>
<td>43</td>
</tr>
<tr>
<td>slaughter residues</td>
<td>61</td>
<td>20</td>
</tr>
<tr>
<td>food processing industry</td>
<td>7</td>
<td>1</td>
</tr>
<tr>
<td>metallurgic industry</td>
<td>28</td>
<td>9</td>
</tr>
</tbody>
</table>
2.10.2 Further market stimulations

The inclusion of STRUBIAS CMCs in a revised EU Fertiliser Regulation 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 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. The P use efficiency of STRUBIAS materials is typically larger than for these organic P sources because the release of P can be better synchronized with plant needs, thus reducing the scope for its loss to deeper soil layers and surrounding water bodies. Moreover, benefits are associated to 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 recovered phosphate salts and ash-based materials 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. land spreading).

[Note that preliminary information on this topic provided in the Background Document of the Kick-off Meetings on such impacts is currently under validation by JRC. These aspects will be further elaborated in a separate section as a part of the Final Report of the impact assessment study.]

The below-mentioned scenarios 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 landspreaded 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 waste waters would increase from 112 kt P yr\(^{-1}\) to 170 kt yr\(^{-1}\) (+ 58 kt yr\(^{-1}\)).

2. As indicated in section 8.2.4, 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 Waste Water 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, also about 924 kt P yr\(^{-1}\) of P 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 is to apply a pre-treatment to manure (e.g. moderate acidification, and the even more promising thermal hydrolysis technology) 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 1377 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.

Sulphuric or other 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). However, careful consideration must be given to the soil and soil pH as application of acidic solid digestate will not always be acceptable, potentially limiting the large scale operation ability of this acidification pre-treatment. The thermal hydrolysis process is a high-pressure, high-temperature steam pre-treatment 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 cell structure within the biomass. As the organic matter is presented to 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 case the pre-treatment process could be applied on manures that will be further processed by anaerobic digestion, an increase in 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 Na₂SO₄; AshDec process) or pyrolysed. The char produced by the process can be used as a soil amendment 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 3.4.2, 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, the fish catches and aquaculture production in Norway equal about 45% of the total fish residues in Europe. Fish excreta and non-digested feed from land-based aquaculture also form a P-source that can potentially be recovered as STRUBIAS materials. The STRUBIAS sub-group indicated that these residues are already used for the production of P-fertilisers in Norway. Giving 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).

8.2.10.3 Substitution effect

Results from the previous section will be used to estimate the substitution effect of mined and synthetic fertilisers by fertilising products containing recovered phosphate salts, pyrolysis materials and ash-based materials 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.

The 2030 apparent consumption of synthetic and mined P-fertilisers is estimated at 1220 kt P yr⁻¹ (see section 8.1.3.2). By summing the values given in section 8.2.10.1, it is estimated that a total of 302 kt P yr⁻¹ could be recovered. Hence, the substitution effect from STRUBIAS materials for the year 2030 is estimated at 25%.

In case the further market stimulations for sewage sludge and manure as outlined in section 8.2.10.2 would materialise, the substitution effect for the EU-28 could increase to 40%.

In summary, it is estimated that 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 recovered phosphate salts and ash-based materials of 25-40%. Moreover, it should be noted that pyrolysis materials have not been included in this assessment because an initial entry is considered more likely into the organic farming sector. The on-the-ground use of pyrolysis materials might enable the detailed testing of the agronomic efficiency of pyrolysis materials, possibly leading to more refined and precise estimates given in section 5.2.4. In case these results lead to increased market and consumer
confidence for the use of these materials in conventional agriculture, the substitution potential will further increase.

8.3 Market outlook for non-fertiliser PFCs derived from STRUBIAS materials for the year 2030

STRUBIAS materials – and more specifically ash-based materials and pyrolysis 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.

Estimating a 2030 market for such fertilising materials is extremely challenging and associated with large uncertainties.

Question 6 to the STRUBIAS sub-group:

Please provide an outlook for STRUBIAS materials other than P-fertilisers, as well as their targeted PFC entry in the revised Fertiliser Regulation (i.e. liming material, soil improver, etc.). Please express numbers in absolute tonnes of material per year, and situate the numbers relative to the total PFC volumes applied in the EU-28 agricultural market. Also information on sales prices/gate fees of the end-materials is welcomed. More specifically, information is requested for following two materials:

- C-rich pyrolysis materials in conventional agriculture, organic farming, and greenhouse farming (i.e. as a growing media);
- Ashes generated by the forest-based industry, including the pulp and paper industry;

8.4 Conclusions on market potential for STRUBIAS materials

Based on the preliminary market assessment for the year 2030, significant market demand and trade is expected for all three STRUBIAS material groups in different segments of the EU agricultural sector. A stable legal framework for all three STRUBIAS material groups is desirable in order to provide a material quality benchmark for producers and consumers of P-fertilising materials containing STRUBIAS.
9 General conclusion
[to be completed for the final report]
10 Questionnaire

10.1 Objective of the questionnaire

The objective of the questionnaire is to validate and, if necessary, correct and complement the techno-scientific information that provides the foundation for the proposed STRUBIAS material requirements outlined in this Interim Report. It is requested to concentrate review efforts on the sections 5 - 8, as your input on the sections 3 and 4 has already been taken into consideration when processing your feedback on the Background Document for the STRUBIAS Kick-off Meeting.

10.2 Procedure

As outlined in the Rules of Procedure of the STRUBIAS sub-group, the sub-group member representatives shall actively collect information and deliver fact-based opinions on the questionnaires that form part of the written consultations. It is important that STRUBIAS sub-group member representatives provide a consolidated opinion that is in line with the views of the member organisations and stakeholders they represent.

Unfortunately, the JRC is not able to accept responses and opinions from organisations and individual persons other than official STRUBIAS member organisations and their selected representatives. The JRC recommends any third party organisations or persons interested in contributing to this work to contact one of the member organisations of the STRUBIAS sub-group. These STRUBIAS members carry the full responsibility for the quality of the information sent to the JRC and may therefore decide to take any external input on board in their reply, or not, after careful consideration and thorough quality-checking.

The STRUBIAS sub-group members shall support their opinions with objective and evidence based arguments. In case of disagreement with the present proposals for nutrient recovery rules, sub-group members shall provide alternative proposals for alternative formulations along with supporting robust techno-scientific data and information.

Sub-group members shall use the channels provided by the Commission for discussion and information exchange. The preferential route for submitting non-confidential information is via the CIRCABC platform as this will facilitate a structured information exchange amid STRUBIAS members. Detailed instructions on how to access the CIRCABC STRUBIAS Interest Group were distributed to sub-group members via e-mail.

Please upload any information in the folder/space entitled "Interim Report – Market study", and then select the matching sub-folders "Written feedback from sub-group". The document

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5 The list of Members of the STRUBIAS sub-group can be found in the Register of Commission Expert Groups Fertilisers Working Group (E01320) (http://ec.europa.eu/transparency/regexpert/index.cfm?do=groupDetail.groupDetail&groupID=1320) Tab "Subgroups" Subgroup of the Commission expert group on Recovery Rules for Fertilising Products
name should start with the acronym of the member organisation. Please note that all information that is uploaded in this CIRCABC folder is publically available. Techno-scientific literature can be uploaded in the corresponding sub-folder that is only accessible to STRUBIAS sub-group Members.

The JRC prefers to receive publically available information in order to support a transparent information exchange process. Nevertheless, it is accepted that some data cannot be made public and should be handled in a confidential manner. If only the data provider or data source is confidential, but not the data itself, it is desirable that member organisations anonymise the data provider/source and upload the document on CIRCABC as indicated above. Confidential data that cannot be publicly shared in any form should be sent via e-mail to JRC-IPTS-FERTILISERS@EC.EUROPA.EU. The document name should include the acronym of the organisation followed by the word "confidential".

The JRC is pleased to take into account any feedback on the questionnaire received from the STRUBIAS sub-group members until the deadline of Thursday 15 March 2018. We guarantee that any input received by the deadline will be taken into account for the further work.
10.3 Questions

Section A: General question

A.1. Have you noticed any **incorrect or obsolete techno-scientific information** in the Interim Report that has an important influence on the market for STRUBIAS materials? If your observation involves an alternative proposal for the STRUBIAS material requirements, please indicate, substantiate and upload supporting techno-scientific information.

Provide your feedback in a structured, tabular format with following headings: observation, page/line numbers and section in the document, correction and/or alternative proposal, techno-scientific rationale that supports the comment raised, reference to techno-scientific data.

<table>
<thead>
<tr>
<th>observation</th>
<th>location in document</th>
<th>correction/alternative proposal</th>
<th>techno-scientific rationale that supports the comment raised</th>
<th>reference to techno-scientific data</th>
</tr>
</thead>
<tbody>
<tr>
<td>e.g. sales prices for mineral P-fertilisers are underestimated by 30%.</td>
<td>e.g. section 2.3.7.2 (line 2864)</td>
<td>e.g. sales prices vary from X to Y EUR</td>
<td>The data found in IFA report (2017).</td>
<td>e.g. IFA. (2017) has been uploaded on CIRCABC</td>
</tr>
</tbody>
</table>
Section B: Specific questions and further data

Please note that all the queries of this section correspond to the questions given in specific sections of the document.

Question 1 on sales prices of STRUBIAS materials on the market (section 7.1.3):

Please provide an indication of:

a) the sales prices of STRUBIAS materials and mono-incineration ashes on the market (Euro per tonne material and P concentration, or Euro per tonne P; indicate if prices are "Free on Board (FOB)" or "CFR (Cost and Freight)" with an indication of the price for transport).

b) Indicate also the physical form of the material (powder, granules, other relevant physical parameters), and to whom materials are sold (blending companies, retailers, end users, etc.).

c) If available, please provide an evolution of the average sales prices in the last decade in order to determine the main factors affecting fertiliser price and their relative importance.

Question 2 on compliance costs (section 7.3):

Please provide information on the following elements that form part of the compliance costs:

a) Cost for REACH registration for fertiliser end-material that will be brought on the market.

b) Cost for compliance under already existing national end-of-waste or similar regimes that enable a market entry for fertilising products derived from STRUBIAS materials.

c) Estimated cost for compliance for P-fertilisers derived from primary raw materials.

d) For facilities that process waste-based materials, information on the cost associated to acquiring waste permits in different EU Member states for non-hazardous and hazardous waste materials. Notably, the costs associated to complying with the obligation for an establishment or undertaking carrying out waste management operations to have a permit or to be registered in accordance with Article 23-26 of the Waste Framework Directive 2008/98/EC.

e) Cost of sampling and analysis through accredited laboratories:

Recovered phosphate salts:

- Nutrients: P, Ca, Mg, citric-acid P
- Metals and metalloids: As, Cd, Cr (total), Cr (VI), Cu, Hg, Ni, Pb, Zn
- Persistent organic pollutants: PAH16
- Biological pathogens: E. coli or Enterococcaceae & Salmonella spp.
- Others: macroscopic impurities, dry matter content, particulate matter < 100 µm.
Ash-based materials:
- Nutrients: P, K, Ca, Mg, S, citric-acid P
- Metals and metalloids: As, Cd, Cr (total), Cr (VI), Cu, Hg, Ni, Pb, Zn, B, Ba, Co, Mn, Mo, Sb, V
- Persistent organic pollutants: PAH16, PCB, PCDD/F
- Others: pH and neutralising value

Pyrolysis materials:
- Major elements: C, Corg, P, K, Ca, Mg, S
- Metals and metalloids: As, Cd, Cr (total), Cr (VI), Cu, Hg, Ni, Pb, Zn, Ba, Co, Mo, Sb, and V
- Persistent organic pollutants: PAH16, PCB, PCDD/F
- Biological pathogens: E. coli or Enterococcaceae & Salmonella spp.
- Other: pH, neutralising value, macroscopic impurities, particulate matter < 100 µm, particle density, volatile organic matter, specific surface area earthworm avoidance test (ISO 17512)

Where analysis packages are available (e.g. sampling + analysis of a series of metals, PAH16, PCB and PCDD/F), please clearly state what the package contains and its cost.

f) Measurement standards currently applied (national standards, ISO/EN standards, etc.)

Question 3 on possible economic benefits and drawbacks (section 7.4):
Please provide information, preferably in a quantitative manner, on following possible economic benefits and drawbacks of producing fertilising products containing STRUBIAS materials compared to equivalent mined and synthetic inorganic fertilising products.

a) reduced waste compliance costs (e.g. changes in the economic valuation of sewage sludge ashes, etc.);
b) reduced externalities (e.g. avoided costs due to eutrophication, positive effects on human health due to reduced contaminant levels, etc.);
c) potential job creation in production and downstream fertiliser distribution and farmer's cooperatives; please relate expected STRUBIAS production volumes to number of persons employed.
d) impacts on the rural economy;
e) benefits of restoring soil organic carbon for soil fertility;
f) cost associated to new logistics for recovered nutrient products;
g) implications for the restructuring the production and distribution of fertilising products;
h) agricultural equipment adaptations.
**Question 4 on slags from the metallurgic industry (section 8.2.8):**

Please provide an outlook for slags from the steel industry that are intended for use in agriculture. More specifically, information is requested on following aspects:

a) Evolution in supply and demand for steel industry slags that are used as P-fertilisers in agriculture, as well a realistic outlook for the market outlook for the year 2030. Please express numbers in absolute tonnes of material per year, indicating the P concentration, or in kt P per year. Please provide separate datasets for different types of slags (blast furnace slag, basic oxygen slag, etc.).

b) Data on the content of metals/metalloids (specifically B, Ba, total Cr, Cr (VI), Co, Cu, Hg, Mn, Mo, Ni, Pb, Sb, and V) in the different types of slag. Please provide full datasets or descriptive statistics indicating the 10th, 25th, 50th, 75th and 90th percentile of the distributions.

c) At present, it remains unclear if the production process is adapted in order improve the quality of the resulting slag, or if any supplementary processing is performed on the raw slags obtained to increase their value as a fertilising material. Please provide any relevant information.

d) Information on the relative agronomic efficiency of different output STRUBIAS materials.

e) Evolution of the average sales prices of steel industry slags in the last decade and future outlook (Euro per tonne material and P concentration, or Euro per tonne P).

**Question 5 on additional STRUBIAS pathways that should be considered for the 2030 market assessment (section 8.2.9):**

Please indicate any additional process pathways that result in the formation of P-fertilisers (pursuant definition and criteria of P-fertilisers as given for PFC 1 in the proposal for the Revised Fertiliser Regulation) and describe their market outlook in term of volumes of P that could be produced by the year 2030. The proposed pathways should meet following conditions:

a) Derived from eligible input materials for each of the three STRUBIAS material groups as outlined in JRC Interim Report on nutrient recovery rules as distributed in May 2017 or derived from eligible input materials that were not listed, but are nonetheless in line with environmental and human health safety aspects and agronomic efficiency as indicated by techno-scientific evidence.

b) Associated to a technological readiness level of stage 6-9.

c) Realistic from an economic point of view based on the current and expected costs/gate fees for input materials, production processes, and revenues of generated output materials.
Question 6 on market aspects for STRUBIAS materials other than P-fertilisers (section 8.3):

Please provide an outlook for STRUBIAS materials other than P-fertilisers, as well as their targeted PFC entry in the revised Fertiliser Regulation (i.e. liming material, soil improver, etc.). Please express numbers in absolute tonnes of material per year, and situate the numbers relative to the total PFC volumes applied in the EU-28 agricultural market. Also information on sales prices/gate fees of the end-materials is welcomed. More specifically, information is requested for following two materials:

a) C-rich pyrolysis materials in conventional agriculture, organic farming, and greenhouse farming (i.e. as a growing media);

b) Ashes generated by the forest-based industry, including the pulp and paper industry;
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