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Sustainable Phosphorus Use

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EXECUTIVE SUMMARY

Sustainable Phosphorus Use

This In-depth Report provides an overview of the current state of scientific knowledge on the phosphorus challenge and research on the sustainable use of this important element. There are clear, well-documented reasons for more sustainable use of phosphorus across the EU and globally. We know that phosphorus is an essential and finite resource with no substitute. We also know that the EU is reliant on imports and, as good quality sources of phosphorus diminish, we will become increasingly dependent on phosphorus reserves that are less accessible and more polluted with toxic elements, such as cadmium and uranium.

An algal bloom in a lake. If excess phosphorus reaches water bodies, the growth of algal blooms may be triggered, which can deplete water-borne oxygen and release toxic compounds.

Although we are not certain of the magnitude of phosphorus reserves, we do know they are concentrated in a few countries. It is estimated that Morocco has the largest continental reserves of phosphorus and these reserves are also amongst those with high levels of cadmium.

Developments in industry, agriculture, waste handling and lifestyle have massively reduced the capacity for phosphorus to be cycled effectively by society and the environment via natural geological processes. The major source of phosphorus used in fertiliser is phosphate rock, which we mine in vast quantities, more than can be replaced by the slow geological cycle. Instead of being recycled, phosphorus moves linearly from its application on land to its consumption on the plate and, at nearly every stage, there are losses. Scientific analyses of phosphorus flows, which study the amounts of phosphorus moving through society and the environment, have indicated that we waste about 25 million tonnes globally (Clift & Shaw, 2012) and 1.25 million tonnes at an EU level (van Dijk *et al.*, preliminary results in preparation).

There is also scientific evidence of the negative impacts of phosphorus on the environment. Past excessive use of fertiliser and intensification of livestock farming mean that phosphorus has been running off the land and polluting water bodies. Wastewater from our sewage systems also contains phosphorus from the food and detergents we have consumed and, depending on the effectiveness of wastewater treatment, this can reach rivers, seas and oceans. Eutrophication, algal blooms and dead zones have been observed in parts of the EU from the Baltic to the Mediterranean Sea, as well as more globally. There are also known negative environmental impacts from mining, the processing of mineral fertiliser and the increasing level of pollutants within the fertiliser itself, such as cadmium and uranium.

Although we cannot turn back the clock, we can improve the sustainable use of phosphorus and there is increasing knowledge on the methods and technologies with which to do this. These range from the low-hanging fruits of efficient farming techniques, to the more technologically-intensive recovery of phosphorus from wastewater. Simply, there needs to be a choice between the options or combinations of options, to provide an economically feasible solution.

Again, here we have scientific knowledge and practical experience to inform decisions. Analyses of phosphorus flows not only give us the tools to identify where major losses occur, but help us evaluate the impact of different options and decisions to curb these losses. There is a significant amount of research on the effectiveness of various methods to recover phosphorus, but we need more information on their real-world feasibility. Evaluation is necessary but this requires the initiation of more projects in the first place. Good examples of large-scale real-life evaluations include those being undertaken by the P-Rex project, which aims to bridge academic knowledge and practical application to demonstrate and assess options for full-scale phosphorus recovery.

Although the technology and amount of research is growing, successful implementation of initiatives promoting sustainable phosphorus use will require policy support and legislation to encourage initiatives. This could be through the use of targets and preventative measures, but also by creating the enabling conditions for markets in waste and recycled phosphorus products to develop. For example, a compulsory measure for fertiliser manufacturers to use a certain blending ratio to mix traditional fertiliser with fertiliser made from recovered phosphate could encourage more sustainable phosphorus use.

Currently, policy response to the phosphorus challenge is patchy across the EU. To a certain extent, this is because the factors that stimulate phosphorus recovery vary between countries. The Netherlands is one of the frontrunners in terms of policy response. However, the conditions that initially caused the Netherlands to take action, i.e. excess phosphorus and water pollution by intensive animal production systems and feed imports, also exist in many other countries. Denmark has shown similar responses to the Netherlands, whilst Sweden has brought in targets on phosphorus recovery for 2015 and Germany may introduce recovery targets specifically aimed at wastewater treatment plants.

Measures must be chosen according to the economic, social and environmental context of each EU Member State, but there is a need for more harmonisation in the prioritisation of this issue and a clear political strategy. This will require science to work in close co-operation with policy and industry.

The Netherlands has already developed greater cooperation between industry, science and policy in the Dutch Nutrient Platform, which brings together private companies, NGOs and academic institutions to contribute to the transition to more sustainable use of nutrients[1]. A similar platform concept has been introduced at a larger scale in the European Sustainable Phosphorus Platform, which is an initiative of joint European forerunners[2]. Such platforms and initiatives can also bring together research that is scattered across sectors, such as agriculture, water, waste, environment and industry.

Alongside this, the EU has produced its Consultative Communication[3] on the Sustainable Use of Phosphorus, which aims to launch a debate on phosphorus use and ways to make it more efficient. More harmonisation is also needed in data collection and research, which can be integrated in the Horizon 2020 framework. The analysis of phosphorus flows through the society and identification of where major losses occur is useful in deciding where action can be taken and comparing different options.

However, in order to inform these decisions effectively, better data are needed, alongside harmonisation of methods for making meaningful comparisons between crops, river basins, regions, countries and options for more sustainable phosphorus use. There is also a need for better understanding and consideration of embodied or 'virtual' phosphorus in imported food and animal feed. In general, applicable comparisons and life cycle analyses of different options for sustainable phosphorus use are needed to assess financial, environmental and social impacts. Similar to the water footprint concept, a phosphorus footprint could be a valuable tool to monitor changes and keep track of progress.

Although better data and further study will help inform policy, existing research and analysis is already calling for action. Currently, the EU imports virtually all its phosphorus and, as reserves diminish, this will be the case for other countries that have large unsustainable phosphorus demands. The US is already heavily dependent on imports and there is likely to be increasing dependence on imports in China. As such, the importers' market will become increasingly pressurised. Past examples of the volatility of the phosphorus market, such as the 800% price spike in 2008 and the subsequent imposition of export taxes by China, have demonstrated the precariousness of this market. By taking action now in the reuse and recycling of phosphorus, the EU can become an active player in the secondary phosphorus market and maintain some control of this valuable resource for its own use.

[1] See http://www.nutrientplatform.org/
[2] See http://www.phosphorusplatform.eu/
[3] See http://ec.europa.eu/environment/natres/phosphorus.htm

Introduction

Phosphorus is essential for all life on the planet. It is vital to the structure of genetic building blocks (DNA and RNA), the production of cell membranes, energy supply, the formation of seeds and fruit in plants and many other biological processes. However, although we can be certain about the essential quality of phosphorus, there is uncertainty about its sustainability in the future.

Our bodies need a minimum of 0.6 to 0.7 g of phosphorus per day (van Rossum *et al.*, 2011) but our actual dietary intake from food is closer to 2-3 g of phosphorus per day (Flynn *et al.*, 2009). We obtain phosphorus by consuming plant and animal products, whilst plants obtain phosphorus from the soil. In the past the soil was naturally replenished through animal excretion and decomposition of organisms alongside the natural weathering of rocks. This cycle remained relatively balanced but, with the introduction of manufactured fertilisers[4] containing mined phosphate[5] to improve soil fertility, a number of environmental and sustainability issues have emerged.

For some time there has been recognition of the impact of phosphorus escaping from the land and entering aquatic ecosystems. This causes eutrophication and algal blooms that can, in extreme cases, lead to 'dead zones' due to a lack of oxygen. More recently, another significant consequence of the use of mineral fertiliser has become apparent: the declining reserves of good quality phosphorus and the repercussions of this on food security.

In its natural form phosphorus only exists as phosphate rock at the surface of the Earth's crust and, unlike elements such as carbon and nitrogen, it is finite and non-renewable. No other element can perform the same role as phosphorus and, since we have grown to be dependent on phosphorus to improve food security, shortages have economic, social and environmental implications. Due to data limitations and restrictions, it is difficult to ascertain the true extent of global phosphate rock reserves (UNEP, 2011). However, what is known is that the future supply of phosphate rock will be of lower quality, less accessible and unequally distributed around the world.

As such, there is a need to enhance phosphorus security and sustainable management of existing phosphorus reserves. This means ensuring access to sufficient phosphorus to produce enough food to feed the growing global population while maintaining ecological integrity and the livelihoods of farmers (Cordell *et al.*, 2012). The issue can be approached from both the demand and supply side of the equation, by promoting more efficient use of phosphorus and improving its recovery and reuse.

Research has a vital role to play in assessing the extent of global reserves, quantifying phosphorus pathways and flows, identifying technological and management solutions and evaluating different options for sustainability (Hilton *et al.*, 2010; Van Vuuren *et al.*, 2010). More than 164,000 scientific publications have been published about phosphorus since the early 1970s (Pellerin *et al.*, 2013b). Recently there has been a surge of analysis and commentary on sustainable phosphorus and several special issues have been published by a range of journals, such as Chemosphere, Proceedia Engineering, Current Opinion in Biotechnology and Plant and Soil. However, it is only recently that there have been publications on phosphorus as a non-renewable resource (see Figure 1) and its links to food security (see Figure 2). This indicates that the scientific community is now researching the phosphorus challenge from a range of different and important perspectives (Pellerin *et al.*, 2013b).

This In-depth Report from Science for Environment Policy aims to give an overview of existing research from these different perspectives within a DPSIR (Drivers, Pressures, State, Impacts and Responses) framework, indicating possible policy interventions and highlighting research gaps.

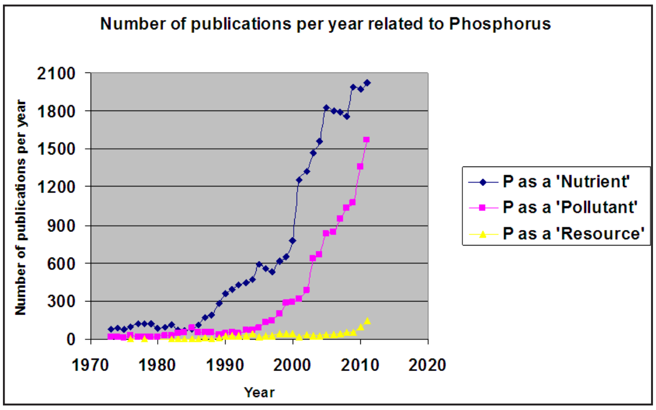

Figure 1: Number of publications per year related to phosphorus as a nutrient, pollutant and resource (Pellerin *et al*, 2013)

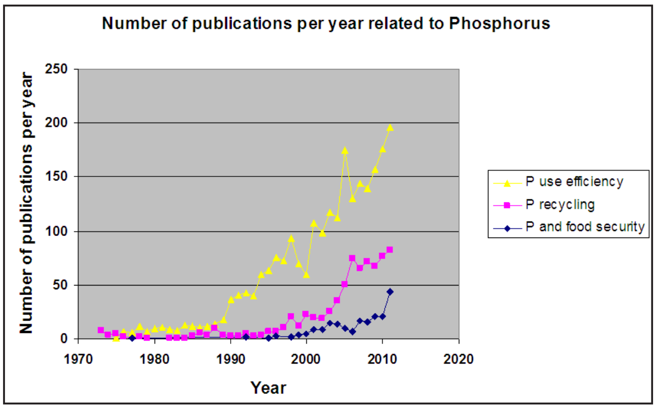

Figure 2: Number of publications per year related to phosphorus use efficiency, recycling and food security (Pellerin *et al*, 2013)

[4]Manufactured fertilisers containing mined phosphorus (as opposed to more natural fertilisers such as manure etc.) will be referred to as mineral fertilisers throughout this report.
[5]Phosphate is naturally occurring form of phosphorus, often found within rocks, which is mined and processed to make mineral fertiliser.

1. Drivers of phosphorus consumption

Over the last century, there have been several societal shifts that have driven the growth in the use of mineral fertilisers containing phosphorus and the reduction in the recycling of phosphorus. Although we cannot undo past developments, it is helpful to be aware of these influences in order to identify options for intervention and consider future trends. Past, current and possible future DRIVERS will be discussed in this section.

1.1 Expanding populations

Probably the most overarching driver of phosphorus consumption is population growth and the accompanying trends in distribution of the global population. Eighty-two per cent of the world's 6.8 billion people live in developing regions (UN, 2009). This expanding population means more mouths to feed, often in places where soil fertility is poor and crops need more fertiliser. Three quarters of Africa's farmland suffers severe soil degradation due to erosion and the loss of vital nutrients from the soil (Henao & Baanante, 2006). Places such as Ethiopia and sub-Saharan Africa have been highlighted as regions where the soil is becoming increasingly depleted in nutrients (Weikard & Seyhan, 2011).

Eradicating extreme poverty and hunger is one of the eight UN Millennium Development Goals that are due to be achieved in 2015, making food security an international priority. Due to transportation improvements, developing countries can now readily acquire mineral fertilisers, although it is often at very high costs, which will only rise as phosphorus becomes scarcer (see Section 4). Africa's agricultural productivity has not improved in the last 40 years and farmers will need better access to fertilisers in order to maintain soil fertility (Connor, 2006).

In the developed world, soils have become saturated with phosphorus and other nutrients from the constant use of mineral fertiliser and manure that was over-and-above the nutrient requirements of the soil. This has been recognised in many countries, which have reduced their fertiliser use and stabilised their phosphorus consumption. However, consumption is expected to increase steadily in developing countries (UNEP, 2011). With the global population set to expand to nine billion in 2050, it is expected there will be a 50% increase in phosphorus use (GPRI, 2010). Two-thirds of this increased demand is likely to come from Asia (FAO, 2007).

1.2 The sanitation revolution

The increase in population has been accompanied by an increase in urbanisation and a growth in mega-cities in some parts of the world. For example, it has been reported that the level of urbanisation in China has increased from 19.4% in 1980 to 44.9% in 2007 (Qiao *et al.*, 2011).

Cities are growing not only because of the increasing population but also because of migration from rural to urban areas. This increased urbanisation has reduced the land available for agriculture due to urban sprawl and brought about the sanitation revolution, i.e. the development of sewerage and wastewater systems to deal with effluent (Ashley *et al.*, 2011), which has implications for the natural phosphorus cycle.

Sanitation systems have replaced the ancient system of spreading human manure and urine on the fields, otherwise known as 'night soil' (Ashley *et al.*, 2011). In medieval times this was relatively easy as cities were close to fields but in the expanding cities of the industrial revolution the distances and amounts became too large for transportation. Alongside this, public health concerns called for safe disposal at a distance from the city rather than the old-fashioned method of spreading manure and urine relatively close to the town. The sanitation revolution shifted the disposal of human waste from a land-based system to a water-based system; meaning phosphorus was discarded into rivers, lakes and oceans rather than recycled back to the land.

As a result, cities are becoming phosphorus 'hotspots', with urine as the largest single source of phosphorus emerging from cities (Cordell *et al.*, 2009). Wastewater treatment plants attempt to remove the phosphorus and the Urban Wastewater Treatment Directive (91/271) requires the removal of phosphorus from sewage if it poses or is likely to pose environmental problems. However, even in developed European countries the percentage of wastewater that undergoes advanced treatment to remove phosphorus has been reported as varying considerably from less than 4% in Turkey to more than 97% in the Netherlands (OECD, 2004).

Another contributor to the phosphorus challenge is the use of detergents. Although the presence of phosphorus in detergent has decreased due to regulation in response to water pollution, it is currently only regulated for domestic laundry. It will be regulated for dishwashing detergents in the near future but industrial detergents are still not covered by regulation.

1.3 The Green Revolution

The importance of phosphorus for plant growth was identified in 19th century but it was not until after World War II that the use of mineral phosphate rock in fertiliser grew exponentially. At the time it was considered a cheap and plentiful supply of phosphorus to help support the rapid population growth and prevent food shortages. Improvements in technology allowed fertilisers to be manufactured in high volumes and the mining of phosphate rock increased. From 1950 to 2000, mineral fertiliser use had grown sixfold (IFA, 2006).

The move from relatively small, mixed farms to commercial, specialised agriculture also influenced the consumption of phosphorus. The recycling of animal waste back to the land on which the animals lived was an important part of the phosphorus cycle. However, this is less likely to occur in intensive and indoor farming systems. It is estimated that only 50% of phosphorus consumed by the 65 billion animals worldwide is returned as waste to the fields where their fodder is grown (Gilbert, 2009).

1.4 The breakdown of the phosphorus cycle

Our dependence on mined phosphate and our creation of a water-based disposal system for human waste means we no longer recycle phosphorus to the same degree. Phosphate rock is mined in only a few locations (see Section 3) and then processed into fertilisers and transported around the world. Once the crops are harvested they are also transported worldwide, and the phosphorus they contain is no longer recycled locally via the decomposition of plant matter. When eaten by humans the phosphorus is excreted (in the EU approximately 0.5 to 1 kg per person per year, depending on country and sex, Flynn & Hirvonen, 2009). However, it is no longer returned to the soil via the application of human excreta onto fields but transported via sewer systems and, depending on the level of wastewater treatment, eventually ends up in rivers, lakes and oceans (see Figure 3).

Since the soil is no longer locally replenished with phosphorus via these organic routes, mineral fertiliser needs to be applied to the soil. We have broken the cycle and phosphorus now moves linearly from mines to oceans at increasing rates. Due to globalisation, geographic inequalities in soil fertility and the restriction of phosphorus reserves to a few locations, we move phosphorus large distances around the world. It is estimated that human activities have amplified the rates of phosphorus movement around the world by about 400% relative to pre-industrial times (Falkowski *et al.*, 2000).

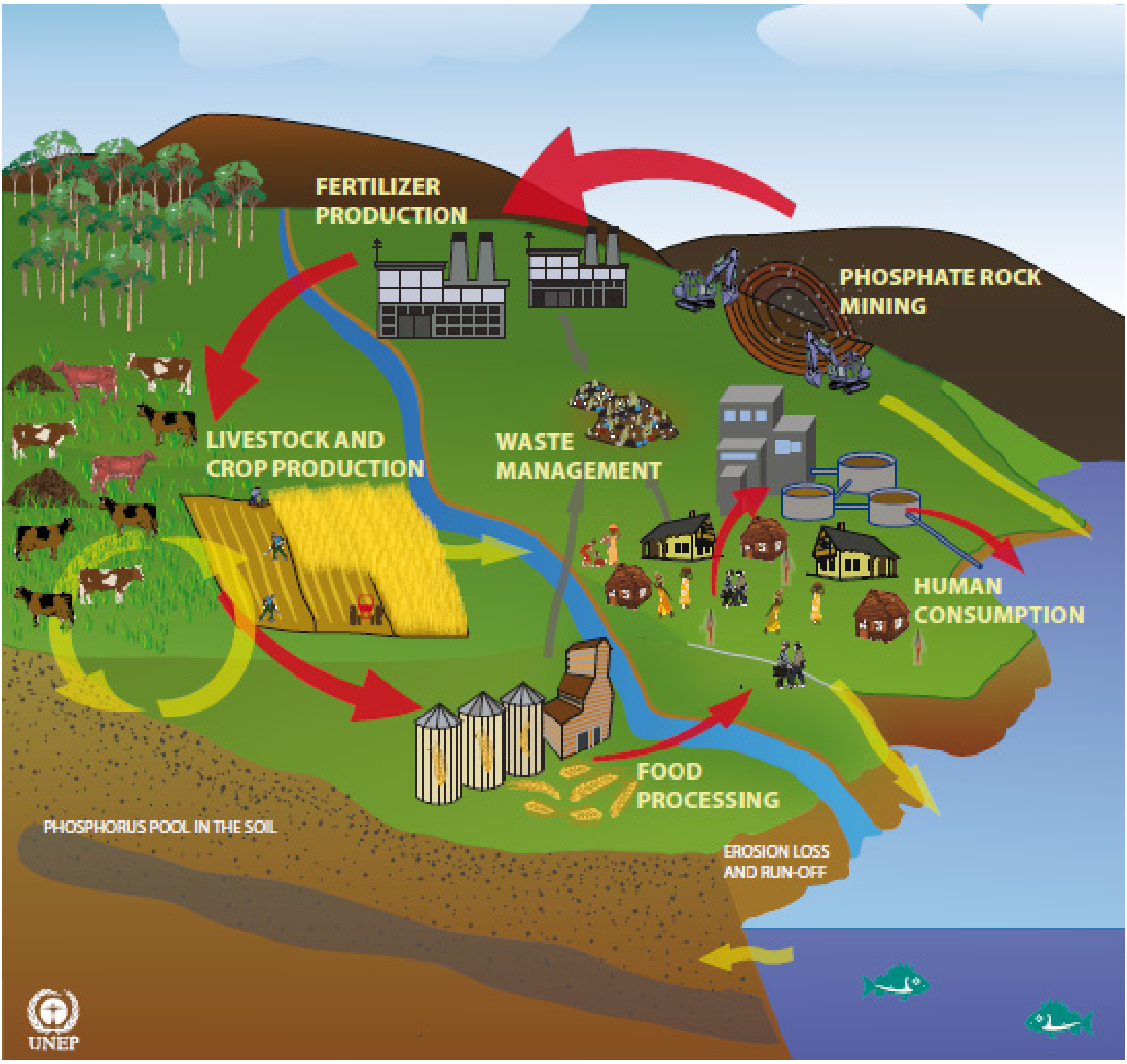

Figure 3: Phosphorus flows in the environment. To enhance crop production, phosphorus is added to soil in the form of mineral fertiliser, manure or biodegradable organic material. Most of the phosphorus not taken up by plants remains in the soil and can be used in the future. Phosphorus can be transferred to surface water when it is mined or processed, when excess fertiliser is applied to soil, when soil is eroded, or when effluent is discharged from sewage treatment works. Red arrows show the primary direction of the phosphorus flows; yellow arrows the recycling of phosphorus. (UNEP, 2011)

1.5 Additional drivers

Alongside the major drivers mentioned in this section, there are other trends and influences that affect the phosphorus cycle. There is a dietary shift, whereby more people are eating meat and dairy products and thus moving away from vegetarian diets. For example, in China it has been reported that the consumption of animal-based foods has increased from 61 g per capita per day in 1982 to 160 g per capita per day in 2002 (Qiao *et al.*, 2011). Meat and dairy products require more phosphorus to produce and it is estimated that the excreta of meat-eaters contains twice as much phosphorus than that of vegetarians (Cordell *et al.*, 2009). This has implications for the amounts of phosphorus in the wastewater system and potentially reaching water bodies (see Box 1). If manure is fully recycled back to agriculture and not wasted, then these dietary trends will have less impact.

High-meat diets also lead to an increase in the numbers of livestock farms that keep animals at high densities and often indoors. This interferes with the natural phosphorus cycle because high livestock densities in fields deplete phosphorus from the land and fertilisers are needed to keep it productive. If the livestock are kept inside they must be fed which requires phosphorus to produce the animal feed. In intensive animal production systems this feed is mostly imported from abroad, for example, the international trade of soya beans.

Another potential influence on phosphorus is the growing interest in biofuels and bioenergy (Ashley *et al.*, 2012; UNEP, 2011). The increasing price of bioenergy crops such as cereal will provide incentives for increasing use of mineral fertiliser to produce them, which will mean more demand for phosphorus (Van Vuuren *et al.* 2010).

Heffer & Prud'homme (2008) identified several potential bioenergy crop developments that would significantly influence the use of mineral fertiliser. Amongst these were increases in oil palm in Malaysia, increases in soya beans and cereals in Argentina, and increases in soya beans, sugar cane, and maize in Brazil. The use of biofuels as a means to meet renewable energy targets has already been criticised due to accompanying land use change and deforestation. Hein & Leemans (2012) analysed various biofuels and suggest that nearly all contribute more to phosphorus depletion than to climate change mitigation. Second or third generation biofuels that use waste products from crops or algae may have less environmental impact in terms of land use, but could have implications for phosphorus use (GPRI, 2010).

1.6 Conclusions for drivers of phosphorus consumption

Population growth, increased urbanisation and intensified agriculture have driven the increasing need and use of mineral fertiliser containing phosphorus. The so-called sanitation revolution and Green Revolution have broken the cycle, whereby phosphorus was returned to the land. Now it is ending up in oceans, rivers, lakes and streams. By understanding past and future drivers we can identify pressures on the environment and the required shifts needed to address the negative impacts.

Box 1. Phosphorus needs for vegetarians vs. meat-eaters

Using WHO (2006) statistics on the amount of phosphorus in the excreta of vegetarians (about 0.3 kg of phosphorus per year) and meat-eaters (about 0.6 kg of phosphorus per year), Cordell *et al.* (2009) estimated how much phosphate rock is required for each type of diet. The figures came to 0.6 kg of phosphate extracted per year for a vegetarian diet (4.2 kg of phosphate rock mined), but more than twice this, 1.6 kg of phosphate extracted per year, for a meat-eating diet (11.8 kg of phosphate rock mined). However, there have been criticisms about the methods and assumptions used to reach these estimations.

2. Pressures on the environment

The DRIVERS discussed in the previous section influence our use of phosphorus, which can exert PRESSURES on the environment. These can be divided into three main types: (i) excessive use of environmental resources, (ii) changes in land use, and (iii) emissions of chemicals, waste, radiation, noise to the air, water and soil (Kristensen, 2004). The pressures on the environment due to our phosphorus use will be discussed in this section.

2.1 Excessive use of resources

About 90% of mined phosphorus is used to produce food and animal feed (Prud'homme, 2010). It is also used in detergent (about 7%) but since the 1960s this has fallen due to bans on detergents containing phosphorus in developed countries (Liu *et al.*, 2008). The average consumption of phosphorus in detergent for the EU-27 is now only 0.2 kg per capita per year. However, there are differences amongst Member States, for example, in 2005 Portugal consumed about 0.6 kg phosphorus per capita as detergent, whilst the figure for Italy was about 0.03 kg phosphorus per capita (Wind, 2007). Other uses of phosphorus include metal surface treatment, flame-retardants and ceramic production but these represent a only a few percent of the total (Liu *et al.*, 2008).

Due to our dependence on mineral fertilisers for food production we are diminishing the phosphorus reserves around the world. Between 1950 and 2000 there was a 600% increase in the global use of fertilisers containing phosphorus, nitrogen and potassium (IFA, 2006). The intensity of global fertiliser application averages about 10 kg phosphorus per hectare (ha) (Liu *et al.*, 2008) but this varies significantly between regions and countries, even within the EU (see Figure 4). China is the top consumer of phosphate (around 30%) (Heffer, 2009) and Asia consumes by far the greatest amount of phosphorus compared to other continents (see Figure 5). The consumption of Europe (including Turkey, Russia etc.) is relatively small although still greater than Africa's.

Among western European countries, application levels have decreased and are now stabilising. However, the consumption of mineral fertiliser in developing countries is increasing: in 1960 fertiliser consumption of developing countries accounted for 12% of the global total and this increased to 60% by 2001 (Wekhard & Seyhan, 2008), which is illustrated in Figure 6. According to the International Fertiliser Development Centre (IFDC, 1998), agricultural intensification in Asia, Africa and Latin America will be the major reason for increase in demand for mineral fertilisers from 2000 to 2030.

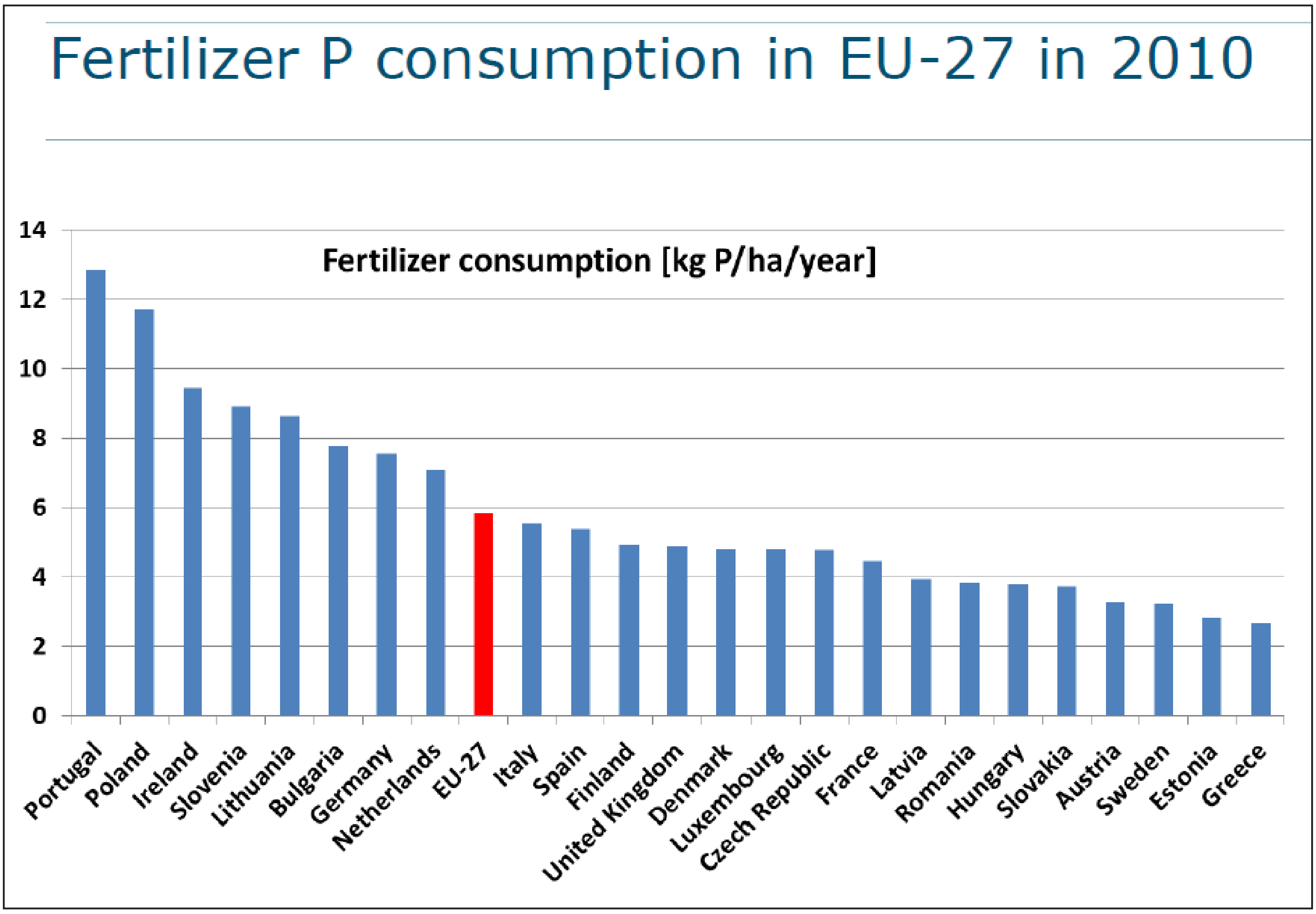

Figure 4: Mineral fertiliser phosphorus consumption in EU-27 for the Member States in 2010 (Van Dijk *et al.*, 2013)

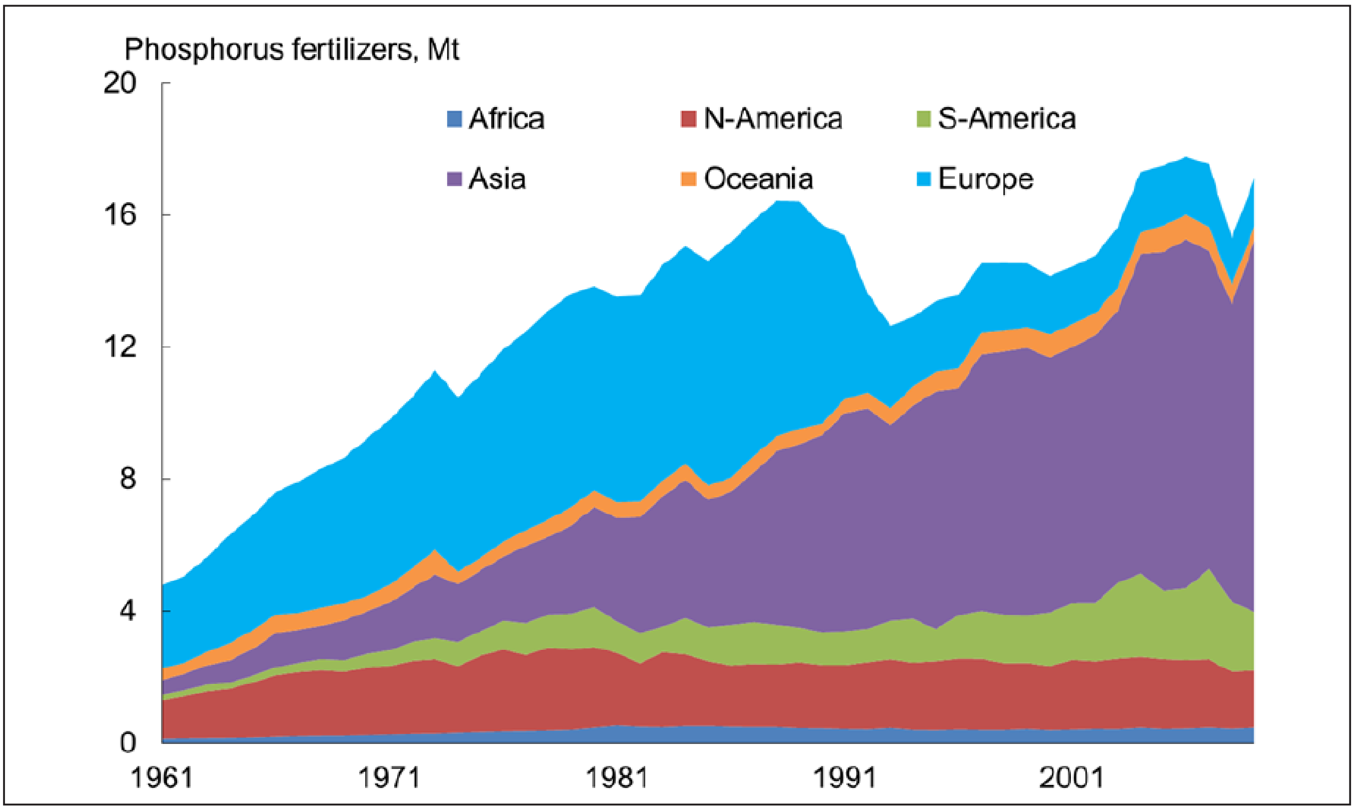

Figure 5: Global fertiliser phosphorus consumption 1961 – 2010. (Van Dijk *et al.*, 2013)

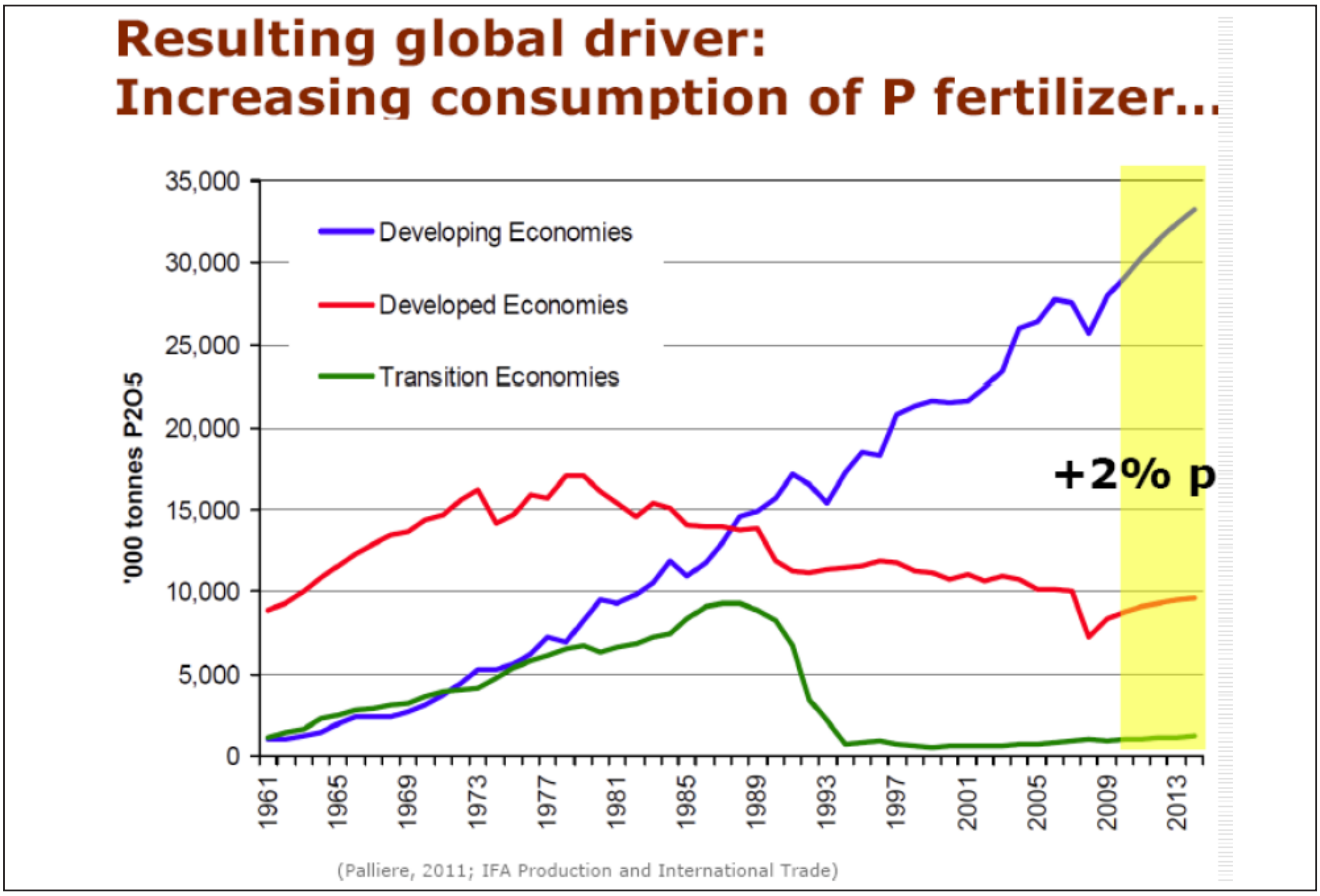

Figure 6: Increasing consumption of phosphorus fertiliser. (Arno Rosemarin, 2013).

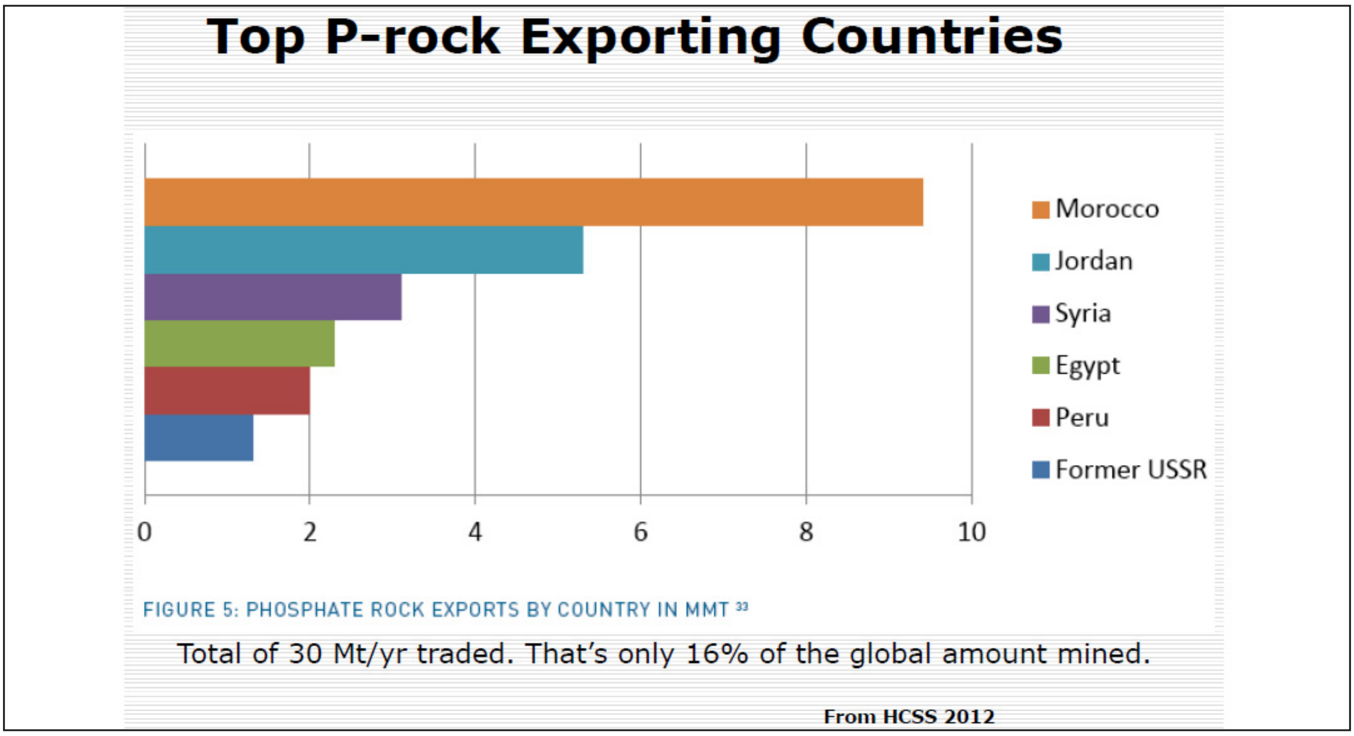

Figure 7: Top phosphorus-rock Exporting Countries (The Hague Centre for Strategic Studies (HCSS), 2012; PotashCorp, 2011).

2.1.1 Geographic distribution of resources

It is not only where phosphorus is consumed that is an issue, but also where the reserves are distributed around the world. Just five countries control 85 to 90% of world's remaining reserves. These are Morocco, China, Algeria, Syria and Jordan (see Table 1). China, the US and South Africa tend to retain their phosphorus for their own use whilst Morocco and Jordan are the largest exporters (see Table 1).

The EU is almost totally dependent on imported phosphate for food and agricultural production (only Finland has reserves) and in recent years changes in exportation taxes have decreased the affordability of phosphorus. For example, after the 2008 global phosphorus price spike, China imposed 135% export tariff on phosphates (*Fertiliser Week*, 2008). This indicates that the uneven distribution of phosphorus reserves around the world could encourage the development of monopolies and volatile pricing (Elser & Bennett, 2011).

In both Morocco and Tunisia the main phosphate rock mines are state-owned, which means they could potentially engage in price-setting behaviour, especially since Morocco has such a large market share of phosphate rock. In addition, large parts of the supply chain are now coming under the control of single firms, for example, a single company may control the mining, processing and fertiliser production. This is known as vertical integration and increases the potential for monopolisation (De Ridder *et al.*, 2012).

Volatile pricing is further influenced by geopolitical tensions. For example, the US imports significant quantities of phosphate rock from Morocco, which currently occupies the Western Sahara and controls its phosphate rock reserves. UN law has condemned this (Cordell *et al.*, 2009) since the question of sovereignty of the Western Sahara remains unresolved and the UN considers it a non self-governed territory. It is the reserves from Morocco that have recently boosted the estimates of global phosphorus by the IFDC (see Section 2.1.4). On-going tensions in Syria are also concerning, and the impact of the Arab Spring in Tunisia is a micro-example of what could happen on a more global scale. Here phosphorus exports fell from €1billion to €0.6 billion as a result of political turmoil (De Ridder *et al,* 2012)

COUNTRY	PHOSPHATE ROCK RESERVES (MILLIONS OF TONNES)
Morocco & Western Sahara	50,000
China	3,700
Algeria	2,200
Syria	1,800
Jordan	1,500
South Africa	1,500
United States	1,4000
Russia	1,300
Peru	820
Saudi Arabia	750
Other Countries	7,800
World total (rounded)	**67,700**

Table 1: World phosphate rock reserves (Jasinki S., 2013)

2.1.2 Ore grade and quality

Where possible, the best quality phosphate rock is mined first as this requires the least work in terms of cleaning and processing to produce mineral fertiliser or phosphate for use in other applications. However, this means the more superior reserves are diminishing quickly and we will be left with reserves with a low concentration of phosphate, or more contaminants such as cadmium. According to the 2012 Hague Centre of Strategic Studies report (De Ridder *et al.*, 2012), the availability of good quality phosphate rock is steadily declining, requiring ever-increasing levels of processing. The higher traces of cadmium in phosphate rock are of particular concern since cadmium is a toxic element that remains throughout the production process and is present in the mineral fertiliser. The concentration of cadmium is highest in the phosphate rock from Morocco, predicted to be the main global supplier in future (De Ridder *et al.*, 2012).

Due to the diminishing quality of reserves, the future production of phosphate fertiliser is likely to require more energy, which is increasing in price. As such the issue of sustainable phosphorus is closely linked to the energy market (Vaccari & Strigul, 2011).

2.1.3 Impacts on price

In a perfect market economy, a shortage of a commodity should trigger a feedback control system of increased efforts to find raw materials for supply. Due to an expectation of high returns on this effort there will be encouragement to find new solutions. This feedback mechanism has been observed with resources such as cobalt during the price peaks of 1978 and 1980. Phosphorus experienced a similar peak in 2007–2008 when the price of phosphate rock increased dramatically worldwide. The average US price in 2008 was more than double that of 2007, and was four-times greater than that of 2004 (see Figure 8). The impact was even greater in North Africa and other importing regions where average prices in 2008 increased more than five-times from the average price the previous year. However, after this peak the price decreased and so did the motivation to explore alternative methods to support sustainable phosphorus (see Section 5). As Figure 8 shows, since 2010 the price has once more started to increase and experts confirm that prices are likely to either remain at the current level or rise, but they will certainly not fall (De Ridder *et al.*, 2012).

Due to the price spikes and variations in price there has been increasing government intervention, which may also affect availability of phosphorus. For example, as mentioned above, China has implemented high export tariffs on phosphate fertiliser, to protect its domestic reserves and keep phosphate fertiliser within its borders. This is in response to concern for food security in the context of growing demand for meat, dairy and cereals in China (De Ridder *et al.*, 2012). As a result, exports have decreased by over 60,000 tonnes in five years (from 102,000 in 2005 to 40,000 in 2010). In the future there may be an increasing amount of access restrictions, trade barriers, export quotas and other protectionist policies.

2.1.4 Phosphorus reserves and resources

There is a great deal of debate in the scientific community and industry on estimations of phosphorus reserves and resources. Reserves are the supplies of phosphate rock that are available with today's technology and are currently economically producible. Resources are the total supplies of phosphate rock, including those in oceans, that may be available in future. Since a reserve is a dynamic concept dependent on current technology and economic situation there are issues around how to define and measure it. As technology improves or prices increase more of the resource is judged to be part of the reserve. There are large uncertainties in estimates such as 'inferred' information from aerial surveys on the area and depth of reserves, which have not been verified or undergone ore-grade analysis. Often ore grades and other

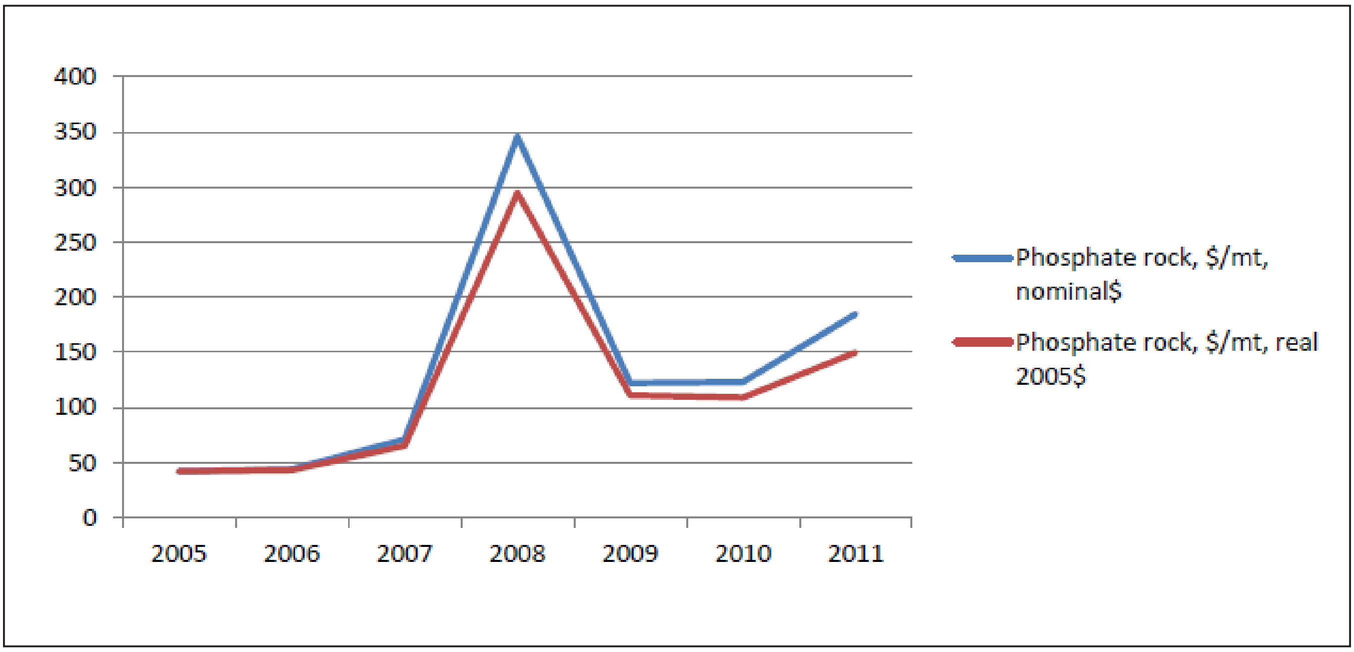

Figure 8: Price of phosphate rock in $ per tonne (The Hague Centre for Strategic Studies (HCSS), 2012).

characteristics of reserves are extrapolated from the analysis of core samples from drill holes, which may not be representative of the quality of the whole reserve area. In addition, figures tend to be self-reported by countries or companies which often use different assumptions for reserves such as the value of the deposit per tonne (Schröder *et al.*, 2010).

According to the latest estimates (2009) from the International Fertiliser Development Centre (IFDC), there are 60,000 million tonnes of phosphate rock reserves, whereas the previous US Geological Survey estimate was only 16,000 million tonnes. Estimations of Moroccan/Western Saharan reserves increased from 5.7 million tonnes to 51 million tonnes when the IFDC report was published in 2009. If this is the case Morocco alone now controls 77–85% of remaining reserves (Elser, 2012). According to Elser (2012) the revision by the IFDC lacks independent confirmation but has now been accepted by the US Geological Survey and its most up-to-date figures at time of publication are quoted in Table 1. The scale of the revision and the debate around the reliability of the IFDC report (see Box 2) indicate the controversy and uncertainty around reserve estimates.

2.1.5 Lifetime of reserves

The simple approach of estimating the lifetime of reserves is to divide the reserve by current consumption rates (about 160 to 170 megatonnes per year), which is the so called reserve/consumption ratio. This yields an estimated lifetime of reserves of 300 to 400 years for global reserves (372 according to Scholz *et al.*, 2013). However, this assumes that 100% of the reserve is accessible and consumption will not increase in the future. Scholz *et al.* (2013) propose that the use of a static indicator in such a dynamic system (where phosphorus reserves change due to technological developments, exploration efforts, population dynamics etc.) is inappropriate. They recommend that it be used only as early warning indicator to trigger more research and policy action regarding increased mining activities, technological innovation and other avenues of development.

Box 2. Responses to the IFDC report on phosphorus reserves

After the IFDC report was released, the Global Phosphorus Research Initiative (GPRI) based in Australia issued a statement, highlighting a number of issues with the IFDC estimates. Firstly, it claimed the IFDC report estimates of reserves were based on secondary sources since much of the real data is unknown or kept confidential by mining companies in China and Morocco. Secondly, the IFDC estimates of Moroccan reserves are based on a 1998 paper and unverified assumptions regarding ore concentration. Thirdly, the large increase in the IFDC estimate was mainly a result of redefining some resources (mostly in Morocco) as reserves. Lastly, by definition, reserve estimations depend on economic feasibility and the price of phosphate rock, but in the IFDC report there is no explicit statement regarding the assumed price on which the estimates are based (GPRI, 2010).

Van Vuuren *et al.* (2010) explored concerns about phosphorus depletion using a range of estimates of phosphorus reserves and various scenarios for future consumption (based on the four Millennium Assessment Scenarios and combined with different scenarios for agricultural, household and sewage systems). Their analysis indicted there were no signs of short-term depletion but in the longer term low-cost, high-grade resources would be in short supply and production would be restricted to only a few countries.

2.1.6 Peak phosphorus

The concept of a ‘peak resource’ has been proposed as a method to represent the phosphorus challenge. It was first proposed by M.King Hubbert in 1949 in the context of oil production (Ashley *et al.*, 2011) and has become a familiar term in policy and the public arena. It describes the point in time when the production quantity of a resource reaches a peak and then declines, constrained by the energy requirements and economics of extracting lower quality and less accessible reserves. This means that the critical point or peak is reached before 100% of the resource is depleted.

Being a non-renewable source, phosphate rock will peak at some point; however, there is much debate across the industry and in the scientific community regarding when the peak will occur. Peak phosphorus modelling takes into account the declining quality, accessibility and increasing costs of mining the remaining reserves and assumes increasing demand (Neset & Cordell, 2011). However, there are numerous caveats to the calculation and uncertainty in estimating both the demand and the supply of phosphate rock (Vaccari & Strigul, 2011), which are discussed below.

Demand-related uncertainties occur around population, patterns in dietary consumption, agricultural efficiency, production practices and waste or losses from the food production chain (Neset & Cordell, 2012). There is also ambiguity regarding the supply: as mentioned above, there are large uncertainties surrounding estimates of the amount and quality of phosphate rock reserves but also around technical advances that may occur to improve efficiency of phosphorus use and recycling. Lastly the fact that the term ‘reserve’ is dynamic brings with it uncertainty. As such there have been various different estimations for when peak phosphorus will occur (see Box 3).

2.1.7 Development of alternative indicators of availability

The concept of a ‘peak’ has been challenged by several sources. Cordell *et al.* (2011) suggest that in reality the peak is more likely to be a lumpy plateau with several small peaks, as production and price fluctuate. Similarly Scholz *et al.* (2013) propose that the basic underlying assumption of a ‘peak’ in the form of a bell-shaped symmetrical curve of demand is unlikely to reflect the situation. Vaccari & Strigul (2011) suggest that although the peak concept has its drawbacks, it should not be abandoned completely. It can be useful as an early warning system and communication device but its application should be accompanied with confidence intervals to communicate the level of uncertainty.

Overall, there is a need for better and more dynamic indicators of phosphorus availability to inform policy and industry. Although the concept of a peak is more dynamic than the reserve-to-consumption ratio, there are some critical factors that it does not include. As mentioned, the inequality of distribution of phosphorus reserves is also an important factor as is the geopolitical risk related to dependency of supply on a certain country, in this case, Morocco (see Box 4 for possible indicators).

The story of peak phosphorus illustrates that when using such models, scientists must also be able to communicate their uncertainty and constraints. Scholz *et al.* (2013) propose that although it is important to recognise the gravity of the situation and encourage developments in exploration, recycling and technological innovation, care must be taken not to induce too extreme a public perception of phosphorus scarcity. They suggest this could cause a soar in fertiliser prices to a point that limits access to phosphorus by some users and results in food insecurity, particularly amongst smallholder farmers in developing countries.

The research conducted by Scholz *et al.* (2013) is part of the Global TraPs project (Transdisciplinary Processes for Sustainable Phosphorus Management)[6] and brings together stakeholders from a range of backgrounds to provide insight into this area. They call for a multi-perspective view that involves scientific and industry experts to help achieve 'socially robust solutions' that incorporate scientific knowledge and data from mining companies, geological surveys etc.

2.1.8 Pressure on water resources

Phosphorus mining also puts demands on water resources, particularly in arid and semi-arid regions where saline or brackish water are used for beneficiation (concentration and cleaning) or recycling measures. Here the availability, and therefore price, of water could become a critical issue in the production of fertiliser (Scholtz *et al.*, 2013). Most countries in the Middle East and North Africa region, including important producers of phosphate rock, suffer from water shortages, mainly because irrigated agriculture requires large amounts of water. This could mean competing demands for water from irrigation and mineral fertiliser production, although both are required to improve food security (De Ridder *et al.*, 2012).

Box 4. Indicators of geopolitical risk and market share

Scholz *et al.* (2013) propose some indicators that could represent these two important aspects of phosphorus availability. The Herfindahl-Hirschman-Index (HHI) represents the market share inequality of a country, where a high HHI signals a dependency of supply on a few countries. For phosphorus this index rose from 2150 in 2008 to 5050 in 2010, due to the increase in estimated reserves in Morocco. To assess geopolitical risk, Scholz *et al.* (2013) propose using the World Governance Indicator (WGI) (Kaufmann *et al.*, 2009), which is based on political stability, quality of regulation and absence of violence or terrorism. They suggest coupling HHI and WGI with available amount of phosphorus and demand to give an effective indicator for its availability.

2.2 Changes in land use

Mining afffects the land and its ecosystems. Phosphate rock, the main source of phosphorus, is found in two forms of deposit: sedimentary and igneous. The majority (83%) is sedimentary and the main producers of sedimentary phosphate, which tends to have a higher grade, are China,

Box 3. Estimations of peak phosphorus

In 2007, Déry & Anderson estimated that peak production of phosphorus occurred in 1988 for the US and in 1989 on a global scale. The estimation for the US seems reasonable, but for global reserves the peak did not occur as production is resuming its growth (Vaccari & Strigul, 2011). The analysis of Déry & Anderson (2007) was inaccurate as it did not assume a total reserve value and only used historical production data until 2006.

Since 2007, production of phosphate rock has been estimated to peak between 2030 and 2040 (Cordell et al. 2009) based on the data provided by the US Geological Survey (Jasinski 2006, 2007 and 2008). This estimate quantitatively incorporated the effects of population changes, dietary changes, increased use of biofuels, soil improvement and better agricultural efficiency.

The more recent IFDC estimate of phosphorus reserves of 60,000 megatonnes (IFDC, 2010) was not accompanied by a peak phosphorus analysis. However, the fourfold increase in reserves is likely to shift peak phosphorus to a later date and a revised analysis by Lindstrom, Cordell & White (2013) that takes into account both the Cordell *et al.* (2009) results and the IFDC reserve figures indicate a probable peak between 2051 and 2092 with an average of 2070 (see Fig 8).

Cordell *et al.* (2011) suggest that, at best, the new estimate of phosphorus reserves 'buys time' for us to make more substantial changes to our phosphorus use. Similarly, Vaccari & Strigul (2011) highlight that time is required for changes in technology and lifestyle. Even if more phosphorus can be made available (see Section 5), such changes must be put in place well before the impending depletion of resources.

[6] http://www.globaltraps.ch/Global_TraPs.html

the USA, Morocco and Tunisia. Igneous phosphates are mainly found in Russia, South Africa, Zimbabwe, Brazil and Finland and are often low-grade.

The primary method of mining of sedimentary phosphate rock is surface or strip mining and consists of clearing the vegetation, removing topsoil and overlying rock before removing the ore. Strip mining changes the environmental system, altering the surface and underground water flows, topography, habitats and biodiversity.

In some countries, such as the USA, reclamation of mining areas is required by law, and usually involves restoring landscapes. Despite great efforts, the changes to the landscape can be permanent, affecting how water seeps into underground streams and flows off the land (Pearce, 2011). These can have relatively greater environmental impacts in ecologically sensitive or highly populated areas (UNEP, 2011).

As high quality phosphate rock is depleted, remaining reserves will be lower in phosphorus content, and a greater area of land will need to be mined to produce the same amount of phosphate. In geopolitically sensitive areas, such as Morocco and Western Sahara, mines may become inaccessible due to disputes, placing greater demand on mines in other regions and driving more land use change.

2.3. Pollution and emissions

2.3.1 Energy consumption and greenhouse gas emissions

Energy is required for phosphate rock mining, processing and transport as well as for fertiliser production and transport. This energy is typically sourced from fossil fuels, which has implications for greenhouse gas (GHG) emissions, pollution and diminishing fossil fuel resources. Phosphate production makes up only 3% of the energy use for mineral fertiliser production compared to nitrogen production, which requires 92.5% (Prud'homme, 2010). Nevertheless, this is still a substantial amount of energy and has been estimated at 13 MJ per kg of phosphorus (Schröder *et al.*, 2010).

Energy requirements are likely to increase as high quality phosphate becomes less available. Moreover, since both fossil fuels and phosphorus are non-renewable and diminishing resources, a vicious circle could arise whereby increasing demand on energy increases its price, which in turn increases the price of phosphorus.

2.3.2 Contaminated waste

Phosphate from sedimentary rocks contains a number of contaminants, which if transferred to the environment can negatively affect ecosystems and humans. Of particular concern are uranium and cadmium, which are radioactive and toxic respectively. Attempts are made to remove these during the concentration and cleaning processes to extract the mineral but they may not be successful. Cadmium concentrations of phosphate rock differ from region to region so the level of potential contamination in mineral fertiliser will vary (see Table 2). Igneous rock has a much lower level of cadmium than sedimentary rock (Linderholm *et al.*, 2012) and these reserves are mainly in South Africa and Russia.

The phosphate rock with the highest levels of cadmium is found in Togo, Senegal and Tunisia (Table 2).

As Figure 9 shows, if we take into account the amount produced by the country then Morocco has the most alarming amount of cadmium that is accessed during phosphate mining.

The processing and production of fertiliser from phosphate rock is predominantly done through the reaction of sulphuric acid with phosphate to yield phosphoric acid. The byproduct of this reaction is phosphogypsum and typically 4 to 5 tonnes of phosphogypsum are produced for every tonne of phosphoric acid (Prud'homme, 2010). Although this byproduct contains phosphorus it is currently unusable due to concerns about the radioactivity of phosphogypsum from the uranium and radium it still contains.

Techniques for recuperating the phosphorus in this byproduct are under development. However, in the meantime mineral fertiliser companies have to dispose of phosphogypsum after processing. It is usually in the form a cemented mass or cake and the most common method of disposal is wet stacking which involves pumping water into the phosphogypsum cake to make it into a slurry that is transported to a disposal area where the solids are allowed to settle out. These 'stacks' can cover hundreds of acres, and there is concern about the radioactive material leaching to groundwater. In a small number of phosphoric acid plants the gypsum is discharged directly into a water body. This occurs in some mines in Morocco where the gypsum cake is slurried with water and discharged into the Atlantic Ocean (Wissa, 2003).

Country	Cadmium content (mg per kg of Phosphorus)	Type of deposit
South Africa	0.04 to 4	Igneous
Russia	0.1 to 2	Igneous
USA	3 to 165	Sedimentary
Jordan	5 to 12	Sedimentary
Morocco	6 to 73	Sedimentary
Israel	7 to 55	Sedimentary
Tunisia	41	Sedimentary
Senegal	71 to 148	Sedimentary
Togo	72 to 79	Sedimentary
Other countries (Algeria, Syria, Finland, Sweden)	0.1 to 28	Sedimentary / Igneous

Table 2: Cadmium content of phosphate rock reserves from different geographical regions (Adapted from Schröder *et al.*, 2010, p48).

2.3.3 Pollution and emissions from application of phosphorus to land

As discussed in Section 1, the use of mineral fertiliser has been growing steadily. In the EU-27 alone, 1.36 million hectares of land are fertilised (European Fertilisers Manufacturers Association, 2008). Not all the phosphorus is from mineral fertiliser, but also comes from sources such as manure and other excreta, and to a lesser extent biodegradable

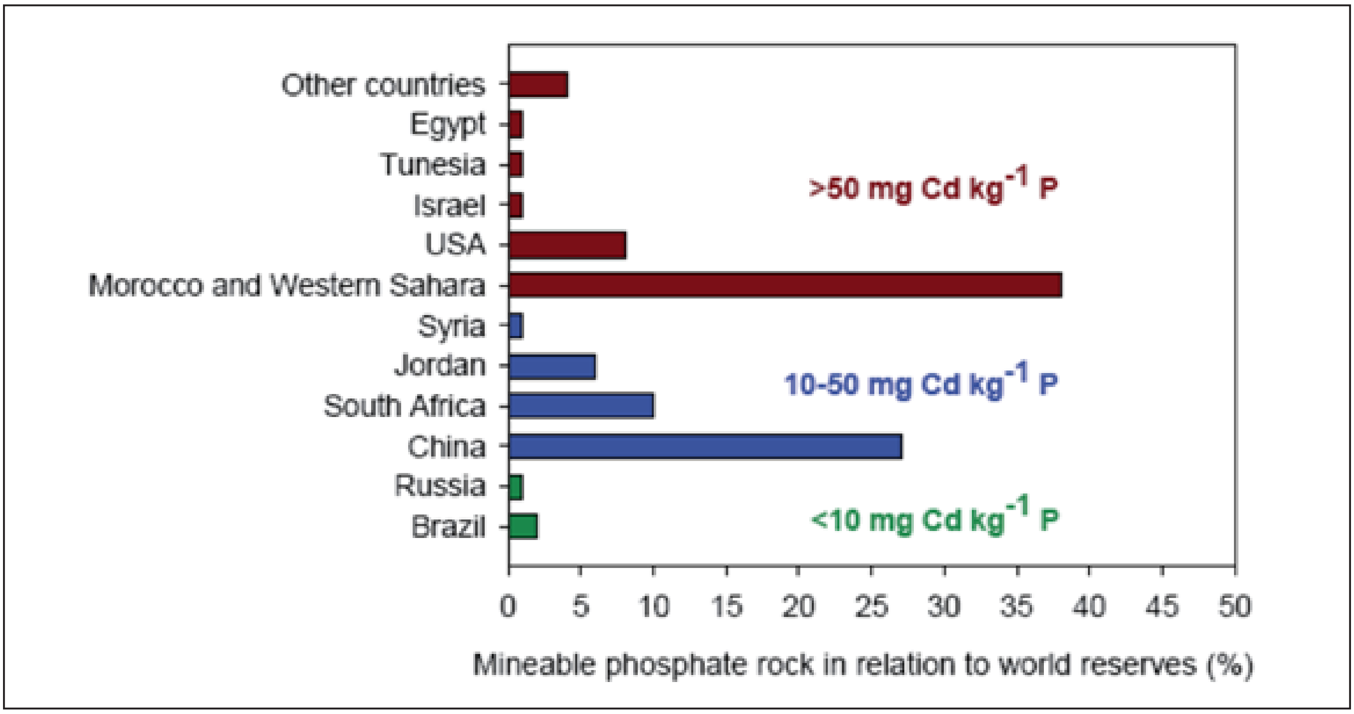

Figure 9: Cadmium levels in of phosphate rock including the relative size of the reserve (Schröder *et al.*, 2010).

organic waste. However, the production of manure by animals is reliant on feedstuffs that are produced using mineral fertiliser phosphorus.

Phosphorus that is not absorbed by plant roots can be lost from agricultural land, either in the form of rainwater runoff, which can leach into the groundwater, or by being blown by the wind when it is attached to soil particles. If such excess phosphorus reaches water bodies it can trigger the growth of algal blooms, which can deplete water-borne oxygen and release toxic compounds. This is known as eutrophication and can damage aquatic wildlife and reduce biodiversity (see Section 4 for more detail).

2.3.4 Pollution and emissions from wastewater and solid waste

Almost 100% of phosphorus consumed in food is excreted in urine (70%) and faeces (30%) which means 3 megatonnes per year of phosphorus is excreted (Cordell *et al.*, 2009), or approximately half a kilogram per person per year (Jonsson *et al.*, 2004). Today, excreta often end up in waterways via wastewater or as sludge in landfills. In global terms, only a small amount of human excreta is actually treated before it is disposed or reused and there is variation among countries in the amount of wastewater subjected to advanced treatment to remove nutrients. For example, it is estimated that in mega-cities in developing countries more than 70% of wastewater enters surface or groundwater untreated (Nyenje *et al.*, 2010). Even in Europe there is variation, with less than 4% of wastewater subject to treatment in Turkey and more than 97% subject to advanced treatment in the Netherlands (OECD, 2004).

2.4 Conclusions on pressures on the Environment

Our current dependence on phosphorus is applying multiple pressures on the environment. These range from land use change to emissions and pollution, to the basic but important depletion of phosphorus itself. Although we cannot be certain about the quantities of phosphorus reserves globally, we do know that they are distributed in few countries and the quality of reserves is decreasing. The debate about peak phosphorus is ongoing and the volatility of the phosphorus market is another pressure with implications for the economy and society.

3. State of phosphorus in the environment

The PRESSURES from phosphate mining, application of mineral fertiliser and waste disposal affect the environment. The resulting STATE is the combination of the environment's physical, chemical and biological conditions, (Kristensen, 2004), which includes the quality of air, water and soil, as well as the health of ecosystems and humans.

Essential to understanding and qualifying how phosphorus-related PRESSURES influence the STATE of the environment are phosphorus flows, which can provide insight into where and why phosphorus is accumulating and diminishing. Flows describe the inorganic phosphorus cycles in soil, rocks, water etc. and the organic phosphorus cycles in plants and animals (Liu *et al.*, 2008). They include mechanical movement of phosphorus as well as the physical, chemical and biological transformations of phosphorus due to processes such as erosion, phosphate rock processing for mineral fertiliser and excretion by animals and humans.

The natural phosphorus cycle is a slow and ancient process (see Figure 10). If left to its own devices, its ultimate ecological fate is as aquatic sediments mainly in the oceans (Childers *et al.*, 2011). Over time, these sediments eventually become rocks with phosphorus rich mineral deposits. However, human activities have significantly intensified the natural phosphorus cycles (Liu *et al.*, 2008) and over four times as much phosphorus flows through the environment now than prior to the introduction of mineral phosphate fertiliser (Smil, 2002). This gives a sense of the magnitude of change that has occurred in the global phosphorus cycle.

Phosphorus flows are most often studied using substance flow analyses (SFAs) or material flow analyses (MFAs) which assess and track the flow of a resource such as phosphorus, water or energy between various processes, typically in a unit mass per year. Although sometimes difficult to quantify due to the numerous and complex inputs and outputs and lack of reliable, available data, the analysis of flows can provide a picture of where the major losses of phosphorus occur and where phosphorus is building up, in so-called 'hotspots' (Cordell et al., 2012).

Phosphorus flow analyses can inform a better understanding of the potential IMPACTS on the environment, society and the economy (see Section 4) and the most effective RESPONSES (see Section 5). In turn phosphorus flow analyses will be valuable to industry to inform the potential market for the use of alternative forms of phosphorus, both spatially and temporally. They are an important first step to creating more efficient phosphorus management system (Cooper & Carliell-Marquet, 2013).

Phosphorus flow analyses can be conducted at local, national, regional and global levels. It is argued that small-scale research can produce more practical insights and appropriate responses to sustainable phosphorus use than large-scale research (Guitar, 2008, Cooper & Carliell-Marquet, 2013). However, research at larger scales is very useful to obtain an overview of the whole situation and differences between regions. Both bottom-up and top-down research approaches to phosphorus flow analyses are needed in order to gain the whole

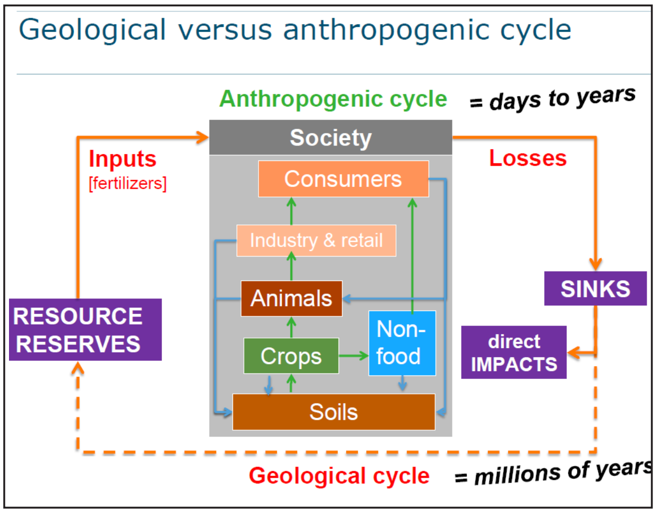

Figure 10: Geological vs. anthropogenic cycle (from Van Dijk *et al.*, 2013).

picture (Van Dijk, in correspondence, 2013). Integrating both levels of research is also necessary to analyse market potentials and future dynamics for more efficient use of phosphorus.

The magnitude and distribution of phosphorus flows varies widely between countries and regions highlighting the importance of analysis at different scales. Cordell *et al.* (2012) conducted a review of 18 recent phosphorus SFA studies, which included global, regional, national, water catchment and city-level analyses.

Some summaries and examples of the phosphorus flow research conducted at these different levels will be outlined in this section. Before considering the range of phosphorus flow analyses it is worth summarising some of the general issues around the flow of phosphorus in soil and water.

3.1 Phosphorus in soil and water

The distribution and availability of phosphorus in soil depends on various biological, chemical and physical processes, which need to be understood to optimally and sustainably manage phosphorus. It is known that a considerable proportion of applied phosphorus reacts with iron, calcium and aluminium to produce insoluble phosphates. As such, this accumulation of phosphorus is often considered to be a loss, as it is not available to crops. Although this is true in the short term, in the long term the accumulated phosphorus could potentially be once again available.

There are two main factors that control the availability of phosphorus to plants: the concentration of the phosphate ions in the soil and the ability of the soil to replenish these ions when the plants remove them (Sattari *et al*, 2012). Soils differ in their ability to replenish phosphate ions but until recently mineral fertiliser has been added to soils with little consideration for this variation and therefore used excessively. As such, there is a large surplus of phosphorus in many developed countries due to the presence of unavailable phosphates in the soil.

There is a lack of knowledge regarding biogeochemical processes affecting phosphorus in soil. Similarly, information on the loss of phosphorus from soil due to erosion and surface runoff is also limited. It is known that erosion varies considerably among countries. Some of the most serious soil erosion takes place in the agricultural systems of southeast Asia, Africa and south America (Yang *et al.*, 2003). In terms of surface runoff Liu *et al.,* (2008) estimated that the global mineral phosphate fertiliser application led to a loss of 0.45 million tonnes of phosphorus per year. Again this varies between countries and in European countries typical phosphorus runoff rates range from 0.2% to 6.7 % but worldwide the maximum can reach 10% for certain soil characteristics and climatic conditions (Waddell & Bower, 1988).

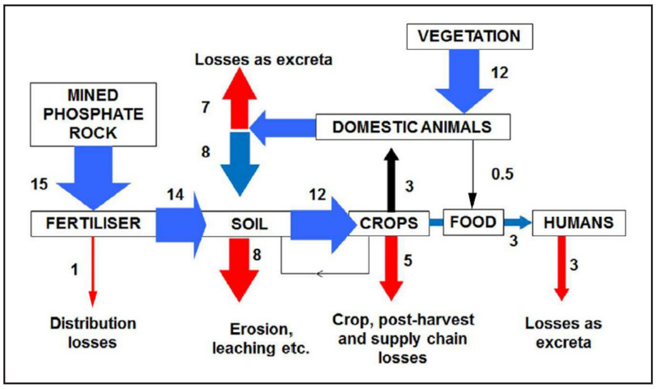

Figure 11: The Industrial ecology of Phosphorus: Simplified Global Mass Balance (Clift & Shaw. 2012)

3.2 Global phosphorus flows

When analysing global phosphorus flows in croplands Liu *et al.,* (2008) make a distinction between natural and societal (or human-driven) cycles. They consider there to be three natural cycles: the inorganic cycle which refers to the phosphorus in the Earth's crust and the weathering of phosphate rock to release it into the soil; the organic cycle that transfers phosphorus from soil to plants to animals and back again; and the water based organic cycle that circulates it among aquatic wildlife. The societal cycles are more complex, consisting of application of mineral fertiliser, crop harvesting, production of animal feed, the wastes produced by livestock and animals, food consumption and human waste.

From previous figures produced by the Food and Agriculture Organisation (FAO, 2005, 2006), Liu *et al.* (2008) estimate that we apply about 14 million tonnes of phosphorus a year in the form of mineral fertilisers, harvest about 12.7 million tonnes per year as crops and recycle about 6.2 million tonnes per year as crop residues, animal manures and human wastes (see Table 3). This gives a total global input of about 23 million tonnes per year, which is 1.8 times the amount of phosphorus removed from the soil by harvesting (12.7 million tonnes per year).

PHOSPHORUS INPUTS	
From mineral fertilisers	14.7 million tonnes
Recycled from crop residues, manure etc	6.2 million tonnes
Weathering and atmospheric deposition	2 million tonnes
TOTAL INPUT	22.9 million tonnes
PHOSPHORUS OUTPUTS	
From harvesting	12.7 million tonnes
TOTAL OUTPUT	12.7 million tonnes
ACCUMULATION	7.5 million tonnes

Table 3: Summary of Liu *et al.*'s (2008) study on global phosphorus flows

In their attempt to hone in on where action is required to improve sustainable use of phosphorus, Clift & Shaw (2012) analysed the total loss of phosphorus from the global system using data from Cordell *et al.* (2009), Liu *et al.*, (2008) and van Vuuren *et al.*, (2010). Unlike the analysis by Liu *et al.,* (2008), Clift & Shaw (2012) consider phosphorus inputs and outputs from livestock farming (see Figure 11). In the figure they provide red arrows to signify losses and blue arrows to signify phosphorus inputs.

They estimate that the total input to food production is around 27 million tonnes per year of which more than half (around 15 million tonnes per year) is mined, which is similar to Liu *et al.* (2008) figure of 14 million tonnes (see Table 3). This 15 million tonnes of phosphorus in mineral fertiliser delivers about 3 million tonnes per year in food. Liu *et al.* (2008) estimated 12.7 million tonnes of phosphorus is removed from harvesting. However, clearly not all the phosphorus in harvested crops ends up in food and as Clift & Shaw (2012) point out there are distribution, post-harvest and supply chain losses of about 6 million tonnes (see Figure 11 and Table 4).

PHOSPHORUS INPUTS	
From mineral fertilisers	15 million tonnes
Vegetation fed to livestock etc.	12 million tonnes
TOTAL INPUT	**27 million tonnes**
PHOSPHORUS OUTPUTS	
Food	3 million tonnes
TOTAL OUTPUT	**3 million tonnes**
PHOSPHORUS LOSSES	
Distribution losses	1 million tonnes
Erosion, leaching losses	8 million tonnes
Post harvest and supply chain losses	5 million tonnes
Losses as human excreta	3 million tonnes
Losses as animal excreta	7 million tonnes
TOTAL LOSSES	**24 million tonnes**

Table 4: Summary of Clift & Shaw (2012)'s study of global phosphorus flows

This means that only a fifth of phosphorus mined for food production ends up in the food eaten by the global population (Cordell *et al.,* 2009). Ninety per cent of the total phosphorus entering the food system (mineral fertiliser and organic manure) is lost without reaching the product (Clift & Shaw, 2012, see Table 4). This is mostly via dissipation into the water system. Accumulation of phosphorus in soil also represents a significant amount, but Clift & Shaw (2012) are reluctant to quantify this in their assessments due to uncertainty surrounding estimates of dissipation by soil erosion and leaching. Indeed there is still much discussion about the estimation of phosphorus accumulation in soil. These two examples of estimates of global phosphorus flows of agriculture and food production indicate the enormous amounts of the mineral moving through the environment and the large amount that is lost and accumulated without reaching firstly the crops or our plate. They also indicate the difficulty in estimating flows at this level and the different figures that can result due to the lack of concrete data, assumptions made in the estimations and definition issues.

3.3 National flow studies

Regional and national flow studies focus mainly on food security, environmental protection and sustainable wastewater management, as these are all important issues to be addressed at this scale. Unlike global flows it considers phosphorus that is imported into and exported out of the country as food and animal feed. Nationally the largest phosphorus losses are due to leakage from agricultural land (for example, up to 66% in the US, Suh & Lee, 2011) or as direct emissions from wastewaters or solid waste.

Research also indicates that losses from the food system and associated industries have a significant role to play. For example, exported products, such as bone meal used for porcelain in the Netherlands or live sheep and cattle exports from Australia, result in losses from the region. Again this depends on the country under consideration.

Within the EU there are at least 11 national phosphorus flow analyses, some of which are mentioned below and synthesised in the research by Van Dijk *et al.* (in preparation) on phosphorus flows in the EU-27 (see Section 3.4). Researchers in China (Ma *et al.*, 2011; Bi *et al.*, 2013) and Japan (Matsubae *et al.*, 2010) are also conducting various analyses, as are researchers in the US (Baker *et al.*, 2011). National and regional phosphorus flow analyses can be conducted from various approaches and work with various assumptions. They tend to focus on the issues or systems that are considered a priority in the country.

The Swedish study reported here by Linderholm *et al.* (2012a) focuses on agriculture (see Box 5a), whilst the study by Smit *et al.* (2010) and de Buck *et al.* (2012) in the Netherlands (see Box 5b) compares the flows in different systems such as agriculture, wastewater, industry etc. The study in the UK (Cooper & Carliell-Marquet, 2013, see Box 5c) focuses on food production and consumption whilst the studies in Finland (Antikainen *et al.*, 2004; Saikku *et al.,* 2008) focus on forestry and energy industries (see Box 5d). Many of the national studies highlight the need for a more detailed analysis after national analysis in order to provide more specific recommendations for different regions and areas (Cooper & Carliell-Marquet, 2013; Antikainen *et al.*, 2004; Saikku *et al.*, 2008).

The regional study in France (Senthikumar *et al.*, 2012) highlights the large regional differences in phosphorus flows and sources. It also conducts the analysis over more than ten years so trends can be identified to see how the situation and drivers changed and what can be done in the future.

As one of the largest consumers of phosphorus (see Section 2), China has been conducting an increasing amount of phosphorus flow research, in a bid to reduce phosphorus contamination. The study by Ma *et al.*, (2011) focuses on the phosphorus flows in agriculture but breaks this down into three different crops (see Box 6) whilst research by Bi *et al.*, (2013) has focused on one county in China where water pollution is a major problem.

Box 5a. National phosphorus flow analysis in Sweden

Linderholm *et al.* (2012a) investigated phosphorus flows to and from Swedish agriculture and the food chain for the period 2008-2010. Inputs consisted of all supplies used in agriculture such as mineral fertilisers and animal feed, atmospheric deposition, food imports from other countries, and phosphorus in wastes recycled in agriculture such as manure, etc. Outputs consisted of phosphorus exported in agricultural products, phosphorus in waste and wastewater (that is not recycled), losses due to erosion and leakage to water.

Overall, for the period 2008-2010, the study calculated that Sweden had total phosphorus **inputs of about 27,573 tonnes per year** and **outputs of 14,894 tonnes per year**, producing a **net input of 12,679 tonnes per year** which works out at **4.1 kg of phosphorus per hectare of land.** Previous estimates for Sweden in 2008 were much lower at around 1 kg of phosphorus per hectare. The reasons for this discrepancy are not clear but do indicate the level of uncertainty in these estimations, most likely due to uncertainties in the values of imports and exports from food and animal feed.

The largest contribution to the flows was from **mineral fertilisers at 10,800 tonnes per year** (3.5 kg per hectare). **Food imports and imported fodder contributed 5,760 and 5,600 tonnes** respectively, whilst the export of food removed 5,560 tonnes from the system, Further analysis showed that the surplus phosphorus was concentrated on farms with high livestock density (more than 0.6 animal units per hectare). This is despite strict animal density regulations related to the phosphorus content of manure and indicates more action could be taken in this area.

Box 5b. Phosphorus flows in the Netherlands

Smit *et al*. (2010) analysed national phosphorus flows in the Netherlands for the year 2005 and de Buck *et al*. (2012) updated this information for the year 2008.

Taking the most recent data, the analysis shows that the Netherlands is a net importer of phosphorus, **importing 115,000 tonnes and exporting 64,000 tonnes for 2008** resulting in a **net import of 51,000 tonnes.** About half of the national import comes from animal feed, mainly for intensive livestock production. This is a much larger figure, almost ten times as much, than the animal feed imported into Sweden (5,600 tonnes per year).

From 2005 to 2008, the net import decreased by **9,000 tonnes**. This was mainly due to **a reduction in use of mineral fertilisers (by 9,000 tonnes),** and an increased manure export (6,000 tonnes) alongside a net import increase of the industry (by 6,000 tonnes). The latter was due to an increase in feed phosphorus import. The decrease in the use of mineral phosphorus fertilisers was probably due to the change in legislation in 2006 that introduced maximum allowed application standards for nitrogen and phosphorus instead of maximum allowed surpluses

Over a third of the accumulation (19,000 tonnes) occurs in agriculture. This is a decrease of **12,000 tonnes** from 2005 (31,000 tonnes). In 2008, returns of phosphorus from waste (household and industry) are negligible: the waste sector produces about 32,000 tonnes of phosphorus annually but only 1,000 tonnes is recycled back into agriculture. Approximately 2,000 tonnes is exported and more than 29,000 tonnes of phosphorus from waste sector ends up in incinerator ash or landfill.

Box 5c. Phosphorus flow analysis of UK food production and consumption

A recent study (Cooper & Carliell-Marquet, 2013) of the phosphorus flows in UK food production and consumption in 2009 calculated the **total imports** of phosphorus to be **138,000 tonnes** and **exports** to be **23,500 tonnes (see figure 14).** As such the UK has a **net import of 114,500 tonnes** of phosphorus (see Figure 12). Over half of the total imports is due to **imported fertilisers (77,500 tonnes)** and **animal feed accounts for 6% (8,300 tonnes).** The largest losses within the systems were those to water, estimated at around **41,000 tonnes per year** and over half of this (23,500 tonnes per year) was discharged within wastewater. The **accumulation of phosphorus** was estimated at **37,500 tonnes per year.**

The researchers suggest more focus should be put on removing and recovering phosphorus at wastewater treatment works (see Section 5). Alongside this they suggest improving the techniques to recycle bulky wastes such as animal manure, food waste and sewage sludge. This would help to reduce soil accumulations and replace imported fertilisers.

There is also growing research in Japan on phosphorus flows. Generally speaking, Japan is in a similar situation to the EU, as it has essentially none of its own reserves of phosphate rock. Its main supplier is China followed by South Africa, Morocco and Jordan. As such, the phosphorus in its imports are of considerable interest and Matsubae *et al.*'s (2010) study takes a novel approach to examine the 'hidden' flows that exist in food and animal feed that is imported into the country (see Box 7).

3.4 Phosphorus flows in EU-27

Using data from Miterra-Europe, CAPRI, FAOSTAT, Eurostat, current reports and articles Van Dijk *et al.* (in preparation) have estimated the phosphorus flows for the EU-27. There are some notable data deficiencies, particularly in Central and Eastern Europe, but overall the study demonstrates that Europe is largely dependent on imports. Preliminary results indicate that total phosphorus imports are about 2.7 million tonnes and exports are 0.6 million tonnes, giving a net import of 2.1 million tonnes of phosphorus. Seventy per cent of this net import (or 1.49 million tonnes of phosphorus per year) is in the form of mineral fertilisers for crops, 20% (or 0.42 million tonnes of phosphorus per year) in animal feed and additives, and 10% in food and non-food materials. The preliminary estimate for the losses from runoff, manure and waste is 1.25 million tonnes (see Figure 14). The average mineral fertiliser consumption for EU-27 is about 5.8 kg per hectare per year with Portugal, Poland and Ireland as the biggest consumers (see Figure 4 in Section 2). There is on-going accumulation in agricultural soils, especially in Western Europe (the Netherlands and Belgium). Recycling rates are generally low: on average only 15% of compost in the EU-27 is re-used (although this varies from 68% in the Netherlands to 0% in some parts of Central and Eastern Europe). On average 42% of phosphorus in sewage sludge is recycled. This ranges from 90% in Luxembourg to 0% in Malta, where it is landfilled. There is also no recycling of sewage sludge in the Netherlands, where application of sludge on land is restricted, so it is incinerated. The remaining ashes are not recycled to agriculture, but used for road building, cement production or landfilling in mines (see Section 5).

3.5 City level flow studies

Kalmykova *et al.* (2012) suggest there is a lack of analysis of phosphorus flows at the city-level because research has tended to focus on agricultural flows. However, cities are often hotspots of phosphorus accumulation. There is a great deal of variation between city-level flow

Box 5d. Phosphorus flows for specific industries in Finland

Antikainen (2004) estimated the phosphorus flows in the Finnish forest industry and consumption of wood fuel from 1995 to 1999. On average the input of phosphorus was **3180 tonnes per year: 2600 tonnes of this was domestic** and **580 tonnes was imported.** Half of the domestic input was from wood products industry and the other half from pulp and paper industry, whereas 88% of the imports is from pulp industry, and 12 % from wood products. Many large forest industry plants are located by the sea and as a consequence a large proportion of nutrient emissions end up in the sea (**40% of the phosphorus emissions**, see Figure 13).

The researchers recommend that returning nutrient-rich wastes, especially those containing ash, back to forests, would improve the situation. Another recommendation is to reduce phosphorus inputs, which in this case could mean debarking the trees in the forests as bark has a disproportionate amount of phosphorus.

Saikku *et al.* (2008) studied the phosphorus flows in the Finnish energy system for 2000. About **6,000 tonnes** flowed into the energy system, mainly in the form of **peat (47%)** and **wood (42%).** The main output was in the form of ash from these fuels: about 60% was recovered and utilised (for example, in earth construction) but only 3% was recycled as fertilisers despite the fact that it is technically a good fertiliser. The remaining 40% of the ashes was deposited in industrial landfills. Although the flows are not as great as the energy sector there is still room to reduce the phosphorus flows by reducing the use of nutrient rich fuels such as peat and coal and increasing the amount of phosphorus returned to the forests as fuel ash.

Box 5e. Phosphorus flows at a regional level in France

Senthikumar *et al.* (2012) analysed phosphorus flows for 21 French regions on a yearly basis from 1990 to 2006 and selected four of these regions to study in more detail. In **1990** the calculated phosphorus level **averaged 17.5 kg per hectare per year and** there were no deficits. Over the years this level declined to **4.4 kg per hectare per year in 2006** with phosphorus deficits in some regions. There were large differences between the 21 different regions. For example, the highest level of phosphorus in the soil was observed in Brittany where it declined from 33 kg per hectare in 1990 to 17 kg per hectare in 2006, while in Centre region it declined from 13kg to 0 kg per hectare during the same period.

The regional levels of phosphorus were highly dependent on the regional farming system. For example, in Brittany, the high number of standing animals led to a much higher flow of animal manure than the quantity of phosphorus removed from soils through crop uptake.

The researchers suggest that more balanced phosphorus flows could be obtained by modifying the agricultural production systems. However, this would require a systemic approach that considers all interacting drivers that lead to these imbalances (see Section 5).

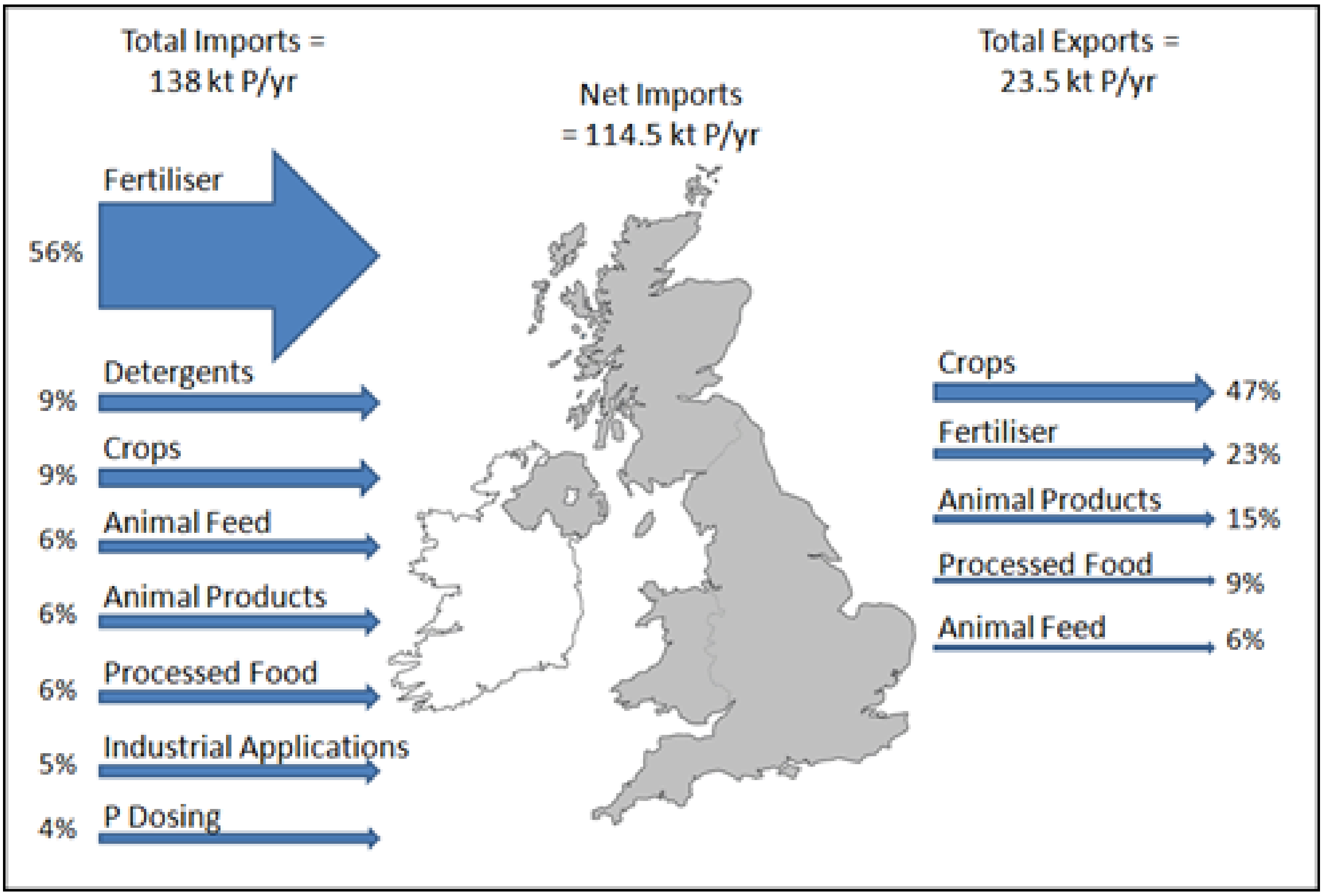

Figure 12: UK phosphorus exports and imports (Cooper & Carliell-Marquet, 2013)

studies with the phosphorus flows (see Box 8 and Box 9) depending on the specific characteristics of the urban system, including size, industry, infrastructure and whether they are in a developing or developed country (Cordell *et al.*, 2012). Typically the largest phosphorus flow is in the wastewater system where phosphorus losses occur when untreated wastewater is discharged to rivers and seas and when sewage sludge is not reused. The final destination of sewage sludge varies between countries and can be in landfilling, agricultural land, ocean or non-agricultural land. In general less phosphorus is recycled in cities than in rural areas. Liu *et al.* (2008) estimated that 20% of urban human wastes and 70% or rural human waste is returned to food production. This varies with country, but with cities growing in size and rural communities shrinking, cities are likely to become even greater hotspots for phosphorus.

3.6 Issues around phosphorus flow research and data

Data availability and data quality are an issue for phosphorus flow research. As can be seen from the above studies, data are sourced from a range of organisations, databases and publications and analysed using different approaches. Often it is difficult to acquire data from industry or from some countries. In addition, countries can define the destinations of phosphorus differently such as what is considered landfill or incineration. As such it is difficult to compare results across

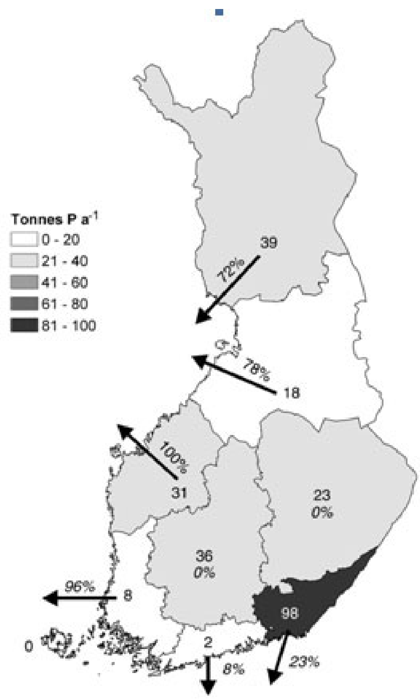

Figure 13: Regional distribution of the phosphorus emissions to water from the forest industries in Finland, 1995–1999. Total emissions and the proportion ending up to the sea. (Antikainen *et al.*, 2004)

Box 6. Examples of phosphorus flow analysis in China

Ma *et al.* (2011) investigated the phosphorus flows for three of the main grain crops in China: **wheat, rice and maize.** Total amounts of phosphorus inputs into the three crops nationwide were estimated to be **1.09 million tonnes, 1.24 million tonnes and 1.12 million tonnes respectively** and approximately four-fifths of this was from mineral fertilisers. However, use of phosphorus inputs is not efficient and there was a high level of accumulation in the fields of 29.4, 13.6 and 21.3 kg per hectare for wheat, rice and maize. The study demonstrates how flow and efficiency of phosphorus use differs between crops and indicates there may respectively be potential to optimise phosphorus use through crop choice.

Bi *et al.* (2013) propose that the growing phosphorus problems in China are due to a lack of effective waste management infrastructure and weak governmental regulation. In 2010, only two-thirds of urban sewage was treated (MEP, 2010) and this rate was much lower in rural areas. As such Bi *et al.* (2013) analysed anthropogenic phosphorus flows for Wuwei County in China, which borders a major lake that is a key target for water pollution control.

The **total phosphorus input into the county was 10,504 tonnes**, mostly in the form of mineral fertilisers and livestock feed. This was more than three times the amount that was exported as farm products or meat (2,857 tonnes). The study estimated **3,578 tonnes were discharged to aquatic environment**, which supports the proposal that a major problem lies in lack of effective waste management. The rest remained on the land and the county had high phosphorus loads both in terms of population (3.35 kg per capita per year) and by area (19.43 kg per hectare per year).

Box 7. Japan and virtual phosphorus ore requirement (VPOR)

Matsubae *et al.* (2011) propose a new concept in the form of virtual phosphorus ore requirement (VPOR) as an indicator that considers both the direct and indirect phosphorus requirements of a society. The direct requirements are those used in agriculture (mineral fertiliser) and industry within the country whereas indirect requirements are those used to produce food, animal feed etc. in other countries that are then exported to Japan.

This 'hidden' phosphorus can amount to large figures, for example the 2,703 tonnes of imported livestock products required approximately the same amount of hidden phosphorus (147,000 tonnes) as that needed to grow cereals (124,000 tonnes) within Japan. This is because livestock production requires about ten times as much phosphorus as the same amount of cereal. Imported fruit also had a large amount of hidden phosphorus at 101,720 tonnes.

In total, the study estimated that the **amount of indirect phosphorus flow associated with imported commodities was 3.74 million tonnes of phosphorus ore** and altogether the direct and indirect requirements came to 6.16 million tonnes of ore, indicating that more than half of the phosphorus requirements are indirect and the rest is used directly to grow crops.[1] Indirect phosphorus flow can therefore play a major part in the cycle, and there is a need for more research in this area. However, such information may create issues in terms of assigning responsibility for responses and setting target

1. These amounts are in terms of phosphorus ore.

studies and it has been suggested that a common framework for data collection and analysis needs to be developed (Pellerin *et al.*, 2013b).

In their study on global flows of phosphorus Liu *et al.* (2008) highlight that there are a number of issues in estimating inputs and outputs. According to the Organisation for Economic Co-operation and Development (OECD), inputs of phosphorus to soil are from mineral fertiliser, manure, atmospheric deposition and seeds. The uncertainty around application of mineral fertiliser, atmospheric deposition and seeds is relatively low but manure input is associated with large error because it is estimated from livestock numbers and national default values for the phosphorus excretion per animal category (Schröder *et al.*, 2011). In addition, the processing of this data is time consuming, for example the last overview of phosphorus soil surpluses provided by OECD was in 2004.

Suh & Yee (2011) in their study of phosphorus efficiency in agricultural use assessed the data quality to understand the limitations in interpreting their results. They found that there was limited data on waste flows and information on phosphorus additives to feed was not readily available due to its sensitivity. They conclude there are a number of important gaps and discrepancies in data provided by national and international authorities such as the US Geological Survey, the FAO and the US Department of Agriculture. They call for a consistent and systematic data infrastructure to collect and analyse data. It is important to communicate these uncertainties alongside the

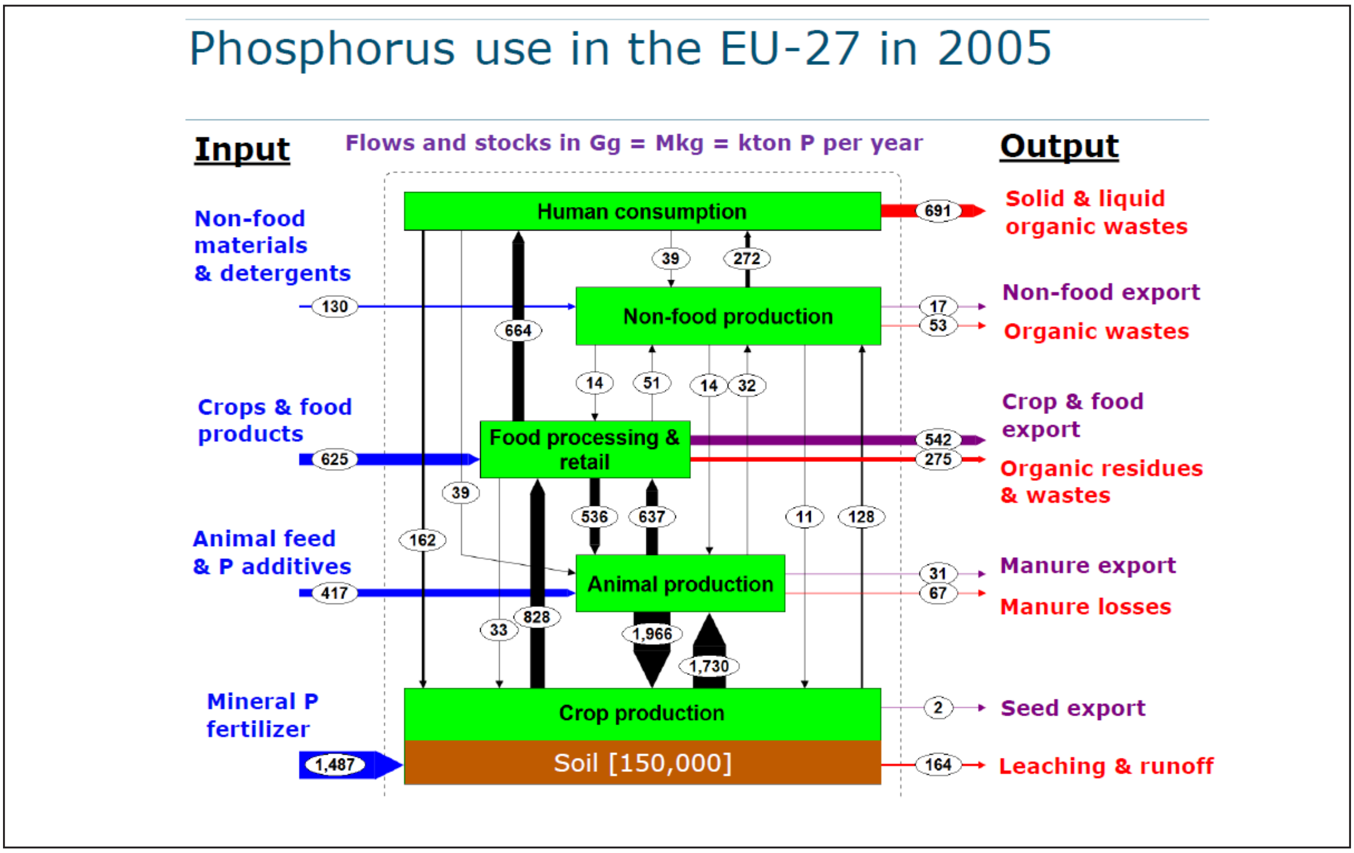

Figure 14: Phosphorus use in the EU-27 in 2005: preliminary results (Van Dijk *et al.*, 2013).

estimates, but also to understand this is the nature of estimating the dynamics of such a complex system.

Additionally, Cordell *et al.*, (2012) propose there is a need for cross-scale and cross-sectoral analyses to identify hotspots to improve management. For example, mining and mineral fertiliser production is considered globally but not often at national scale. There is also a need for better data on phosphorus flows in the food production and consumption sectors in order to account for losses and inefficiencies from this chain. Agricultural data are reasonably good but currently there is insufficient data on the amount of phosphorus in the food system, waste system and from different uses of phosphorus in the industries ,such as detergents and flame retardants. In particular, availability of data on the amount of phosphorus in food waste and additives to animal feed are an issue, as well as the 'embodied' phosphorus involved in food trade. Lastly, there is a need for phosphorus flow analyses to be supplemented by qualitative analyses of stakeholder views, farmer livelihood and the influence of geopolitics.

3.7 Phosphorus losses

With current diets, humans require approximately 1.2 g of phosphorus per day (a minimum of 0.6 to 0.7 g a day), amounting to 3 million tonnes per year for global population. Around five times as much as this amount is mined annually to produce mineral fertilisers (Cordell *et al.*, 2012). This excessive amount is due to the substantial losses and inefficiencies that occur from mine, to field, to fork in the global food and sanitation systems (Cordell *et al.*, 2012). Figure 15 outlines the main losses by the red arrows marked with 'L'. The largest phosphate loss is via agricultural runoff and from livestock (manure and other waste products). There are also substantial losses from the food chain and human excreta, and losses from crops and harvesting. Some brief explorations of where the different losses can occur are exlpained in Sections 3.7.1-3.7.4.

3.7.1 Losses from mining, beneficiation, mineral fertiliser production and distribution

After the phosphate rock is mined it goes through a beneficiation process where it is cleaned and concentrated by removing contaminants, which are either stored or discharged into the river. Some of these, such as iron phosphate, still contain phosphorus, which is therefore lost from the system. There can also be losses via spillages during storage and transport. In the most common process of mineral fertiliser production (the acid route), the reaction of sulphuric acid with phosphate rock to yield phosphoric acid produces phosphogypsum as a by-product (see Sections 3 and 4).

Although there is a substantial amount of phosphorus in these phosphogypsum stockpiles, it currently remains unused due to radioactivity concerns. Current developments in safe processing techniques may allow for the recovery of phosphorus from this source. Finally, at this stage there may again be losses due to spillages, spoilage and during handling, storage and transport of phosphogypsum. Trends towards locating processing plants nearer to the mines should decrease losses due to transport and storage. In addition, the average grade of

Box 8. Phosphorus flows in Gothenburg, Sweden

Kalmykova *et al.* (2012) conducted a study of phosphorus flows in Gothenberg, Sweden. This considered a wide range of inputs and outputs: ingestion of food, food waste, pet food, newspaper and packaging, stormwater runoff, wastewater and solid waste. Phosphorus flows in Gothenburg are very different to those at national or global scales. In 2010 **importation of food commodities accounted for 50% of the input (235 tonnes per year)** and agricultural flows were marginal compared to those related to the urban waste management infrastructure. Both **wastewater and incineration ash** each account for about 40% of the phosphorus output **(374 and 332 tonnes per year respectively).** Currently, Swedish phosphorus recycling targets are based on the amount of nutrients present in sewage sludge, however, this result suggests it may be beneficial to also consider recycling from solid waste as well, such as food waste etc.

Box 9. Phosphorus flows in Chinese megacities

Cities are becoming hotspots for phosphorus and this is particularly the case for large and expanding mega-cities in developing countries. Qiao *et al.* (2011) analysed the phosphorus flows through food consumption in Beijing for the year 2008.

A lot of food is imported, for example, **more than 90% of vegetables in Beijing are supplied by other provinces.** Although sewage treatment techniques are improving, there is still some untreated sewage where the phosphorus is piped directly into natural aquatic systems, especially in the peri-urban areas.

In Beijing **5,374 tonnes of phosphorus were consumed in food by urban residents in 2008.** The largest outflow was to sewage treatment plants of about **3,861 tonnes.** Of this about a tenth (394 tonnes) was discharged into natural aquatic systems and 544 tonnes were recycled as reclaimed water. The remaining **2,923 tonnes were landfilled within the city** in the form of sewage sludge. The research calls for greater recycling of phosphorus either by transporting the sewage to rural areas for agricultural use (although this has implications for cost and energy use) or by recycling it from the sanitation system (see Section 5).

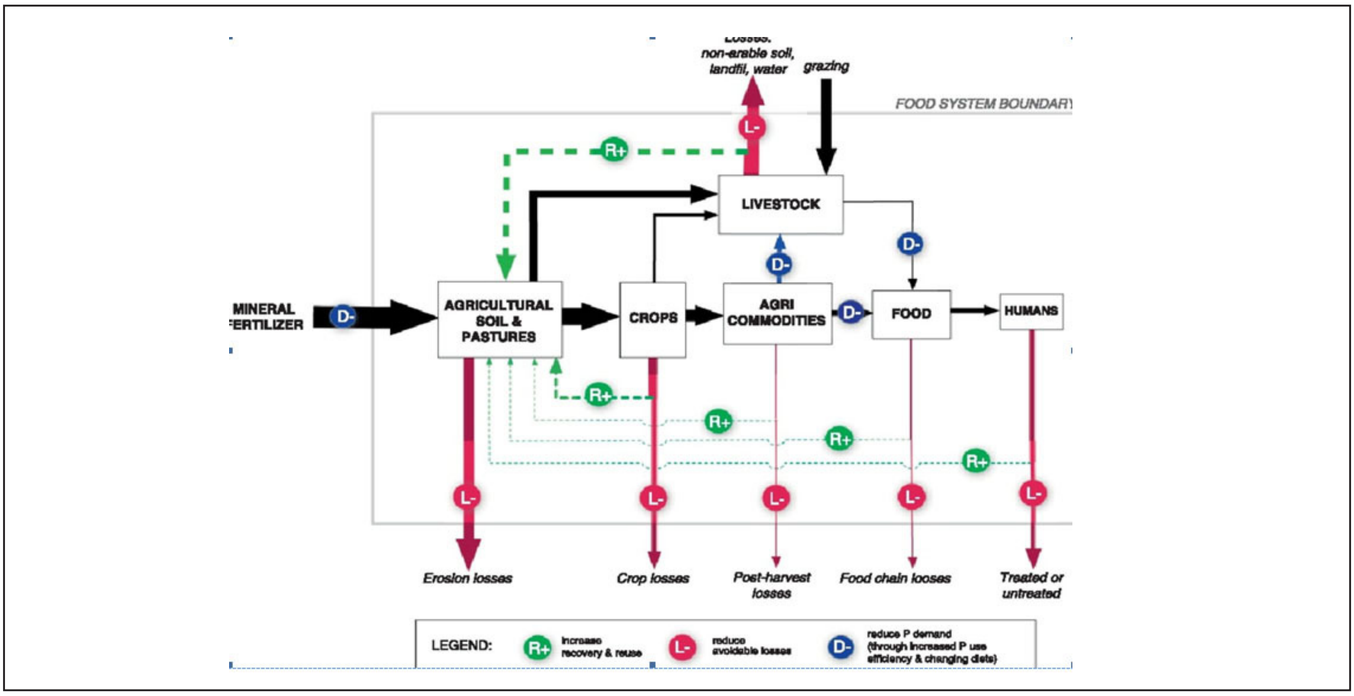

Figure 15: The phosphorus mass balance: identifying 'hotspots' in the food system as a roadmap to phosphorus security. (Cordell *et al.*, 2012)

phosphate rock is dropping year by year and so it is anticipated that there will have to be industry developments to minimise losses in the mining and processing stages (Prud'Homme, 2010b).

3.7.2 Losses from agriculture

Phosphorus has a low capacity to bind in the soil and it is thought that it can be easily washed or blown away. As such, at a global level, erosion could potentially contribute the most to phosphorus loss from soil. Global losses from the soil to fresh water are estimated at 18.7 to 31.4 million tonnes per year (Liu *et al.*, 2008) and in the EU-27 losses in leaching and run-off are estimated at 0.16 million tonnes per year (Van Dijk *et al.*, in preparation, see Figure 14).

Losses from the crop itself are relatively small because once phosphorus is taken up by plants it becomes part of the crop mass. Crops are then harvested or exported for consumption or, if eaten by livestock, deposited as urine and faeces, which can be available for crop growth.

3.7.3 Losses from food production, processing and retailing

Until recently 'post-harvest' losses have been ignored due to the focus on reducing losses from agriculture. However, it is estimated that about half of the 7 million tonnes of phosphorus harvested per year is lost in food production and consumption (Schröder *et al.*, 2010). There are a number of stages in the food commodity chain at which phosphorus can be lost. Losses can occur from harvested crops during storage due to pests and spillage.

The processing of food can also incur losses ranging from the removal of grain husks to below-standard products that are discarded. Due to the globalisation of food chains there is also potential for losses during transport etc. and at the retail stage when supermarkets, markets and other food outlets discard food.

This is particularly the case for food that has passed its sell-by date but is completely edible (Gustavsson *et al.*, 2011). Lastly, food losses can occur within homes prior to consumption. In some parts of the developed world such as the UK, 60% of food waste is estimated to be edible and could be avoided with better food and meal planning (WRAP, 2008).

3.7.4 Excretion and solid waste management

Virtually all of the 3 million tonnes of phosphorus that is consumed in food per year is excreted in urine and faeces. However, globally only 10% is reused as wastewater, sludge or ash from incinerated sludge or used directly from composting toilets or direct defecation. The remaining 2.7 million tonnes per year is discharged into water or non-agricultural land as effluent or landfilled.

In addition, before either reuse or disposal, only small fraction of human excreta is actually treated to reclaim phosphorus. At an EU level it is estimated that 0.69 million tonnes of phosphorus is lost from human consumption/households as liquid (e.g. detergents, excreta) and solid waste (e.g. food waste, kitchen waste) (van Dijk *et al.*, in preparation, see Figure 14).

3.8 Conclusions of state of phosphorus in the environment

Phosphorus flow analyses provide valuable insight into how phosphorus is moving through the environment and where losses are occurring. They can be performed at a number of scales, ranging from the global to the riverbed, providing useful information on where action needs to be taken. In general, the analyses indicate that, although agriculture is a major source of phosphorus losses, cities are increasingly becoming phosphorus hotspots due to the growing amount of human waste that needs to be processed. More specifically, analyses can provide information on the phosphorus flows in different crops, different waste streams and the amount of 'hidden' or embodied phosphorus in imports, particularly animal feed.

Losses are occurring at all stages of phosphorus flows: mining, mineral fertiliser distribution and application to land, crop harvesting, food production and waste processing. Research on phosphorus flow can inform us about the differences between countries, systems and processes, and where to take action. It can also provide comparative information on the impacts of different responses in the transition towards sustainable use of phosphorus and this will be illustrated in Section 4.

4. Impacts of phosphorus consumption

DRIVERS, PRESSURES and STATE of phosphorus consumption result in a number of IMPACTS. In the context of sustainability, these will be summarised under three sections: environmental, economic and social. Although described separately, these impacts are interactive and should be considered in a wider context from a holistic viewpoint.

4.1 Environmental

Negative impacts on the physical, chemical or biological state of the environment arise from both the production and consumption stages of mineral phosphorus fertiliser. This in turn affects the quality of ecosystems and the welfare of human beings that live alongside ecosystems and rely on the services they provide. The various impacts will be described in the following sections.

4.1.1 Phosphate rock mining

As mentioned in Section 2, mining of phosphate rock causes air pollution emissions, land disturbance, water contamination, noise and vibration in the area local to the mines (UNEP, 2011). The severity of these impacts depends on the residential and ecological sensitivity of the local areas.

Strip mines that extract sedimentary phosphate change the natural landscape. Waste materials produced from mining (tailings) are often left in mounds on the ground and can still contain phosphate rock. Runoff from these mounds has a relatively high level of dissolved phosphorus (Kuo & Carpena, 2009), which can cause eutrophication (see Section 4.1.4).

Once mines are closed the original ecosystems remain disturbed. In America mining companies are required to 'reclaim' the land, which involves filling in the mining pits and planting new vegetation. However, it is debatable whether an ecosystem can truly be restored. Brown (2005) suggests affective restoration requires a tight coupling of research, applied ecological engineering practices of industry and government regulation. Mining also requires water and its removal may impact the surrounding environment. Pearce (2011) suggests that when large amounts of water are used during mining this prevents freshwater flowing downstream, which can cause increases in salinity to the lower-level waters. During the cleaning or beneficiation process there is also potential for pollution of nearby rivers by contaminants such as iron phosphate.

4.1.2 Phosphorogypsum

Probably one of the major environmental impacts during processing is the generation of phosphogypsum stockpiles, which is discussed in Section 2. There are some uses for it, such as manufacturing gypsum board and Portland cement, but the majority remains unused due to the toxic materials it contains.

During storage in stockpiles there are concerns about radioactive material leaching to water and being blown to adjacent land that may be used for agriculture. If wet-stacking is used the phosphogypsum is pulped with either fresh or seawater before being transported. Excess and possibly contaminated water is decanted and can discharged back to the sea (Wissa, 2003). Once dry, the surface of the stacks may be disturbed by wind or vehicles and, if transported, spillage may occur from conveyor belts. The phosphogypsum may also seep into groundwater, although regulation in the US has tried to prevent this with the use of liners for stacks.

4.1.3 Cadmium and uranium pollution of soil through fertiliser use

Cadmium and uranium are natural components of phosphate rock. They should be removed during the production process of mineral fertiliser; however, this depends on the rigour of the production company and does not always occur. As such, the use of mineral fertiliser can lead to soil pollution from cadmium and uranium. Furthermore, as mentioned in Section 2, there is a trend towards phosphate containing higher traces of cadmium. Phosphate rock from Morocco and Western Sahara in particular has high levels of cadmium and is a source the EU is likely to rely more upon in the future (see Figure 16).

Cadmium occurs in organic fertilisers, such as manure, as well as in mineral fertilisers. It is easily taken up by plants and can enter the food chain. This is worrying, since cadmium contamination is associated with kidney failure and cancer (De Ridder *et al.*, 2012). Industry statistics indicate that the global production of low-cadmium phosphate (1-20 mg cadmium per kg of phosphate P2O5) is roughly equal to the EU's consumption but that the EU only has access to 10% of low-cadmium phosphate (Chemicals Unit of DG Enterprise, 2003).

The link between mineral fertiliser use and the presence of cadmium in food has been demonstrated by a study carried out by the Finnish Environment Institute (2000). Rock sourced in Finland contains virtually no cadmium, and the study claimed that if Finland applied an 'average' fertiliser that is used in the EU rather than fertiliser made from Finnish phosphorus, then the dietary intake of cadmium would increase by more than 40% over 100 years.

The European Food Safety Authority has set the tolerable level of cadmium at 2.5 μg per kg of body weight but it remains difficult to ascertain what level of increase in cadmium from mineral fertiliser would result in this level in humans, causing serious health damage. OECD countries monitor cadmium levels in soil and crops and the Swedish government advises that the load should not exceed 10 mg of cadmium per kg of phosphorus based on an application rate of 22 kg of phosphorus per hectare per year (Eriksson, 2009).

The level of uranium in soil is also rising due to the use of fertiliser, and tests in Germany have shown increases in top-soil

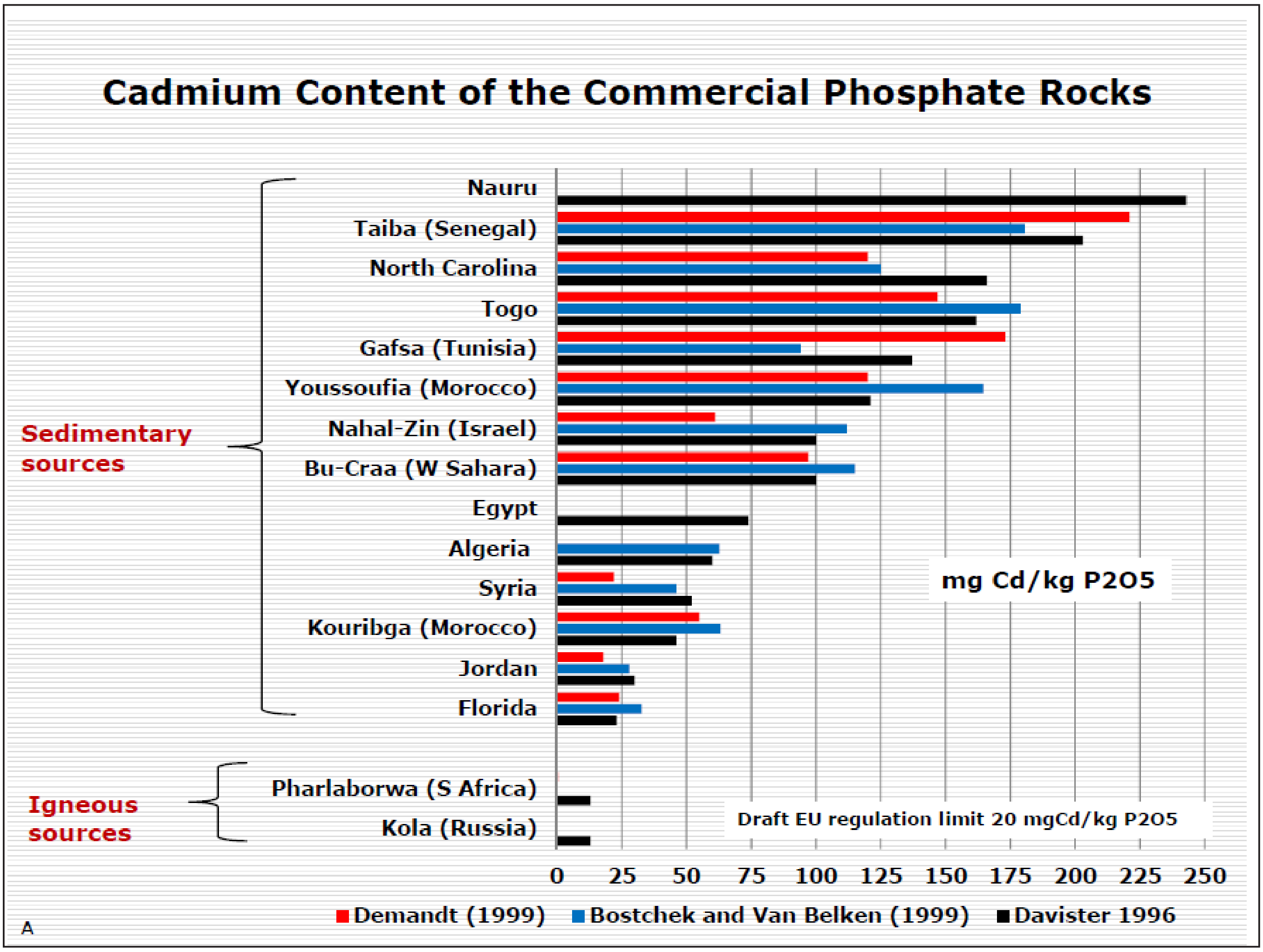

Figure 16: Cadmium Content of Commercial Phosphate Rocks (Arno Rosemarin, 2013)

contamination (Smidt *et al.*, 2012). Uranium and thorium not only risk contaminating food but may also affect the health of agricultural workers. However, research done in a phosphate mine in Egypt (Khater *et al.*, 2001) indicated that the phosphate rocks had the lowest radioactivity levels measurable. In addition, radiation levels in phosphate rock vary a great deal and there is no standard procedure to measure soil radioactivity caused by the application of mineral fertiliser.

4.1.4 Eutrophication

Eutrophication is caused by excess nutrients in water bodies and phosphorus is one of the major culprits. The phosphorus comes from both natural (i.e. manure) and inorganic fertilisers, which have not been absorbed by plant roots or soils, or from phosphorus that exists naturally in the soil. It can come from point sources such as discharges from industry and wastewater, or more diffuse sources such as losses from agriculture, losses from scattered dwellings and deposition from the atmosphere into water bodies. Nutrients, including phosphorus, can be transported in water and, although they may not cause immediate impacts in the coastal areas where they are discharged, there may be impacts further away that are difficult to trace.

Once phosphorus is discharged into surface waters the nutrient overloading of seas, coastal waters, lakes and rivers results in a series of adverse impacts on the aquatic environment. Naturally occurring phosphorus in inland, estuary and coastal waters is often limited, so when excess phosphorus is discharged into these water bodies it enables organisms such as algae to bloom to the point of threatening other life forms present in the water. This is mainly because the algae will eventually die and, as they decompose, they use up oxygen and release toxic compounds.

Reductions in oxygen can kill wildlife or force animals to leave the affected area, triggering changes in ecosystem state and depletion of ecosystem services, for example, reducing fish stocks, impairing recreational use, affecting human consumption of water and reducing nursery habitat for valuable species (Piehler & Smyth, 2011).

The major concern is that overenrichment with nutrients (including phosphorus) can push aquatic ecosystems beyond natural thresholds causing abrupt and possibly irreversible shifts in ecosystem structure

Box 10. Costs of eutrophication

It is difficult to quantify the impact of eutrophication on ecosystem services but it is worth remembering that marine ecosystem services alone are estimated to be worth $20.9 trillion per year, and the majority of this comes from coastal systems ($10.6 trillion per year) (Costanza *et al.*, 1997). Dodds *et al.* (2009) estimated that the annual cost of eutrophication in the US could be as high as US$2.2 billion. The impacts in developing and transition countries are also becoming noticeable and the percentage of Chinese lakes experiencing eutrophication has increased from 41% in the 1970s to 77% in the late 1990s (Bi *et al.*, 2013)

and functioning (Rockström *et al.,* 2009). The relationship between phosphorus levels, eutrophication and degradation of ecosystems is not linear and predictable, making it difficult to know when these thresholds are being approached. This could mean that, in some cases, reparative or remedial action may come too late. Even when phosphorus loading into waters is reduced, recovery will be lengthy as the amount of phosphorus in the water can be maintained by cycling of phosphorus between bed sediments and the water column (Spears *et al.*, 2013).

In large urban centres the problem can be exacerbated when phosphorus from excreta and detergents is concentrated in wastewater streams and discharged into water bodies (see Section 3). If these cities do not have adequate wastewater treatment facilities the problems are even more concerning. Nyenje (2010) estimates 70% of wastewater from mega-cities in developing countries is untreated before it enters surface or groundwater.

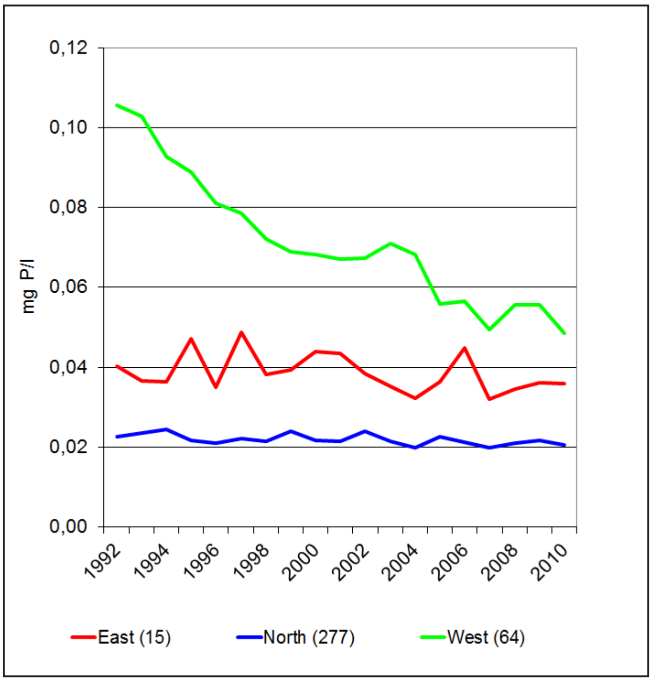

Figure 17: Phosphorus concentrations in lakes (total phosphorus) between 1992 and 2010 in different geographical regions of Europe. (Source: European Environment Agency, 2012)

Within the EU the most persistent eutrophication occurs in the Baltic Sea, the coastal region of Brittany and some Mediterranean coast zones (Schröder *et al*, 2010). The Baltic Sea is of particular concern and almost every summer it develops massive cyanobacterial blooms, which are toxic to mammals.

Figure 17 shows the phosphorus concentrations in lakes in EU regions from 1992 to 2010. Northern Europe shows stable and low phosphorus concentrations in lakes and rivers probably due to the measures taken in the last decades, whilst in Western Europe the decline is more recent since measures were only taken in the 1990s. These measures may be a result of policy response on this area (see Box 11). In Eastern Europe and Southern Europe there are less clear trends, which may be due to the combination of better management and an increase in use of mineral fertiliser.

4.1.5 Energy (transport and production)

Both the mining and processing of phosphate rock use energy and rely on fossil fuels to produce this energy. Currently this has environmental implications in terms of producing greenhouse gases (GHGs) and depleting fossil fuel resources. As and when high-grade, easily accessible reserves diminish, more energy will be required to extract the phosphorus from the rock and remove any contaminants. This will have an impact on price (see Section 4.2), not only because more energy is required, but also because energy prices are likely to increase.

Box 11. Responses to eutrophication

Although in the EU excreta and wastewater should be treated in accordance with public health standards, not all systems also ensure protection of environmental health. The EU Water Framework Directive does require minimum standards for nutrient recovery but not all existing systems such as onsite septic tanks have been retrofitted with the appropriate nutrient removal technology (Schröder *et al.*, 2010). The extent of eutrophication varies but it has been recognised as a global environmental problem (Millennium Ecosystem Assessment, 2005).

However, there is difficulty in determining a threshold phosphorus load to prevent eutrophication from occurring. The causes of excess phosphorus also vary, so any policy action must take this into account. For example, in the Netherlands and Switzerland roughly 50% of the phosphorus in surface water is from agriculture and the other 50% is from the effluent of wastewater treatment. However, in the Baltic region algal blooms are mainly caused by runoff and erosion with Poland as the largest source due to its intensive agricultural activities.

Life cycle analysis (LCA) is a method to estimate energy consumption at all stages of phosphorus cycle including mining, packaging, transportation and farm application. Schröder *et al.* (2010) conducted a basic LCA using estimates from European and US sources. This put the energy consumption from mining, beneficiation and waste disposal at 10 megajoules (MJ) per kg of phosphorus.

Chemical extraction in Europe consumes about 0.3 MJ per kg of phosphorus but the global average is much greater (about ten times as much). Packaging and transport consume nearly 4 MJ per kg whilst farm application consumes less than 1 MJ per kg. In total this brings the estimate to 20 MJ per kg. This LCA is based on general estimations of the stages and, in some cases, the data are more than 10 years old.

4.2 Economic impacts

The current and future patterns in phosphorus use have several implications for global and national economies, which are briefly discussed below.

4.2.1 Price volatility

The uncertainty around the estimations of global and national phosphate rock reserves and the time or existence of a future peak in phosphorus contributes to instability in its price. This is evidenced by the recent soar in prices in 2008 and fluctuations since this time. Although it is impossible to predict how the price will change in the future, what is certain is that the current price is likely to rise and become more volatile unless action is taken to release more reserves.

4.2.2 Ore quality

The volatility in the price of phosphorus is exacerbated by the fact that reserves are becoming less accessible and their quality is decreasing in terms of the lower concentration of the ore and the greater presence of contaminants. This means that reserves require more energy and water for mining and processing and, when linked with the increasing energy prices and limited water supply, this can affect the price of phosphorus. In an ideal world traditional market forces (e.g. increases in phosphorus and food prices) should trigger a feedback mechanism (Scholz *et al.*, 2013) to encourage phosphorus conservation but it appears that either this has yet to begin or does not apply in the case of phosphorus (Childers *et al.*, 2011; Shu *et al.*, 2006).

4.2.3 Competition for reserves

The unequal distribution of phosphorus reserves around the world has economic implications in terms of increasing competition to secure phosphorus supply in a market controlled by a few countries. Currently reserves are concentrated in Morocco, China and the US. Although originally home to one of the largest reserves, the US is now dependent on imports and these are likely to increase, whilst China is also becoming a net importer, with India importing the greatest amount globally.

The distribution of reserves can have a disproportionate impact on countries with no phosphorus reserves and countries that are politically conflicted with phosphorus-producing countries. The EU has very few phosphate reserves and will thus face mounting competition in securing its supply, whilst poor and vulnerable nations that are in most need of mineral fertiliser, such as sub-Saharan Africa, may also suffer. The competition for supply is further exacerbated by export limitations imposed in producing countries such as China, and political instability in countries such as Syria. The only solution for the EU is to work towards more sustainable use of phosphorus, which includes recycling phosphorus to reduce the dependency on other countries (see Section 5).

4.2.4 Monopolies and vertical integration

In addition to concentration of reserves in a few countries the production of phosphorus fertiliser is becoming the realm of fewer companies. Vertical integration occurs when the same company that mines the rock, often in the same location, processes phosphate rock. The increasing trend for vertical integration means there is less processing occurring in the EU. In the past, phosphate rock would be transported to the EU and processed into fertiliser, whereas now it is imported as pre-made fertiliser. This reduces the control or input of the EU into the phosphorus market and also increases the potential for a monopoly by a single company. There are already cases of this happening, particularly in state-owned companies that do not experience high levels of competition. In some cases it may be that processing companies in some countries do not have to abide by the same regulations as EU, and can hence gain a competitive edge (De Ridder *et al.*, 2012).

4.2.5 Benefits to local economy

Phosphate mining and fertiliser processing can support the economy and livelihood of the local population. However, in some cases, this has to be ensured by international intervention and its success can be questionable. For example, in Western Sahara, the area is in fact disputed and is a UN non self-governing territory. A UN ruling states that mines owned by the Moroccan company must support the local population but this can be difficult to enforce (De Ridder *et al.*, 2012).

4.2.6 Future scenarios of global phosphate rock market

In order to provide insight and stimulus for discussion in this area the Hague Centre of Strategic Studies report (De Ridder *et al.*, 2012) has identified four different scenarios of world systems, which would each produce different effects on the EU's ability to influence the global phosphate rock market. This depends on whether the world is dominated by a small number of state actors or a large number of players. It also depends on whether there is an atmosphere of

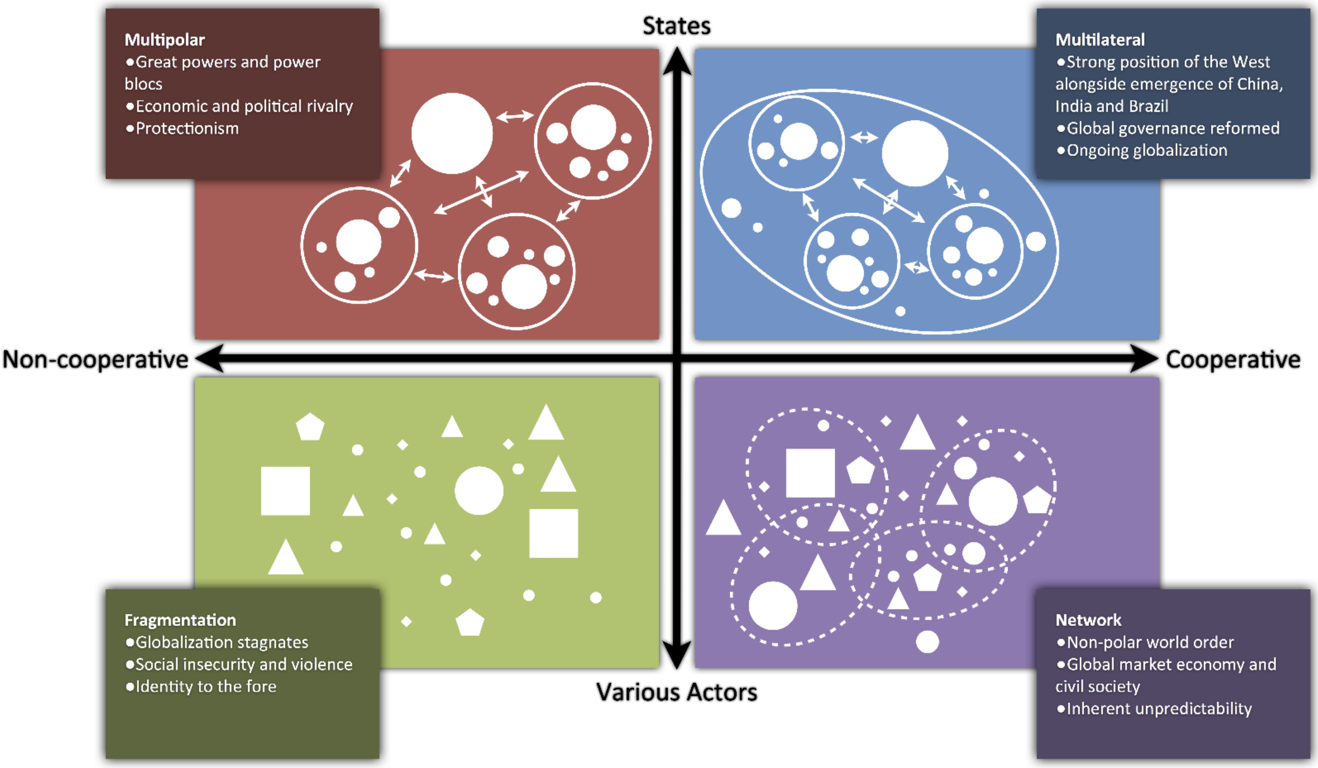

Figure 18: Cooperate or Agitate? A Typology of Scenarios Characterizing the World System. (The Hague Centre for Strategic Studies (HCSS), 2012).

cooperation or competition. The four scenarios are outlined in Figure 18 and there is a brief description from the original report as follows:

1. Under the multilateral scenario, international cooperation is the main means to resolve conflicts of interests. A global agreement could be reached on sharing phosphate rock resources, using less, recycling more and mining sustainably. Free trade will be the dominant rule by which global markets abide which creates a more conducive environment for the EU to secure the imports it needs.
2. In the network scenario, the global market and technological innovation are the major driving forces. Non-state actors (such as NGOs) may play an important role in increasing awareness for the environmental dimensions of phosphate production and consumption, potentially increasing the public support for reduction, efficiency and recycling.

3. In the multipolar scenario, a formation of power blocs has occurred and international conflicts of interests have become more pronounced. With international disputes and conflicts, such as in Western Sahara and Syria, unlikely to be resolved, the options for the EU to diversify its suppliers or forge strategic partnerships with producer countries are limited.

4. In the fragmentation scenario, globalisation is stagnating because the opposing forces have succeeded in getting the upper hand. Escalation of conflict should not be ruled out. Under this scenario, both diversification and the forging of strategic partnerships are difficult, if not impossible.

4.3 Social impacts

The supply of phosphorus has a major role to play in food security, poverty and human health. The Millennium Declaration (adopted by the UN General Assembly in 2000) cites the eradication of hunger and poverty as a primary goal. Many of the world's estimated 925 million undernourished people are small-scale farmers (IAASTD 2009, FAO 2010) and they are often unable to obtain phosphorus-based fertilisers, which limits their productivity (Buresh *et al.*, 1997) and reduces their level of food security. In extreme circumstances this can lead to food riots as seen in 2008 (Bangladesh, India, Haiti and Mexico) and 2010 (Mozambique, Egypt and Pakistan).

The political instability of several of the main phosphorus producers has implications for those living in those countries and those dependent on their reserves. When economic advantages are unequally distributed either between countries, states or companies there is always potential for unrest and, when combined with other political upheavals, the impacts on society can be damaging.

Lastly, there are several potential impacts on human health, which has social consequences. The health impacts from pollution on those living in the vicinity of mines are potentially concerning but uncertain. However, there are even larger widespread impacts in the form of pollution from the use of mineral fertiliser.

Most cited is pollution from cadmium within the phosphate that remains in the mineral fertiliser but there could also be possible contamination from uranium and thorium. Currently little is known or proven in terms of the potential impacts on human health but, unless precautions are put in place, the level of these pollutants in fertiliser and subsequent food contamination is likely to increase. In turn, this could cause increased pressure on health systems and institutions within countries to deal with the impacts.

4.4 Conclusion on impacts of phosphorus consumption

The current state of our phosphorus use is affecting all three pillars of sustainability: the environment, the economy and society. These impacts are interlinked and, although this makes the situation complex, it also means that initiatives to improve the sustainable use of phosphorus could produce positive impacts in several sectors. For example, at a very general level, by using less phosphate rock and recycling more phosphorus, we can decrease eutrophication and other environmental impacts associated with mining, alongside stabilising the phosphorus market and improving food security.

More sustainable phosphorus use could also have positive impacts in terms of energy and water use. This possibility for addressing multiple issues is considered in more detail in Section 5.

5. Possible responses to improve sustainability of phosphorus

According to the DSPIR framework, RESPONSES can occur at any part of the chain between driving forces and impacts (Kristensen, 2004). Awareness of the DRIVERS and subsequent PRESSURES can provide insight into possible larger social and economic shifts or adaptations that need to occur to support phosphorus sustainability such as diet changes, design of city infrastructure and development of a recycled resource market.

Research conducted on the STATE of phosphorus imports, flows and losses highlight where potentially large efficiency gains can be made in agriculture, mining, fertiliser production, food processing, retail and consumption. Phosphorus flow analysis has been done at a variety of scales from the city to river basin and can therefore inform responses at various levels. An understanding of the various IMPACTS, both current and in the future, provides insight on possible mitigation measures in terms of protecting the environment, society and the economy.

This section will outline the possible responses to improve the sustainability of phosphorus and summarise the research into their effectiveness.

5.1 Reducing mining losses

As mentioned in Section 3, there are significant losses from mining and processing of phosphate rock. There is no clear picture of the level of these losses although Kippenberger (2001) estimated that about a third of the phosphorus in rock is lost during mining and processing, whilst another estimation puts the loss through transportation and handling at 10% (Lauriente, 2003). Technological and logistical improvements in the mining process can reduce losses and increase extraction efficiency.

Technology to remove or reduce cadmium in fertiliser could also improve efficiency by decreasing the losses due to contamination. Purification at an earlier stage is likely to be the most effective and affordable option (Ahmed *et al.*, 2007). This will not only mean less phosphate rock has to be discarded because of its cadmium content but there is reduced probability of negative impacts on the environment and human health due to cadmium pollution from both mining and the use of fertiliser. However, there needs to be more evaluation of possible negative environmental impacts of the cadmium removal processes, such as GHG emissions, to ensure these do not outweigh the positive environmental impacts

Another approach to reduce losses from mining and processing is to further develop safe and efficient methods to recover phosphorus from phosphogypsum, approximately 110 million tonnes of which are produced every year (see Section 4). Despite having high phosphorus content, virtually all remains unused due to concern about the levels of radium and uranium. Technologies are being developed to make phosphogypsum usable and remove these radioactive contaminants.

5.2 Agricultural efficiency

A large number of phosphorus flow studies have focussed on agriculture and indicated the large losses from this system (see IMPACTS). There is a range of measures to minimise these phosphorus losses, including reducing erosion in soils, exporting manure, genetic modification of crops and more efficient use of mineral fertiliser, which are summarised here.

5.2.1 Optimising land use for phosphorus efficiency

This can involve using land only with a sufficiently high potential level of productivity for agriculture rather than using large amounts of fertiliser on less productive or marginal land. It can also involve ensuring that land does not have excessive amounts of phosphorus from, for example, animal manure.

5.2.2 Reducing erosion

Erosion can occur via a number of routes: water erosion, wind erosion and erosion due to soil particles adhering to crops pulled from the ground such as sugar beets. It is difficult to quantify the level of soil erosion, let alone translate this into phosphorus losses, especially as it depends entirely on the soil itself. However, although uncertainties are likely to remain, losses do occur and there are initiatives to address these.

There are several potential measures to improve the capacity of the soil to hold water and therefore minimise runoff. These include a reduction in tillage whilst keeping crop residues on the land (mulching), terracing the land, inserting buffers of vegetation to hold the run-off, and converting arable land into grassland and reforestation (Louwagie *et al.*, 2009). Bans on the spreading of fertilisers and manures on frozen or snow-covered land also reduces the risk of phosphorus loss.

Natural or engineered landscaping can absorb and solidify phosphorus so as to reduce runoff into nearby rivers and lakes (Kröger *et al.*, 2013). For example, wetlands are a feature that can reduce phosphorus run-off and these can be natural or artificial. There is growing interest in ecological engineering, which uses ecosystem services and minimum human intervention, for example, the use of vegetative buffers and wetlands. Hydrological management, such as controlled drainage, both on and below the surface, can also help reduce phosphorus loss (Kröger *et al.*, 2013).

5.2.3 Maintaining soil quality

The presence of phosphorus alone does not ensure its absorption by plants. A number of soil qualities can influence uptake of phosphorus by crops. These include the right level of acidity (pH) and organic matter as well as a good level of biodiversity within the soil. It is a balance of these soil characteristics that determines the availability of phosphorus to plants. For example, phosphorus will become less available if the pH is suboptimal, the soil contains too much or too little water or the soil is too compacted. If soil quality is low, more phosphorus is required, usually in the form of mineral fertilisers, to attain the necessary yield. Kröger *et al*., (2013) suggest that the addition of substances to soil such as biochar or calcium-rich substrates, such as gypsum, can increase phosphorus availability to crops. However, on the whole, actions taken to optimise soil quality will need more localised research to ascertain exactly what is required to improve quality in certain locations

5.2.4 Fertiliser (organic and mineral) recommendations and placement

As mentioned, the uptake of phosphorus by crops is not simply down to its presence in the soil. Not only does it depend on soil quality, but also on the age and species of the crop. As such, assessing the amount of available phosphorus in soil is not easy, which makes it difficult to make sound recommendations on the amount of phosphorus needed to maintain sufficient crop growth. As reflected in Section 3 there are high phosphorus surpluses on many European soils. Römer (2009) estimates that 70 to 80% of soils in European countries have an average or high-level of phosphorus, which could maintain yields for several years without phosphorus fertilisation. Römer calls for a revision of the system of recommended levels, and this is supported by Jordan-Meille (2009) who concludes that soil tests used to assess availability of phosphorus vary widely and can provide different recommendations for the same soil.

As well as improving soil assessments to inform fertiliser practice there is large potential to tailor agricultural practices to encourage the most effective uptake of phosphorus. For example, placing fertiliser a small distance below the seeds can enhance plant uptake because it is nearer to the expanding root system. Crops grown in rows may also benefit from positioning the fertiliser near to the crop row. According to research in the Netherlands (van Dijk, 2003), twice as much phosphorus is needed for similar yields if fertiliser is spread onto the soil surface rather than placed sub-surface, close to seed rows. The extent of the benefit will depend on location and will be lower in some contexts but in general there is an indication that a considered placement of fertiliser is more effective than spreading it in a uniform blanket.

There is also a need to address the local concentrations of phosphorus, which occur when parts of fields become over-fertilised or when animals cluster in search of shelter or water. This patchy distribution does not induce efficient use of phosphorus. There is a need for more localised research to ascertain this inequality in distribution and the specific needs of a geographical area.

5.2.5 Improving uptake through the use of certain crops and symbiotic fungi

(A) Selection of crops

Some plants have certain characteristics that provide them with a better ability to acquire phosphorus from the soil, for example the length, branching and distribution of roots all influence a plant's phosphorus-absorbing capacity. Plant species with a good capacity to develop root hairs are more efficient in exploiting phosphorus in the soil and in fertiliser, and it has been shown that there is significant genetic variability in the length and density of root hairs between different varieties of many of our major crops (Gahoonia & Nielsen, 2004).

Perennials (plants that live for more than one year) appear able to provide good yields even when phosphorus levels are low because they have extensive root systems that can absorb nutrients from a larger volume of soil more or less permanently. Annuals (plants that only live for one year) typically have smaller root systems and there is only a short period of time when they are actively absorbing phosphorus. Growing perennial instead of annual crops could theoretically be a promising way to use phosphorus more efficiently (Scheinost *et al*., 2001). However, it will require a massive investment in plant breeding programs to change agricultural production from the dominant annual crop based food supply of the world today (Jensen, in correspondence, 2013).

(B) Genotype improvement

In basic terms, phosphorus use efficiency depends on two traits or genotypes: uptake efficiency as covered in 'Selection of Crops' but also utilisation within the plant i.e. the production of crop component, such as seeds or grain, per unit of phosphorus. Little research has been conducted on the ways to optimise these two traits although some has been done on the variation of certain characteristics helpful in phosphorus absorption, such as root hairs (Gahoonia & Nielsen, (2004), see above). However, it is difficult to know if sufficient genotypic variation exists to breed plants that will have the required combination of characteristics. Decisions to invest in such research depend on value for money it provides and it may be that other options such as crop management or soil improvements provide better cost-efficiency.

(C) Symbiosis with arbuscular mycorrhizal fungi

The capacity of crops to take up phosphorus from soil can also be improved through a symbiosis with beneficial arbuscular mycorrhizal (AM) fungi. Grant *et al.* (2005) propose this is due to the AM fungi enlarging the root system. The nature of the relationship is not certain but it is thought to be beneficial since crops suppress the AM fungi when there is a high phosphorus level in soil. It should be noted, however, that some management techniques such as tilling and crop rotation might not support AM fungi.

(D) Low phytic acid mutants

Phytic acid is the principal form of storage of phosphorus in many plant tissues, especially bran and seeds, but it is not digestible by humans or non-ruminant animals (those with one stomach). Therefore the use of plants with low phytic acid content for food and animal feed may promote more efficient use of phosphorus and reduce phosphorus fertiliser requirements. Some research has also investigated the possibility of genetically modifying animals to digest phosphorus more efficiently. So-called Enviropigs produce phytase in the saliva, which helps their digestion of phytic acid (see http://www.uoguelph.ca/enviropig/).

5.2.6 Exporting manure and livestock distribution

With the sanitation revolution and intensified livestock farming (see Section 1) the traditional way of recycling phosphorus by returning excreta to the land has become difficult to sustain. Cities have become hotspots for phosphorus build up from human excreta, and certain parts of farmland with high livestock density have large phosphorus surpluses from animal excreta. Solutions to this could be attempted by encouraging a more even distribution of livestock rather than dedicating large areas of land to specialised crops (monocultures) and intensive animal farming. A return to smaller scale farming with mixed systems of crop and livestock would allow the supply and demand for manure to be in closer proximity.

Another approach is the exportation of manure and excreta to land that is in need of phosphorus. This works best when the distance of exportation is minimal, for example, excess manure from a livestock farm is exported to a neighbouring horticultural farm. In order to tailor it to the nutrient needs of the land in question, there may be potential for using the liquid fraction to provide nitrogen (without a build-up of phosphorus) and the phosphorus-rich solid fraction for those soils depleted of phosphorus. A dewatering process achieves this separation into solid and liquid, and also makes transportation easier.

Animal manures have the potential to supply up to 50% of all phosphorus required for agricultural use in Western Europe and 25% of that required for the United States (Smil, 2000). However, the bulky nature of manure, its uneven spatial distribution and the separation of its source (livestock production) from where it can be used (crop production) are all barriers to using it to its full potential. Patterns of phosphorus surpluses in areas of livestock production and phosphorus deficits in areas of crop production occur throughout the developed world (Smil, 2000) and intensified farming has exacerbated this (Bateman *et al.,* 2011). With dietary changes causing more demand for meat and dairy, these trends could be heightened (Cordell *et al.*, 2009).

Recycling of phosphorus from animal manure could play a large part in meeting future phosphorus needs. However, in order to overcome the spatial and temporal mismatches (see Box 12) a logistical management system is needed to ensure cost-effective and environmentally compatible redistribution of the phosphorus in manure. Dewatering animal manure at a regional level can facilitate transport and storage and may maximise phosphorus recovery and minimise economic and environmental impacts.

Such a flexible management system must take into account other wastes sources that may be disposed on land, for example, in Northwest Germany, manure, sewage sludge and residues from biogas plants all compete for the same land area for disposal. In addition the management system must respect legislation that is in place at national and EU levels. For example, the Nitrates Directive designates farmland as Nitrate Vulnerable Zones and dictates a closed period in the year during which the spreading of manure with high nitrogen content is prohibited.

5.2.7 Adjusting livestock diets

The response of reducing livestock numbers or adjusting their distribution has been mentioned above but it may also be possible to adjust the amount of phosphorus in livestock diet and therefore in manure. Firstly, older animals could be given food that is less rich in phosphorus (so-called phase feeding) as these animals produce less in terms of milk or meat.

Secondly, the daily ration of individual animals could be tuned to their production level so they are only fed phosphorus-rich food at times during the year when it is needed. Lastly, the use of artificial enzymes (so-called phytases), could improve the availability of phosphorus in food to animals. Phytase is already added to pig feed in all the main pork producing countries and regions such as the Netherlands, Denmark and Brittany.

Research in the Netherlands by Van Krimpen *et al.* (2010) has estimated that using grass silage with low phosphorus concentrates could reduce the amount of phosphorus in their manure by 20% without decreasing the economic performance of the farm. The same piece of research estimated that in the case of phase-feeding in pig farming, the addition of phytase to their food alongside diets with low levels of indigestible phytase could reduce the amount of phosphorus in manure by 25%. Again the research indicates the adoption of these measures would not compromise the economic performance of farms.

Box 12. Recycling manure in England

Bateman *et al.* (2011) analysed the potential for England to recover and use phosphorus from animal manure and identified some of the challenges to recycling. They estimated that the total phosphorus content of housed livestock wastes is 80,700 tonnes. With a phosphorus requirement for crop production of 113,7000 tonnes this means the phosphorus in manure could supply 71% of the requirements.

A regional study indicated that six of the eight regions in England had a deficit in phosphorus whilst two regions had a surplus. In order to rectify this imbalance an annual transfer of 4700 tonnes of phosphorus (or 2.8 million tonnes of manure) must take place from the west to east of the country.

The study indicated there is also a temporal imbalance between supply and demand of animal manure, where housed manure production peaks between October and February, requiring 23,000 tonnes to be stored until it can be used on crops between March and May when nearly 60% of annual phosphorus fertiliser is required.

5.2.8 Conclusion on agricultural efficiency measures

Some of these measures require relatively little cost or effort such as improving fertiliser recommendations and application methods and adjusting livestock diets. In comparison, investments in erosion control, the maintenance of soil quality and the development of new genotypes are more costly.

In Kröger *et al.*'s review (2013) of downstream approaches to phosphorus management in US agricultural land, such as use of wetlands and drainage, they suggest that the most appropriate approach will depend on geology and cropping practices, i.e. the right practice is needed for the right place.

Even if scientists, engineers and land managers have a firm understanding of soil phosphorus dynamics and transport processes, finding the right practice for the right place requires an integrated and holistic approach that involves a range of stakeholders, particularly the farmers.

5.3 Efficiency in food commodity chain

Phosphorus losses also occur between harvest and food consumption and these can be reduced through increased efficiency within the chain. Better practices and management can lessen losses and spillages that occur during crop storage, processing and trade. Shortening food production and consumption chains by producing food closer to the point of demand reduces the likelihood of losses occurring. It also reduces waste that occurs when food has to be stored and transported long distances, and the use of energy and other resources for the actual transportation. Placing greater value or subsidising locally sourced produce as well as encouraging urban and peri-urban agriculture can contribute to improvements in this area.

Wastage and spillage also occurs during the sale of food but probably one of the biggest losses is within the home itself when edible food is thrown away and not consumed. At an individual and household level this could be lessened by improved meal planning, shopping to reduce wastage, use of leftovers and avoiding the discard of foods that have passed their use-by date but are nonetheless edible. At a food industry level, changes in package size and unfeasible quality standards could reduce unnecessary food waste. Unavoidable waste can be composted to enable phosphorus recovery for local reuse.

5.4 Reconnection of phosphorus cycle – recovery and reuse from waste streams

5.4.1 Closing the loop and the Fourth Revolution

In the past, excreta and urine were directly recycled to land (see Section 1) but urbanisation has made recycling more difficult to achieve (see Figure 19). The requirements of cities and megacities mean that the simple recycling of excreta is not suitable for sanitation and health reasons. Recycling from the sanitation system is difficult because the current centralised waterbased systems dilute the nutrients. In addition, the primary focus of sanitation systems is to remove waste rather than to recover phosphorus so they are not configured for this role and will require adjustment (Mihelcic *et al.*, 2011).

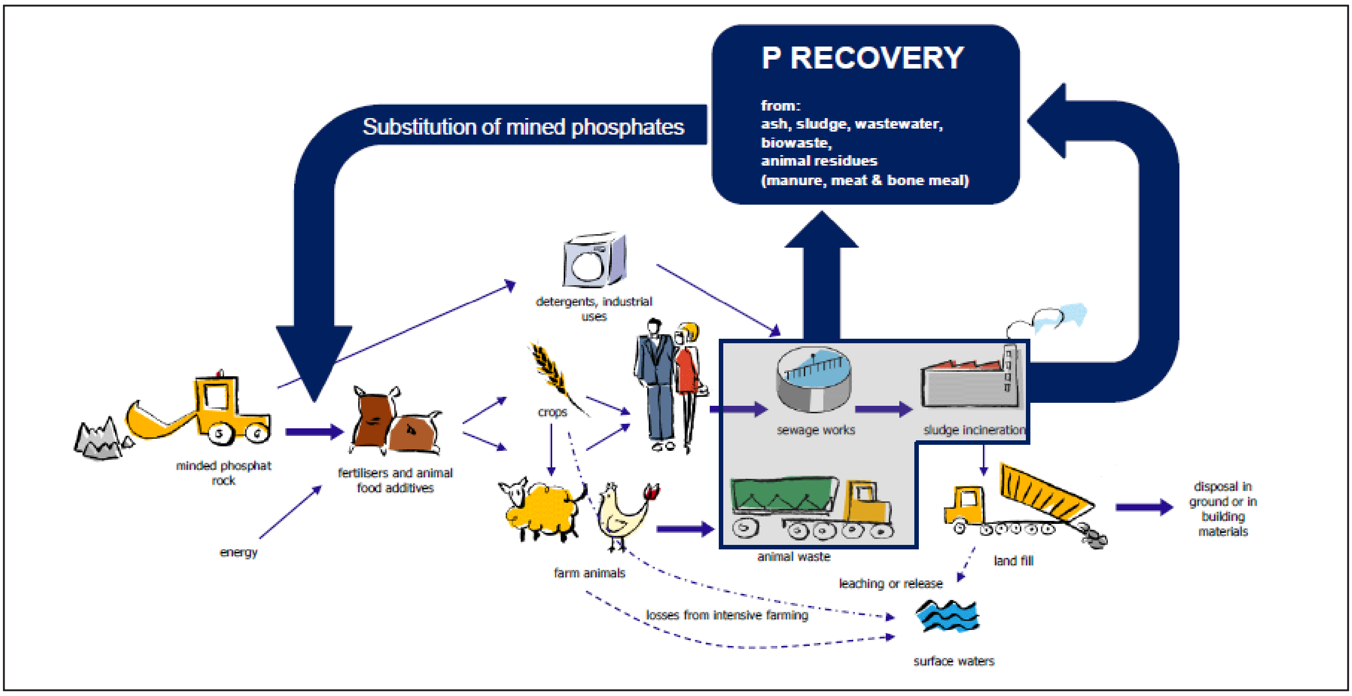

Figure 19: Closing the loop (C. Kabbe and R. Mancke, 2013a)

However, numerous techniques have been developed to recover phosphorus from a range of waste sources. They vary from low-tech small-scale solutions, such as direct urine use to high-tech, large-scale solutions, such as recovery of struvite (magnesium ammonium phosphate) from Wastewater Treatment Plants (WWTPs). In fact for the recovery of phosphorus from WWTPs alone, Schenk *et al.* (2009) and Hermann (2009) identify over 30 processes and, since 2009, the number has increased. The development and design of these techniques is strongly influenced by the motivation behind their creation. For example, the removal of phosphorus from wastewater was developed primarily to prevent the nutrients from entering the waterways where it can lead to eutrophication (Driver *et al.*, 1999) and the reuse of sludge in agriculture was a secondary reason. If its use as a fertiliser is the main motivation for development, then a technique that produces purer phosphate with good plant availability is desirable, such as struvite recovery from WWTPs.

5.4.2 Sources for phosphorus recovery

There are a number of sources from which phosphorus can be recovered and reused. For example, mixed wastewater streams or separated waste fractions, such as urine, faeces and greywater. Animal manure is another source (see Section 5.2.6), as are crop residues, animal carcases, and slaughterhouse and food waste.

Globally, human urine and faeces make up about 14% of lost phosphorus (60-70% contained in urine) and these are the largest sources of phosphorus coming from urban areas. Mihelcic *et al.* (2011) estimated that, globally, the potential phosphorus content of human urine and faeces for 2009 was about 1.68 million tonnes, with a similar amount available from faeces. Together this could account for 22% of the global annual phosphorus demand. Due to population growths and the move towards a protein-rich diet, the phosphorus concentrations could be even greater in the future.

Mihelcic *et al.* (2011) suggest that Africa and Asia could benefit the most from recovering phosphorus from human excreta because their sanitation systems are in their infancy and would not suffer the barriers of retrofitting onto existing systems as in developed countries. For example, regions in Africa and Asia could use systems based on urine diversion (see small scale recovery in Section 5.4.3) and reuse, or composting or recovery of solids that have previously not been integrated into existing systems. There is an opportunity to provide sustainable technology to 2.6 billion people that lack sanitation and incorporate efficient recycling of phosphorus.

Animal manure is widely used in some rural areas in the EU, but in others it is heavily restricted. The amount of phosphorus in manure depends on the livestock numbers, animal type and feeding regimes for manure. It also suffers logistical difficulties for distribution (see Section 5.2.6). The use of animal parts such as blood and bone has decreased markedly since the bovine spongiform encephalopathy (BSE) outbreaks in the 1990s.

Recycling of food waste is becoming more commonplace as household composting increases across Europe. The amount of phosphorus within food waste varies from place to place, depending on the population size and diet.

Local pressures and drivers influence the form of phosphorus recovery that is adopted. In countries such as Sweden, Switzerland and France, severe and recurring algal blooms have motivated advances in phosphorus removal from municipal wastewater treatment systems.

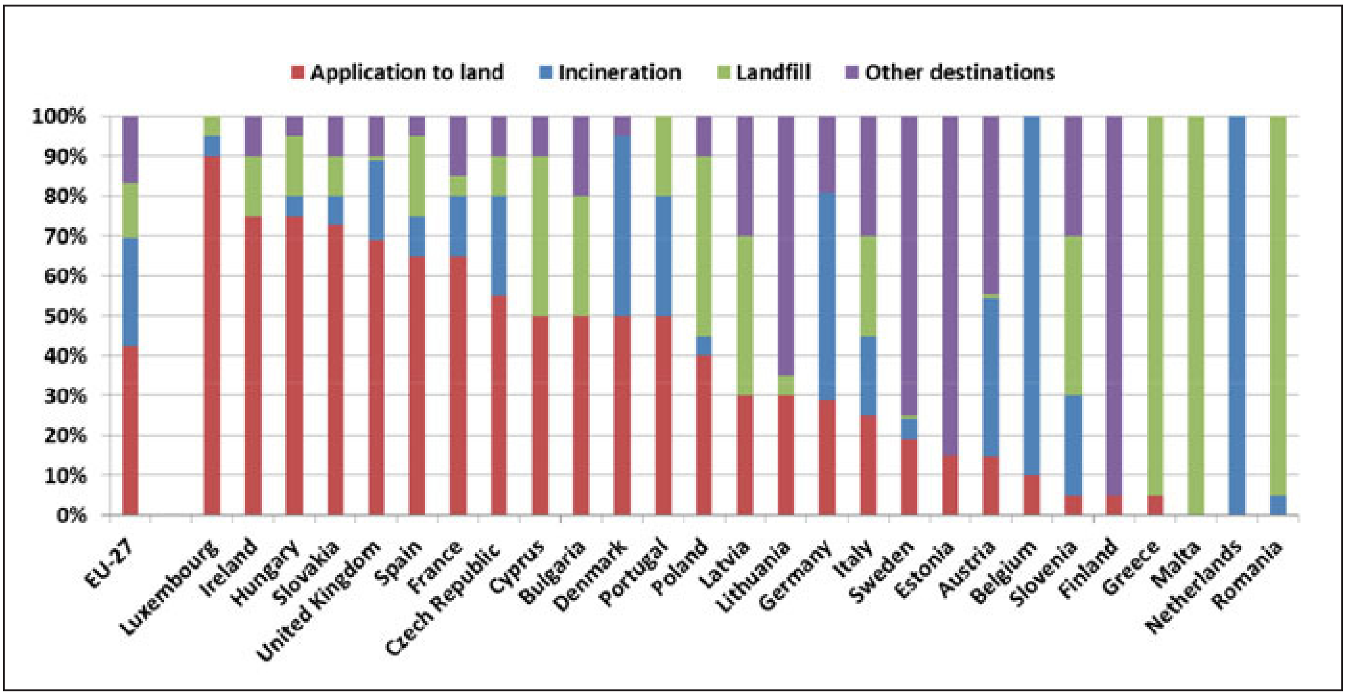

Figure 20: Destinations of sewage sludge in EU-27 (2010), from Van Dijk *et al.*, (2013)

Regulation plays a significant role and in the Netherlands and Switzerland where the spreading of sewage sludge on the land is severely restricted, the incineration of sewage sludge has become popular and there is some recovery of phosphorus from the ash. Figure 21 indicates the variation in how sewage sludge is used.

5.4.3 Techniques and processes to recover phosphorus from wastewater

(A) Small-scale decentralised

This refers to systems that use waste from individual houses, sites or communities. In Europe technologies have been developed to separate urine, faeces and flush water so that recycling can be achieved more effectively. This form of ecological sanitation is usually decentralised to occur at the household or community level. However, once separated and collected, the excreta still needs to be stored and transported which requires some form of centralised storage system and logistics for transport. Some small-scale decentralised systems can concentrate the nutrients after recovery (Ganrot *et al.*, 2009) and this concentration at source can overcome some of the issues around storage and transport.

There are numerous examples of these systems around the world including Southern Africa, China, Vietnam and Mexico and, in the developed world, Scandinavia, the Netherlands and Germany. Although effective at a local or community scale, Qiao *et al.* (2011) suggest that such technologies will need adequate testing before they applied at a larger-scale.

In some examples the systems also produce biogas, which can be used as a local energy supply. In India and Brazil projects have collected blackwater (wastewater containing urine and faeces) and greywater (wastewater from washing and bathing) and anaerobically digested them to produce biogas that can be used for cooking and lighting and sludge for fertiliser on agriculture and gardens. Other small scale systems include composting toilet and food waste, treatment of household greywater for irrigation, reusing sludge from biogas production, and onsite industrial waste treatment and reuse.

These smaller decentralised processes have the advantage over centralised systems of reduced energy consumption, costs, water use and raw material use. For example, 50 to 70% of the costs of a centralised WWTP can be in the establishing and running of the sewage network to convey or transport the waste materials etc. In addition, separating wastes at source allows them to be treated according to their different properties, which cannot be done with a mixed centralised wastestream. However, there are barriers to the success of the decentralised systems, such as the lack of required land space (particularly in dense urban areas) and the logistics for storage and transport. There are also challenges around the responsibility for management and maintenance at an individual or community level and scaling up the systems to serve larger populations (Schröder *et al*, 2010; Qiao *et al*, 2011).

(B) Large-scale centralised recovery

Techniques to remove phosphorus from centralised wastewater systems have developed over several decades. Their original motivation in the 1970s was to remove nutrients to reduce pollution of waterways so the focus was on the removal of phosphorus and not on the recovery of phosphorus in a high quality form. However, over past decades, there has been growing interest in its reuse and the development of a range of techniques to recover phosphorus in a reusable form.

Removal is generally done by converting phosphorus from the dissolved form into a solid form that is not necessarily suitable for reuse or recycling. Recovery, on the other hand, generally involves converting dissolved phosphate to a solid form that can be recycled or reused (Milhelcic *et al.*, 2011). A range of centralised techniques that recover phosphorus are described below, alongside a summary of the research evaluating their feasibility. There are many different varieties of technologies and combinations of technologies.

i) Biological Uptake (Enhanced Biological Phosphorus Removal – EBPR)

This uses bacteria (polyphosphate accumulating organisms or PAOs) to remove phosphorus from sewage sludge. Like all living organisms, bacteria need phosphorus to survive and under anaerobic conditions they will remove phosphorus from sewage sludge to produce polyphosphate within their own cells. The bacteria are then separated from the treated water at the end of the process and the phosphorus removed. In Europe, EBPR is used in about 30% of WWTP (Kabbe, in correspondence, 2013) and is often performed before the precipitation of struvite (see below) to concentrate the nutrients. In most plants it occurs in huge tanks with aerobic and anaerobic zones but other methods are being developed (see Box 13). It has the additional benefit of producing biogas which can be used as energy.

ii) Precipitation (including struvite recovery)

Chemical precipitation to remove phosphorus from wastewater has been established since the 1950s and more recently this has shifted focus to the recovery of phosphorus. Precipitation can occur spontaneously but is usually triggered by the addition of a metal ion, most frequently magnesium or calcium, for recovery. One of the most common techniques recovers phosphorus as struvite (magnesium ammonium phosphate hexahydrate).

Precipitation is often performed on sludge that is digested anaerobically i.e. micro-organisms have broken down the sludge in the absence of oxygen (see above). This sludge has a high concentration of phosphorus and ammonium in its liquid component. There are two options for struvite precipitation: before or after the dewatering process (2a

Box 13. Enhanced biological phosphorus removal (EBPR)

Kodera *et al.* (2013) proposed a new process using EBPR in the form of a biofilm enriched with polyphosphate accumulating organisms (PAOs). This controls the PAOs so they absorb and release phosphate by alternating between aerobic (with oxygen) and anaerobic (without oxygen) conditions. The concentration of phosphorus in resulting solution is 25 more times concentrated than the original wastewater.

and 2b in Figure 21, respectively). Dewatering separates the sludge liquor from the solids and the precipitation can be conducted either within the sludge directly after anaerobic digestion (2a) or subsequent to the dewatering process within the resulting liquid part or sludge liquor (process water in 2b). Precipitation is done by the addition of magnesium ions.

Struvite precipitated from the sludge liquor has fewer impurities and produces a slightly higher quality product than struvite precipitated directly from the sludge before dewatering. However, struvite recovery directly from sludge provides several operational benefits. Firstly, removal of phosphorus before the dewatering actually increases the efficiency of the dewatering process so subsequent separation is easier. Secondly, the precipitation process from sludge requires fewer chemicals than precipitation after dewatering. Thirdly, there are lower maintenance costs as, once the phosphorus has been removed, there is less clogging of pipes and abrasion of centrifuges. Finally, depending on the country, there are lower disposal costs for sludge if the phosphorus has already been removed. In some companies it has been estimated that, overall, these benefits could produce a reduction in operational costs of several hundred thousand euros per year, and this is without the financial gains from selling the struvite (Kabbe, 2013b).

Several studies have demonstrated that struvite contains similar components to standard fertilisers, such as diammonium phosphate and superphosphate (Rittmann *et al.*, 2011). It has been argued that in certain contexts struvite is superior to regular manufactured fertilisers because it is a slow-release fertiliser that does not 'burn' roots when applied in excess, unlike some ammonium-phosphate fertilisers. However, slow-release fertilisers may not always be appropriate, particularly if using the technique of precision fertilisation (see Section 5.2.4) since there is uncertainty around the timing of their effects. It is proposed this slow-release characteristic is good for coastal agriculture, as it reduces run-off (Shu *et al.*, 2006) and for turf growing, orchards, forests, potted plants, golf courses and ornamental lawns (Gaterell *et al.*, 2000).

The process has been applied commercially by several companies in Northern America, Asia and Europe and there is continuing research to evaluate the large-scale feasibility of a range of processes such as in

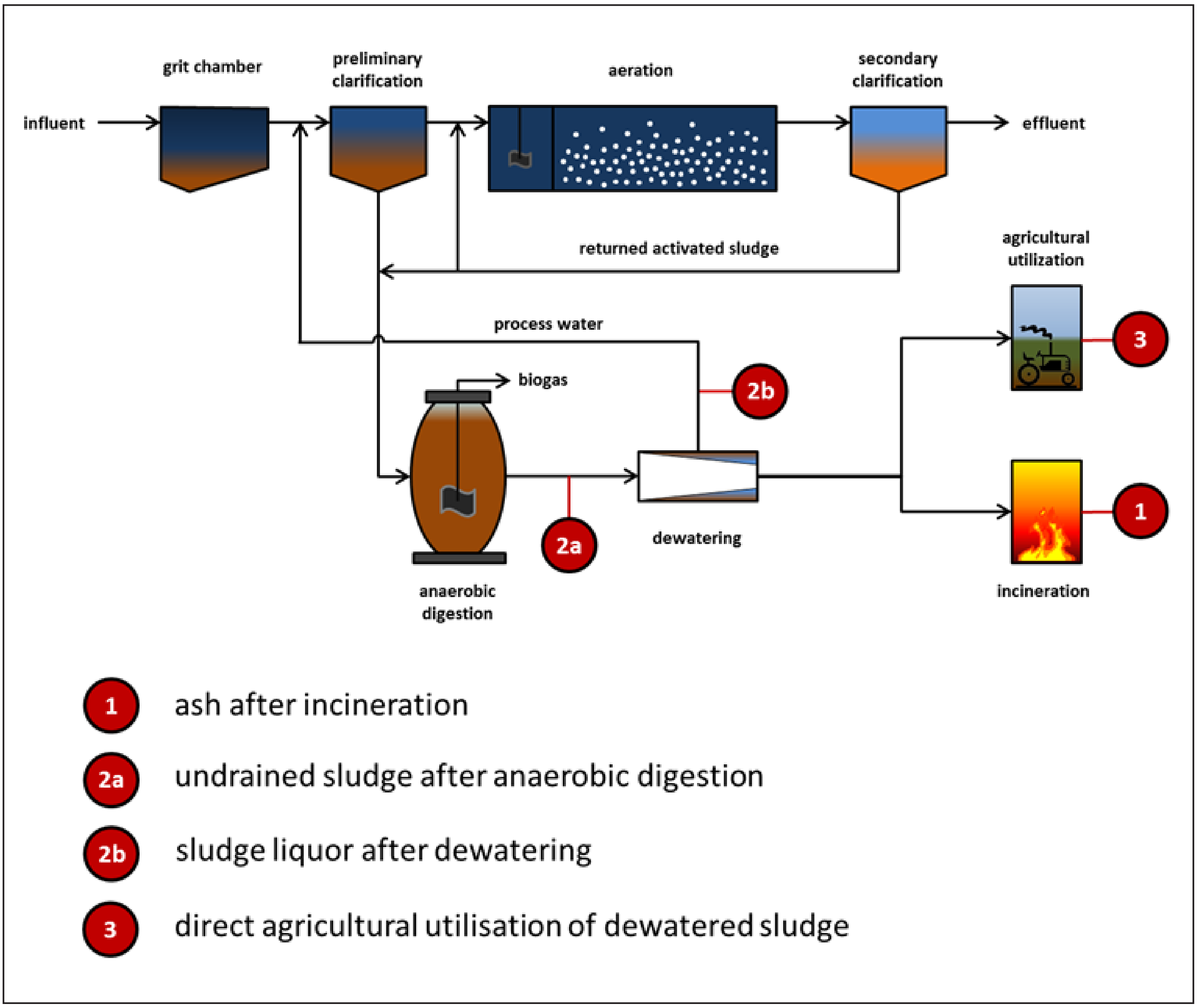

Figure 21: Centralised options for phosphorus recovery from the wastewater treatment systems. (Kabbe, C. 2013b)

the P-REX project (see Box 15). There are various methods that could potentially optimise the recovery process. For example, a proven option to enhance the struvite yield is to use thermal hydrolysis on the sludge before anaerobic digestion.

Thermal hydrolysis basically boils the sludge under high temperature and high pressure to disintegrate it. This increases the production of biogas in the anaerobic digester and improves struvite recovery by transforming insoluble or hardly soluble phosphates into a more soluble form so they are available for struvite precipitation in the sludge liquor. There are plans to demonstrate the process in full-scale in some German WWTPs (Kabbe, in correspondence, 2013).

Bradford-Hartke *et al.* (2012) suggest that the recent awareness of water shortages in some countries and need to recycle water could improve the potential to recover phosphorus. In Australia there is growing development of water recycling projects using either microfiltration or nanofiltration membranes to filter blackwater. As well as producing high quality water, this also concentrates the waste streams so they have a high level of phosphate, making them more accessible for struvite recovery (see Box 14). However, this system is very energy intensive; an important consideration when choosing a recovery system.

iii) Adsorption

An adsorption process transfers phosphorus from a liquid to solid by using another substance to attract and hold the phosphorus to its surface. This is often done using by-products from other industrial processes, such as aluminium sulphate or ferric chloride. The technique of adsorption can also be used to manage phosphorus in lakes (Spears *et al.*, 2013) by using by-products or commercially produced substances to strip the phosphorus from the water and settle it on the lake bed. However, the phosphorus is strongly bound to these elements meaning it is not readily available to plants and although new phosphorus adsorbents are being developed, such as polymers and nanomaterials, their impacts on human health and the environment must be fully assessed.

iv) Ion exchange

This involves removing phosphorus by exchanging its ions for those of another substance. Compared to adsorption it provides a more selective means of separating ions from solution and has recently gained recognition as a promising process for phosphorus removal. However, its effectiveness has only been proven at a small scale with only a limited number of studies conducted on municipal wastewaters (Rittmann *et al.*, 2011).

v) Incinerating sludge to ash

The incineration of sludge cake into ash is a viable method to concentrate phosphorus and produce a pure, high quality, reliable product (process 1 in Figure 21). It is particularly popular in the Netherlands and Switzerland where the spreading of sewage sludge on farmland has been heavily restricted since 1995. However, the process is energy intensive and costly (Linderholm *et al.*, 2012). In order to produce a usable phosphorus fertiliser it requires treatment to remove contaminants and make the phosphorus available to plants. There have been attempts to make the ash marketable and some fertiliser companies use it in their portfolio of different products.

vi) Wastewater fed aquaculture

Wastewater fed aquaculture is a similar concept to EBPR (see Section 5.4.3 (i)) and makes use of biological treatment. Algal ponds already in use for wastewater treatment can be harvested to recover the phosphorus incorporated in algal biomass. The process could be optimised to increase more phosphorus content, for example, in the case of algae, phosphate content could be increased from 1% to 3% by altering light intensities and temperatures (Shilton *et al.*, 2012). Wastewater can also be treated in wetlands using plants that form mats with roots suspended in water so the whole plant (including roots) can be harvested. Compared to recovering phosphorus from terrestrial plants, aquaculture requires about one tenth of the area (Shilton *et al.*, 2012). There is also potential to expand algal-based phosphorus recovery to offshore sites.

Box 14. Modelling the feasibility of struvite recovery

Zheng *et al.* (2004) estimated that approximately 1 kg of struvite can be precipitated from 100 m^3 of wastewater. Using this figure, Shu *et al.* (2006) demonstrated that struvite recovery was economically feasible for an Australian WWTP, estimating that recovery of 1 kg of struvite would reduce the operating costs of the plant by AUD $1.1333 by reducing the amount of sludge to be handled and disposed. They estimate a five-year payback period for the introduction of struvite plant that processes 55,000 m^3 of wastewater per day. A commercial plant in North America has a 90% recovery rate from the dewatered sludge liquor (not the wastewater) yielding 500 tonnes of struvite per year. The plants are costly to set up but it has been estimated that the costs can be recovered in three to five years (Britton *et al.*, 2009). However, the outcome of Shu *et al.*'s (2006) model is highly reliant on the initial estimate by Zheng *et al.* (2004). In reality the amount of struvite will depend on the phosphorus content of the wastewater and the technology, so may not be as feasible as this study suggests.

Bradford-Hartke *et al.* (2012) modelled the potential for recovering struvite from wastewater using a nanofilter membrane and an anaerobic reactor, which can also generate electricity from methane produced. This indicated the system would produce about 59-62 kg of phosphorus per day and require 260 kW of power to recover 1 kg of phosphorus. However, the study does not really address the feasibility of WWTP operators adopting this technology in terms of the costs and barriers of installation, running and maintenance, for example.

vii) Conclusions on large-scale phosphorus recovery
In general, centralised recovery systems have the potential advantages of economies of scale and the ease of end-of-pipe additions. Centralised systems also have the potential to generate marketable products that could be used on a par with fertiliser from mined phosphate. However, the start-up costs are large and they are perceived to have an increased risk of 'technological lock in' i.e. committing to a technology which may be superseded in the future. Therefore, operational benefits are currently an important trigger for implementation. Already there are several full-scale phosphorus recovery facilities in the EU but these tend to be concentrated in certain countries (see Figure 22).

There are constant developments in the field of phosphorus recovery and 'additional processing steps' that can improve efficiency but these can come at economic costs. Balanced decisions need to be made about investing in technologies but these decisions should not be delayed for much longer. To inform decision-making in this area there is a need for evaluations and comparisons of the different processes at a real-life scale and the P-Rex project (see Box 15) is aiming to meet this need by bridging together academic knowledge and application.

Box 15. The P-Rex project

For industry to have the confidence to implement new technologies in the market there needs to be proof of their capability and feasibility. The P-REX project will demonstrate technical solutions for phosphorus recovery and recycling in full-scale.

Based on real operational data, the performance and feasibility of a range of available processes will be assessed, as well as the quality of their recovered products. This will be combined with an analysis of the market barriers and potential to develop strategies to provide recommendations for efficient and wide-spread phosphorus recovery to substantially increase the European phosphorus recycling rate from municipal wastewater.

The project started in September 2012 and runs until August 2015. See http://www.p-rex.eu.

5.5 Potential recovery from other phosphorus sinks

Seawater
i)Seawater contains phosphorus but researchers have concluded that the concentration is too low for cost-effective recycling via current available methods. It has been estimated that seawater would have to be processed at a rate of 2.4 x 105 km^3 per year, more than 70 times greater than global consumption of freshwater (Duncan Brown, 2003).

ii) Sediments from oceans
A potential future resource that is not often considered is offshore phosphorite nodules where phosphate has been deposited in clusters scattered throughout the ocean floor and where rivers flow into oceans. There are some pilot projects underway to recover phosphorus from these deposits in New Zealand and Namibia but, before full-scale action goes ahead, the environmental impacts must be carefully assessed (Scholz *et al.*, 2013; van Vuuren *et al.*, 2010).

Figure 22: Full-scale operation and demonstration of technical phosphorus recovery from wastewater stream in the EU 2013/2014. (Kabbe, C. & Mancke, R., 2013)

5.6 Taking a systems approach to phosphorus recovery

Ideally phosphorus recovery and reuse should not have a negative impact on energy security i.e. it should not result in a net increase in consumption of energy produced by fossil fuels. More research is needed on the life cycle of phosphorus recovery and the possibilities for using renewable energy. Some individual studies exist such as a comparison between urine separation and recovery with the production and use of mineral fertilisers in the Swedish context (Tidåker *et al.*, 2007).

However, larger scale and more comprehensive analyses are required across the range of recovery systems. Again this is addressed in the P-Rex project (see Box 15), which analyses barriers and enabling conditions for phosphorus recovery initiatives. Failure to address the whole system could lead to costly technologies and possibly conflict with related services (Cordell *et al.*, 2011).

Phosphorus recovery could also have implications for water and sanitation provision as well as food production and environmental protection. As such, recovery systems need to consider future trends and drivers within these sectors to ensure they do not cause unwanted side-effects. 'Sunk' infrastructure costs can be a barrier, where previous investment in infrastructure causes a reluctance to invest in alternatives. However, it has been suggested that many western sanitation and water systems are due for an overhaul, which could provide an opportunity to introduce new technologies. When that does occur the technology should be ready to work at a realistic scale. In developing countries there is the potential to bring in recovery systems from an early stage in city planning.

Trends or developments in other areas may also impact phosphorus availability. For example, the generation of biochar from agricultural waste to replace coal can permanently remove the phosphorus contained in the original biomass from the food system or convert it into an unavailable form (Cordell *et al.*, 2011)

Linking phosphorus scarcity to other sustainable development measures, such as climate change, food security, freshwater use and sanitation, can help meet challenges (Neset & Cordell, 2011). For example, a change from meat and dairy intensive diets to plant-based diets can have positive effects for both phosphorus availability (see Box 16) and climate change. Phosphorus recovery from wastewater can also go hand-in-hand with treatment to recycle water for consumption and address issues of water scarcity.

These potential win-win situations require integrated planning that use life cycle analyses (LCAs) to assess the impact of these approaches and ensure that energy requirements and environmental impacts are improvements on the use of traditional mineral fertiliser use. LCAs provide very valuable information but still need to be considered in the context of what is feasible, as demonstrated by Linderholm *et al*'s (2012) LCA analysis of four phosphorus recovery options in Sweden (see Box 17). It must also be remembered that LCAs are only as good as the data they use and the assumptions they make about the boundaries of the system and what is included in the analysis. To some extent this is also exemplified by the example in Box 17, which does not consider the fact that phosphorus in sewage sludge is not 100% available to plants so would not be as effective as some of the other options of applying phosphorus. In addition it only considers cadmium although several other pollutants can be present in sewage sludge. Overall there is a call for better data from industry itself on the GHG emissions

Box 16. Combination approach

Suh & Yee (2011) estimated that only 15% of the phosphorus extracted for the provision of food in the US is eventually eaten by humans. Since the average dietary input in the US is excessively phosphorus rich they suggest a reduction in phosphorus intake by 30% to 1.77g per day per person. This, in combination with a 30% improved phosphorus use efficiency and food waste reduction, could reduce phosphorus resource inputs by half. However, this is dependent on the original estimate of percentage of phosphorus in food that is eaten by humans and some have calculated this to be higher.

Box 17. Life cycle analysis (LCA) to compare options of phosphorus recovery in Swedish agriculture

LCA can be used to compare options of phosphorus recovery. Linderholm *et al.* (2012) assessed four alternative phosphorus fertilisers for use in Swedish agriculture: mineral fertiliser, certified sewage sludge, struvite recovery and phosphorus recovery from sludge incineration. The LCA compared the options in terms of impacts of global warming, energy use and eutrophication. Cadmium flows were also assessed. Their results indicated that direct use of sewage sludge on farmland uses the least energy and produces the least GHG emissions. Recovery from ash uses the most energy and produces the most GHG emissions, however, it does contribute the least cadmium to soils whereas direct use of sewage sludge contributes the most cadmium. Struvite recovery uses comparatively little energy and has low levels of cadmium. However, it may not be suitable for Sweden as it has very few WWTP with anaerobic digesters (which are necessary for struvite recovery) since these require higher temperatures to work effectively. As such, it will require large start up costs to install struvite recovery into Sweden's current infrastructure.

and other environmental impacts of mining and processing. This will be facilitated by the work of platforms that bring together industry, academia and policy, for example, the Dutch Nutrients Platform (http://www.nutrientplatform.org) and European Phosphorus Platform (http://www.phosphorusplatform.eu) and by projects such as the P-REX project (see Box 15).

5.7 Integrated response – weighing up the options

It is unlikely that there is one single solution or response that will be most effective to promote sustainable use of phosphorus. Integrated approaches are needed that use both supply side measures (e.g. phosphorus recovery from mine waste, sanitation and food waste) and demand side measures (e.g. increasing efficiency in fertiliser and food production). When implementing responses there is also a need to integrate the viewpoints of all relevant stakeholders, such as the farming and fishing community, the fertiliser industry, general public and nature conservation NGOs.

Different locations and different scales (regional, local, river basin) require different data to inform decisions. A useful input into the decision process is the use of phosphorus flow analyses (see Section 3) to compare different options (see Box 18a for European examples and 18b for examples in China and the US).

At the EU-27 level, Van Dijk *et al.* (in preparation) analysed the possible effects of four different scenarios that combined different supply and demand responses. These were as follows:

1) No import of mineral phosphorus fertiliser (for a range of reasons including inability to import due to trade restrictions or increasing cadmium levels in fertiliser);
2) No import of mineral phosphorus fertiliser and no import of animal feed phosphorus;
3) No import of mineral phosphorus fertiliser plus the use of best management practices (BMPs) such as increased agricultural efficiency and phosphorus recovery;
4) No import of mineral phosphorus fertiliser and no import of animal feed phosphorus plus the use of BMPs.

Results show that all response scenarios have an effect compared to the business-as-usual scenario (BAU) (see Figure 23). The BAU scenario is represented by the EU-27 phosphorus flows for the year 2005 with no changes in phosphorus use and recovery. Stopping imports of fertiliser and animal feed phosphorus produced the largest decreases of phosphorus surpluses in soil and runoff. However, this will reduce the capacity of agricultural soils to maintain the present production of crops when application levels of phosphorus are low (buffering capacity) and this could mean that ultimately there will be a decrease in crop productivity. The implementation of best management practices could potentially counterbalance this to an extent by recovering phosphorus and increasing the efficiency of its use to improve crop productivity.

Box 18a. Using phosphorus flow analysis to target responses in the EU

A number of studies have conducted phosphorus flow analyses (see Section 3 for more information of the studies) and evaluated possible responses. Some of these are reviewed below:

(i) **Options on a city level in Sweden.** Kalmykova *et al.* (2012) studied phosphorus flows at a city level in Gothenberg, Sweden. On the basis of their findings they investigated the impact of possible management strategies. According to the modelling, the separation of 70% of the food waste from households and businesses would allow a **total of 88 tonnes of phosphorus per year** to be collected and returned to agriculture. Implementing urine diversion has similar potential whilst blackwater and food waste separation would enable the most complete recycling of nutrients, with a transfer of **245 tonnes of phosphorus per year back to agriculture**

(ii) **Options on a national level for the Netherlands agricultural system.** De Buck *et al.* (2012) considered a range of scenarios to potentially reduce the phosphorus surpluses on agricultural land in the Netherlands which are estimated at 22,000 tonnes in 2008. Firstly they modelled the impacts of meeting the 2015 phosphorus application standards **set by current EU Nitrate Directive** and estimated this would decrease the surplus to **10,000 tonnes**.

The study also considered tailoring phosphorus application to the soil and its availability of phosphorus. Balanced fertilisation is the application of phosphorus in line with the phosphorus removed from harvested products. If balanced fertilisation is used for soils with a neutral to high level of available phosphorus and 2015 application standards are used for soils with low phosphorus availability, this would reduce the phosphorus surplus by **5,000 tonnes**.

Lastly, the report considered the impacts of reducing animal feed imports. **Feeding concentrates containing 10% less phosphorus for dairy cattle** would **reduce the surplus phosphorus by 1,800 tonnes and the same strategy for intensive livestock would reduce surplus by 5,000 tonnes.** Using breeding to enhance the digestive efficiency of cows so they can effectively absorb phosphorus (see Section 5.2.6) **could potentially reduce phosphorus surplus by 4,000 tonnes**. However this would require a large amount of effort and commitment from farmers.

The study also modelled the total intake of phosphorus in food at the household level in the four scenarios. This food intake decreased in all scenarios (Figure 24) but not to a level that was below that needed to fulfil human phosphorus dietary intake requirements (red dotted line in Figure 24).

The household requirement of 0.6 kg of phosphorus per year is based on the Dutch situation, taking into account the highest dietary phosphorus intake and average dietary phosphorus consumption per person (Van Rossum *et al.*, 2011). As such it appears the food production system would be relatively robust to a 70% reduction in the total net import of fertiliser and a 20% reduction in animal feed.

If best management practice (BMPs) are also introduced this would produce increases in the amount of phosphorus in food to above this threshold, since there is an increase in crop efficiency and a decrease in the losses in the system. This suggests that a combination of these demand and supply approaches can reduce the surpluses of phosphorus in the soil whilst allowing good crop yield and ensuring the levels of phosphorus in food to remain at adequate levels.

As these examples indicate, comparing options for recovery is a complicated process. As a basic guide in deciding on which recovery approach to adopt, Cordell *et al.* (2011) have proposed a framework to examine different alternatives. They describe eight steps (see Box 19) which do not have to be done in order, for example it is important to identify institutions and stakeholders right from the beginning.

5.8 Shifts in industries, markets and societies

There are a number of social shifts that would be influential and necessary to move towards more sustainable phosphorus use, in particular shifts in diet and views on using urine as fertiliser. Currently there tends to be 'urine blindness' amongst professionals and householders and a reluctance of farmers to use wastes because of perceived or real contamination concerns. In their assessment of phosphorus flows in the Netherlands, Smit *et al.* (2010) highlight the large potential for recycling phosphorus but identify several barriers: environmental concerns about heavy metals in sewage sludge, hygienic concerns about the direct use of faeces and urine and concerns about spread of diseases such as BSE. Alongside this there needs to be changes in infrastructure and the development of alternatives to Western sanitation systems e.g. decentralised, waste-separation at source, particularly in developing countries.

In order for phosphorus recovery to become feasible on a large scale, there needs to be a market in place for waste and recovery. The advantage of this is that it will allow the EU to play a larger role in the phosphorus market. Already there are initiatives, such as the Dutch Nutrient Platform and the European Phosphorus Platform, which bring together industry, academia and policy.

Seyhan *et al.* (2012) suggest that, since the price of phosphorus is not providing a natural push for the recovery and recycling market, it may need incentives such as subsidies, taxes, investment grants and

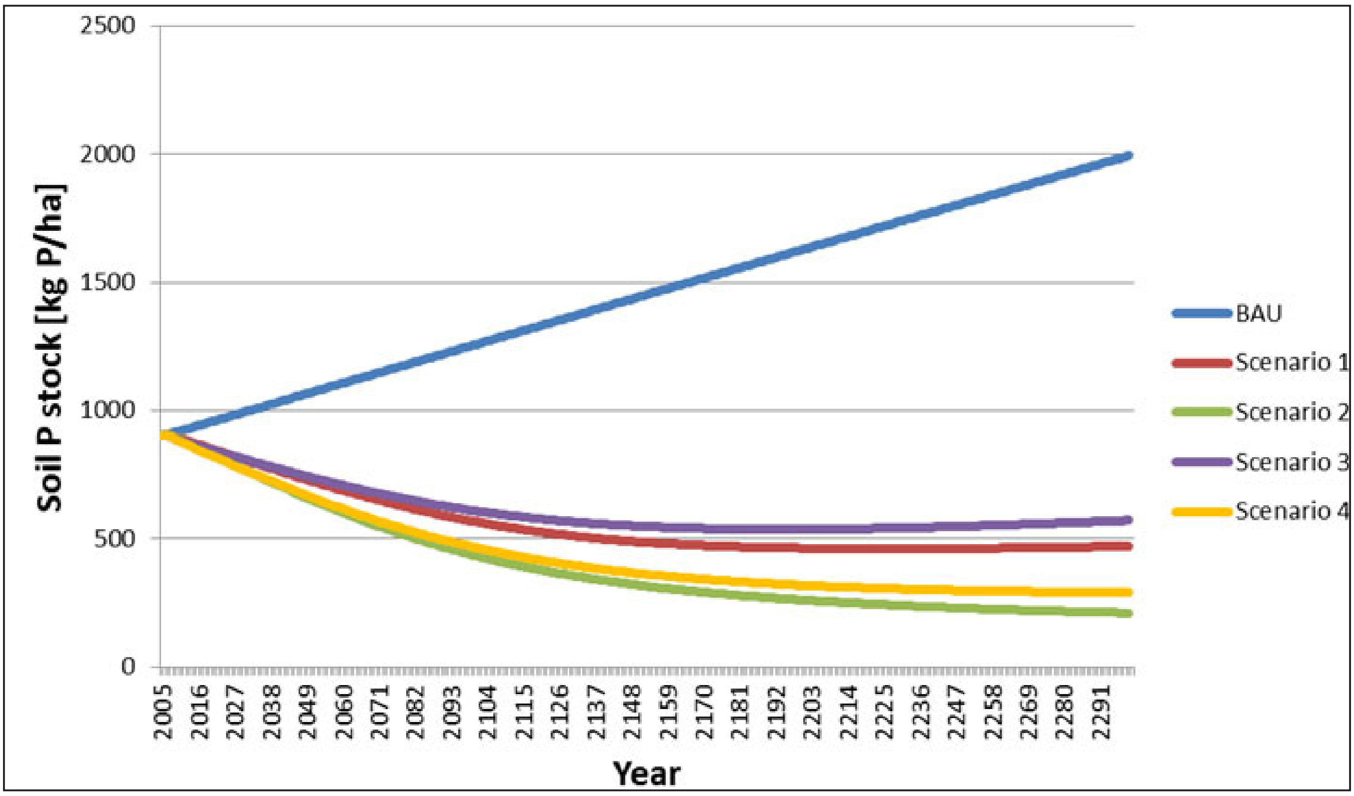

Figure 23: Changes in soil phosphorus stock per scenario for 2005-2300. BAU shows the 'business-as-usual' scenario; see text for explanation of other scenarios. (Van Dijk, 2013).

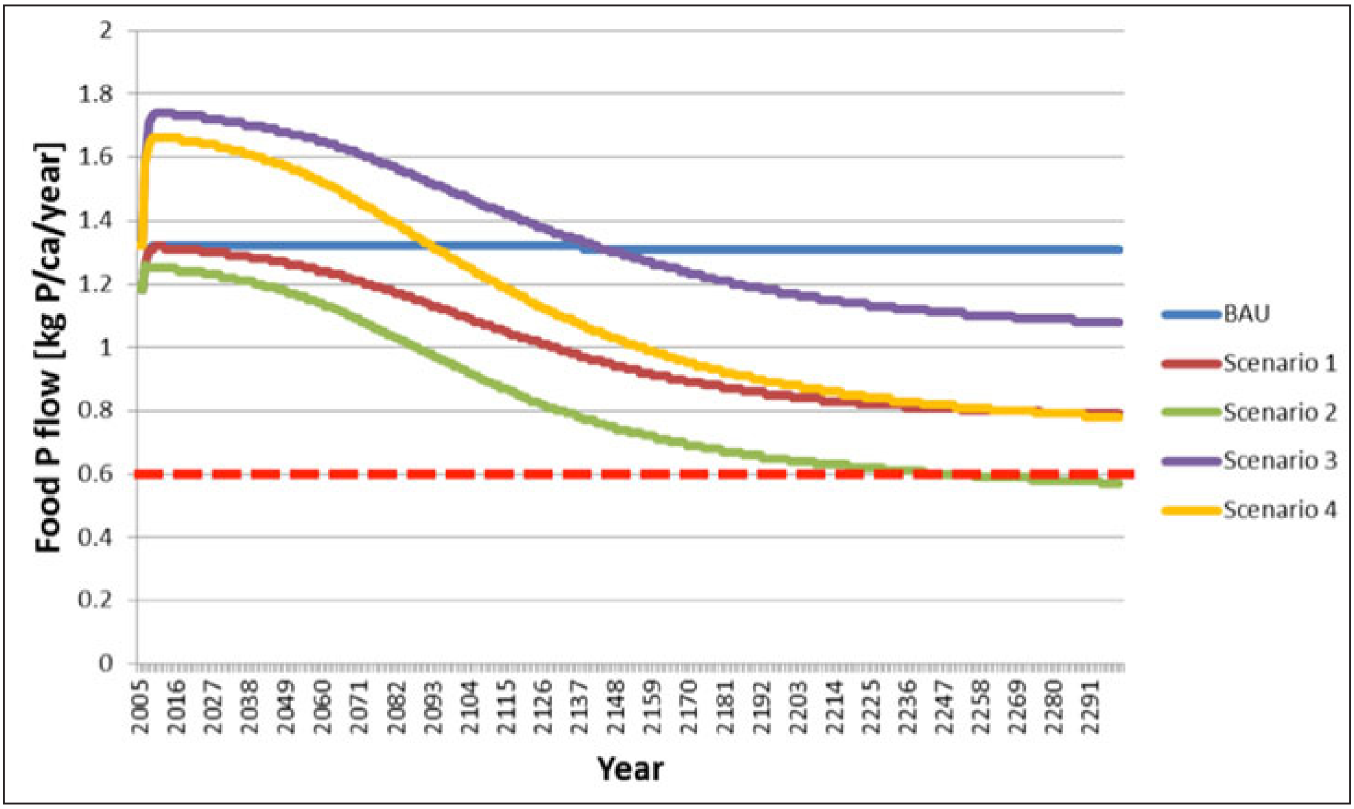

Figure 24: Changes in total phosphorus in food per capita in EU-27 per scenario for 2005-2300 see text for explanation of each scenario. (Van Dijk, 2013)

Box 19. Cordell *et al.* (2011) framework for examining different options for phosphorus recovery

1) Identifying drivers or motivations for recovery which can shape decisions, for example, pollution prevention or renewable fertiliser production
2) Defining the system boundary e.g. household, city, food system, country
3) Quantifying available phosphorus from different sources and potential phosphorus recovery rate
4) Identifying recovery systems and techniques from small-scale to large-scale (this will depend on system boundaries and available phosphorus)
5) Identifying logistics of collection, storage, transport and use. This will include analysis of physical distance between source and end point and the land space available for intermediate storage and reuse.
6) Identifying life-cycle costs to ensure this is an environmentally feasible operation
7) Identifying synergies and conflicts with the aim of maximising the former and minimising the latter
8) Identifying institutions and stakeholders - these include phosphorus mining companies, fertiliser manufacturers, farmers, food producers, dieticians, consumers, water and sanitation service providers, environmental managers and policymakers. Early engagement with stakeholders can ensure a socially robust system that is more likely to be implemented. Institutional arrangements must also be considered, including policies and regulations.

the support of demonstrations or trials. However, when designing economic instruments, care must be given to the possibility of unwanted side effects. For example, a tax on mineral fertiliser could make livestock farmers compensate for the cost by importing more animal feed rich in phosphorus. This 'hidden' phosphorus must be considered in policy responses.

Industry also needs to undergo some shifts. Not only in terms of using recovered phosphate in fertiliser but in terms of selling fertilising or food security services, rather than fertiliser products (Cordell *et al,* 2011). For example, the concept of 'integrated plant nutrient management' proposed by the International Fertiliser Industry Association (IFA). More research into upscaling recovery technology into feasible real-life solutions is needed, as in the P-Rex project (see Box 15).

The food industry could also play a role by a transition to local self-sufficient food chains rather than importing crops, feed and food

Box 18b. Using phosphorus flow analyses to target responses in China and the US

(i) Different options for different crops in China Ma *et al.* (2010) investigated phosphorus flows in China for three different crops: maize, wheat and rice (see Section 3). As food demand in China is so large, it requires a management strategy that can increase efficiency and crop yield. The study suggested different responses for different crops; for wheat and rice the most effective strategy would be to increase phosphorus uptake by crops and phosphorus recovery from food. For maize the key points where action could be taken are again the uptake of phosphorus by crops, but also recovery from plant food and more efficient use of phosphorus in animal feed (for which maize is mainly used).

(ii) Options on a regional scale in China Bi *et al.* (2013) mapped the phosphorus flows for Wuwei County in China (see Section 3), which encompasses a major lake that is a key target for water pollution control. The study analysed three potential measures to mitigate loss of phosphorus. Focusing on the agricultural system, the use of precision fertiliser techniques and increased use of manure would eliminate at least **470 tonnes of phosphorus discharge** into the lake, but it would require the hiring of technical personnel and education for rural residents. The second option is to promote phosphorus waste recycling through the construction of a sewage treatment plant and village biogas pools. It is estimated that this would **reduce discharge by 179.4 tonnes**. However, the construction costs are likely to be large and would have to be covered by rural residents. Thirdly, the study examined the possibility of promoting multiple recycling of phosphorus waste including livestock waste, which would **eliminate 291 tonnes of phosphorus discharge into the lake.**

(iii) Options on a watershed scale in US. Baker *et al.* (2011) studied the phosphorus flows in Twin cities Watershed in the US (see Section 3) and investigated possible solutions to improving phosphorus sustainability. This study proposes several techniques to reduce phosphorus inputs: changing diet to that of 30 years prior to the US obesity epidemic, banning lawn fertiliser, laws to reduce phosphorus in automatic dishwashers and reducing the size of dogs through an education campaign. The study estimated that adopting these measures would r**educe phosphorus inputs by 15%, reduce the amount stored in the system by 70% and reduce leakage by 74%**. The study calls for greater understanding of how diet choices and food production systems influence phosphorus flows between cities and agricultural systems. Since sustainability implications of phosphorus conservation are so dependent on the site or area under consideration, Baker (2011) also suggests that, ideally, studies should include a number of cities that span a range of conditions.

products. Global food chains could also close the phosphorus cycle by striking a better balance between the supply and demand of phosphorus (van Dijk *et al.*, 2012).

5.9 Responses to eutrophication

Remediation after eutrophication is complicated and ecosystem recovery can take decades. In most cases the disturbed ecosystem never returns to its former self due to changes in the food-web, climate and the social and economic pressures (Fulweiler et al., 2012).

Nevertheless this does not mean it should not be attempted. More and improved research is needed to monitor existing conditions and model future scenarios to provide effective mitigation and remediation strategies (Fulweiler *et al.*, 2012).

Adsorption techniques can be used to control the release of phosphorus in lakes and Spears *et al.* (2013) compared the adsorption ability of six industrial waste products and one commercial product in a UK lake. This demonstrated that all except gypsum were capable of reducing phosphorus concentrations, with sander dust and mag dust showing the greatest capacity. However, more research is needed on potential ecotoxicological impacts of these materials.

5.10 Policy responses

The role of policy in improving the sustainable use of phosphorus is an important but sensitive one. With so many stakeholders involved, as well as different policy sectors and scientific disciplines, policy has to walk a difficult line to ensure fair decisions are made. The EU already has a number of policies in place that emphasise the need to reduce nutrient losses from soil to surface water and groundwater in order to meet European water quality targets. Examples of policy initiatives in this direction are the Directives on Bathing Water (76/160/EEC), Sewage Sludge (86/278/EEC), Urban Waste Water Treatment (91/271/EEC), Nitrates (911/676/EEC), and on Integrated Pollution Prevention Control (96/61/EEC), all of which are part of the EU Water Framework Directive (De Ridder *et al.*, 2012). Recently the EU published its Consultation Communication on the Sustainable Use of Phosphorus[7] to gain insight from relevant stakeholders.

Although there is continuing debate about the amount of phosphorus reserves, it must be recognised that the sustainability of phosphorus is an issue of high concern. Research in this area has been providing the same messages for some time, but as yet there is a reluctance to put this high on the policy agenda. It has been proposed that the inclusion of phosphorus on critical list of strategic raw materials could at least initiate awareness and eventual action (Houttuin, 2013).

[7]http://ec.europa.eu/environment/consultations/phosphorus_en.htm

At a national level, some action has been taken, both as a result of implementing EU legislation and from national initiatives. This has taken a number of forms. Several countries, most notably the Netherlands, Germany, Sweden and Finland have set limits on the use of agricultural spreading of sewage sludge in terms of its quality and maximum heavy metal content (more stringent than EU limits). So much so that since 1995, regulatory requirements in the Netherlands have prevented almost all use of sewage sludge in agriculture. However, one of the major consequences is that incineration of sludge without reuse of the final ashes has become one of the main disposal routes for phosphorus.

In Sweden, regulation on the use of cleaner sludge and requirements for it to audited and certified has improved the quality of Swedish sewage sludge. For example, in 1998 average sludge contained 43 mg of cadmium per kg of phosphorus, whilst ten years later the content was 29 mg of cadmium per kg of phosphorus (Linderholm *et al.*, 2012). Limits on levels of cadmium in fertiliser could produce an incentive to increase uptake of some recovered phosphate fertilisers as these have lower cadmium levels than traditional mineral fertilisers. There have been proposals at the EU level to regulate the amount of cadmium in fertiliser, for example, setting maximum concentrations that become increasingly strict over a period of time. Some EU Member States have already set some kind of national limits, such as Sweden, Germany, Finland and Austria, but there is reluctance from the fertiliser industry to impose a EU-wide limit. It is not certain what level of investment will be needed to implement decadmiation technologies and how much the cost would add to the price of mineral fertiliser. One estimate puts the increase at 5–10 % of today's price (von Horn & Sartorius, 2003). Another consideration is the potential advantages and disadvantages such a regulation would give to different market players, depending on the level of cadmium that their reserves contain.

The Swedish government has also proposed an interim target for recycling 60% of phosphorus in sewage for use on productive land by 2015. It also stipulates that at least half of the recovered phosphorus should be used on arable land (Swedish EPA, 2005). The aim of this target was to necessitate measures to improve sludge quality and to recycle phosphorus from sludge. Various reports were written to inform the action plan for the recovery of phosphorus from waste and evaluate different possibilities for phosphorus recovery. The effectiveness and cost of various options were summarised by Hultman *et al.* (2005) and are reported in Box 20. However, these are relatively old estimations and based on the assumption that 100% of the phosphorus in sludge is recyclable when in reality its availability to plants is limited.

On the basis of this, Hultman *et al.* (2005) concluded that even the most cost-effective systems for phosphorus recovery could not be motivated alone by possible sales of recovered phosphorus at current market prices. Other shifts are needed to encourage phosphorus recovery, such as the development of more cost-effective technologies to reduce operational costs, legal requirements for level of recovery to provide a reliable motivation to recover phosphorus and 'green' taxes on phosphorus use and discharges to include societal cost.

Economic tools could help to encourage more sustainable phosphorus use, especially as phosphorus price alone will not be enough to rein in our use of mineral fertiliser. The German Federal Environment Ministry (BMU) has suggested focusing on phosphorus recovery in wastewater treatment plants (SCOPE, 2003). It further proposed that existing taxes on wastewater could be used to support the technical development of phosphorus recycling. There has also been discussion in Germany about legislating for traditional fertilisers to contain a certain amount of recycled phosphate fertiliser so that manufacturers have to work to a blending ratio with which to produce this 'hybrid' fertiliser. But this will only work on EU level, since the mineral fertiliser manufactured in Germany covers only 5% of the fertiliser demand and the rest is imported.

De Haes *et al.* (2009) in their policy memorandum in the Netherlands suggest that a tax or levy on phosphate should be considered, as this would provide a stimulus to use phosphate more efficiently and to recycle more phosphate. This stimulus would be even greater if the revenue from the tax was invested in technology to improve more efficient phosphate use and recycling. However direct taxation of fertilisers at a national level can give the country a disadvantage compared to those with no taxes and therefore negatively impact their competitiveness of farmers. There are also more general questions about whether taxes will actually lead to more efficient use and curb consumption of phosphorus. Especially in the context of existing agricultural subsidies, which may counterbalance the deterrent effect of taxes and prevent them from producing the desired result.

Shakhramanyan *et al.* (2012) investigated possible economic tools in the USA. They explored the combination of continued increases in phosphate prices and external damage taxation based on the costs of the negative impacts of eutrophication (Dodds *et al.*, 2009). Their results indicated that this combination could reduce the use of mineral fertilisers and increase the area supplied with organic phosphorus sources e.g. manure.

In addition there would be little impact on crop and livestock production and trade. Although the production costs of farmers would increase, this would be balanced by an increase in income due to market price adjustments. However this would mean a transfer of costs from the farmers to the consumers. Indeed corrective food pricing has been suggested as a way to reduce excessive phosphorus consumption (Suh & Yee, 2011).

Box 20. Evaluation of options for phosphorus recovery in Sweden (Hultman *et al.*, 2005)

- The best recovery (95%) and lowest cost was obtained for direct agricultural use of sludge but this is already restricted by legislation
- Extraction of phosphorus from wastewater use has a recovery rate of about 60% and has an estimated cost of about 14 Swedish Kronor (SEK) per person per day
- Extraction of phosphorus from sludge or ash may recover about 60 to 70% with a cost of 26-42 SEK per person per day
- Source separating systems may involve high costs as much as 550-900 SEK per person per day

Although it is positive that the options for phosphorus recovery from wastewater are a subject for national policy discussion, ongoing and lengthy debate can contribute to the perception of uncertainty amongst WWTP operators. In simple terms, WWTP operators want to dispose of their sludge at costs that are stable in the long-term. However, if they are uncertain which alternative may gain policy support they may opt for incineration, which has relatively well-known costs. As such, clear decisions need to be made at a policy level.

With so many competing interests and views there may be a need for the development of governing structures for phosphorus that cross food production, sanitation and agricultural domains. The European Sustainable Phosphorus Platform aims to promote, develop and implement better stewardship of phosphorus, a greater level of recycling and the creation of green jobs in the circular economy. Such institutional developments could help diminish some of the existing social and market barriers that are delaying the development of large-scale phosphorus recovery. This could also lessen the uncertainty surrounding the adoption of alternative options.

5.11 Conclusions on possible responses to improve sustainability of phosphorus

There is a range of possible responses to improve the current phosphorus situation. Some of these are easier and less costly to implement, such as those aimed at improving agricultural efficiency. Others, such as struvite recovery from wastewater, are more complex and may involve rethinking existing systems and infrastructure and potentially creating new markets.

Research indicates that they are all possible and, as with many cross-sectoral, multi-disciplinary issues, improving the sustainable use of phosphorus will require not one solution but a combination of approaches. Although the price of phosphorus has on the whole been increasing, it is not enough to 'naturally' trigger responses to improve efficiency and increase recovery from waste streams. Policy will have a role to play in encouraging action to be taken.

6. Conclusion

There is an impressive and ever-expanding body of research into the sustainability of phosphorus, to which this report can only begin to do justice. The knowledge and understanding imparted from this research can provide insight into all five stages of the DPSIR framework. This in turn can provide an overview of the implications of current phosphorus use for the three pillars of sustainability: environment, society and the economy, and help assess the impact of political choices.

The past, present and future **DRIVERS** will influence our use of phosphorus and its distribution around the world. Population growth, particularly in cities and developing countries, means more mouths to feed and greater need for sanitation and wastewater treatment systems to deal with phosphorus-rich excreta produced by city dwellers. These changes increase the demand for nutrients, including phosphorus, to ensure land can produce crops for food and also mean there is an excessive amount of phosphorus in cities within the wastewater. Alongside this, urbanisation and dietary changes lead to intensified farming which creates more demand for phosphorus in crops and feed, and more soil phosphorus build-up in areas where livestock are kept.

These drivers produce a number of **PRESSURES** on the availability and sustainability of phosphorus, as well as on the environment. There is plenty of research and scientific discussion surrounding phosphorus reserves but placing a figure on the availability of this vital element is still controversial due to data and methodological issues. More dialogue is needed between research and industry in order to ensure access to more accurate data across the world. Nevertheless, what researchers do agree on is that phosphorus is a finite resource with no replacement. Even if there is enough to meet current demands, its quality and accessibility are decreasing. This undoubtedly means more energy, money and better technology will be required to extract usable phosphate. Mining of phosphate and production of mineral fertiliser also places pressure on water and energy reserves, as they require both these resources. Being almost fully dependent on imported phosphorus, the EU must contribute to the transition towards more sustainable use of this essential element.

As well as pressures on the resource itself, there are **PRESSURES** in terms of land use change due to phosphorus mining and the use of phosphorus to convert land of a relatively low productivity into land that can produce crops or feed livestock. There are also pressures from emissions and pollution which science has highlighted and analysed. Of the greatest concern is water pollution by phosphorus, which leads to eutrophication and there is a large body of research evaluating and quantifying the problem. There is also concern about land pollution due to waste products of the mining process, particularly stacks of phosphogypsum contaminated with toxic metals. Application of phosphate fertiliser can also cause soil pollution since it contains cadmium and radioactive elements such as uranium and thorium. The level of these contaminants is very variable but scientific research has shed light into the sources and processes of this pollution, which can inform responses to limit this potential hazard.

These pressures of resource depletion, land use change and pollution creates an imbalanced **STATE** in the environment. The historical phosphorus cycle has been opened so that phosphorus taken from soil to grow food is no longer replenished by 'waste' flows of humans and animals. Its path is now linear from mine to fork and there are many opportunities for phosphorus to be lost along this path, causing surpluses and deficiencies. The analysis of phosphorus flows provides insight on the location and levels of these losses that, in turn, can inform mitigation strategies. Since we have to take action at many scales from local to global, by agricultural bodies and municipalities, research must be done at several levels. This can be at the global, national and watershed level, and can focus on agricultural or urban systems. However, there are issues around the data and methodology used to estimate phosphorus flows. Although data on levels of fertiliser application are relatively well-documented, other data are more reliant on proxies, such as the amount of manure returned to land from livestock. There are gaps and limitations in the data on phosphorus mining and processing, food production and waste handling, and variability in the quality and availability of data between different countries. This makes amalgamating analyses at a EU or global level a difficult task. As such, there is a need for more harmonisation between data collection and also between the methods used for flow analyses.

Changes in the state of the environment have **IMPACTS** on the functioning of ecosystems, the services they provide and ultimately on human health. The impacts in turn have social and economic consequences. One of the most scientifically documented and visible impacts of unsustainable phosphorus use is the eutrophication of water bodies. The detrimental effects on aquatic ecosystems are well-known and more recently there has been research into economically quantifying the costs of these effects. Other environmental impacts are cadmium pollution from phosphogypsum stacks near mines and from the application of mineral fertiliser on soil. Although the toxicity of cadmium and its presence in phosphate fertiliser is known, there is a knowledge gap about how much of the cadmium within fertiliser actually reaches human food. This means it is difficult to place evidence-based limits on the amount of cadmium that should be present in fertiliser. The placing of limits on cadmium content would be both good precautionary practice to limit impacts on health and the environment, and may well have the additional effect of triggering more sustainable phosphorus use. The possible economic impacts of the current phosphorus situation have also been analysed. Price spikes have already occurred and the uneven distribution of reserves has large implications, especially for the EU, which imports virtually all of its phosphate from abroad. These environmental and economic impacts have social impacts with one of the main concerns being food security.

In the DPSIR framework the **RESPONSES** can feed into any of the other four elements and their impact then feeds back so the response can adapt appropriately. This report outlines a wide range of responses, all informed by research and knowledge on the level and distribution of phosphorus reserves, phosphorus flows and environmental, economic and social impacts and future trends. Although much of the research compares different responses, there is a general recognition that the most effective solution will be an integrated one that combines several approaches and is appropriately adjusted to the context. The phosphorus challenge must be seen from a holistic viewpoint, taking into account other resources such as energy, land, water and micronutrients. On a broad level this could involve approaches that attempt to limit demand, for example, through more efficient use of fertiliser, but also using alternative supplies such as phosphorus recovered from wastewater.

Van Dijk *et al.* (in preparation) propose that opportunities for more sustainable use and mitigation of phosphorus losses could be conducted by a multi-track strategy that consists of a pallet of different options that can be classified as the 4 Rs:

Redefine the system, human choices and diets. This includes rethinking the food production-consumption chain and increasing awareness of the impacts of our choices and diets.

Reduce phosphorus input at the system, sector and process level. Firstly, by restricting the phosphorus input derived from phosphate rock reserves and secondly, by maximising effective output in all processes by improving efficiency.

Reuse phosphorus containing organic materials and by-products, such as food residues, slaughter by-products and compost.

Recycle phosphorus from wastes and leakages to close the phosphorus loop by recovery from various sources (e.g. animal manure, wastewater) and subsequent use in agriculture and industry.

Research has a major part to play in choosing and tailoring responses to achieve effective solutions to the issues of sustainable phosphorus use. However, for such a complex issue it must be remembered that science cannot provide definite answers and guaranteed outcomes. There are a number of research gaps that need to be filled (see Box 21) and already there have been initiatives to work towards meeting this challenge. In 2011 the 1st Scientific European Phosphorus Workshop (SEPW) was held in Bordeaux, France, on 'Designing phosphorus cycle at country scale'[8]. As a follow up, the 2nd SEPW on 'Sustainable use of phosphorus' was organised in the Netherlands in February 2013[9]. This workshop gathered researchers from 21 European countries, and the network is aimed at jointly addressing the key challenges in sustainable phosphorus research. It proposes to do this by: developing a common conceptual framework and standardised datasets for phosphorus accounting in food systems; promoting phosphorus budget calculations in Eastern and Southern countries and by identifying promising technological and organisational innovations for improved phosphorus resource management.

However, although it is important to strive for better data and analysis this must not delay action as there is already a sufficient evidence-base to warrant the initiation of a transition towards more sustainable phosphorus use. Due to the time taken for mitigation actions to be implemented and for these to have an impact, if responses are not put into place soon it may be too late. Research can already inform the picking of some low-hanging fruit that require little

Box 21. Research gaps in the sustainable phosphorus challenge

- Analysis of economic dynamics of phosphorus, including risk and consequences of future phosphorus spikes and investigation into possible responses to inequitable distribution of and access to phosphate.
- New and transparent scenarios on future demand and supply of phosphorus for alternative futures to support decision makers. This requires better and more data on phosphorus reserves as well as phosphorus flows in society.
- More feasible indicators of phosphorus emissions and losses.
- Increased focus on virtual flows of phosphorus i.e. embodied phosphorus required to produce traded commodities to determine real national and regional dependency on phosphorus.
- Identification of specific regional vulnerabilities, hot spots of phosphorus and spatial distribution of supply and demand to identify possible synergy clusters.
- More knowledge on how to make unavailable soil phosphorus available to plants, including research on phosphorus fixation and fertiliser use efficiency.
- Feasibility studies into the export and import of animal manure from areas of surplus to areas of deficiency so it is distributed effectively, both in terms of cost and environmental impact. This could involve spatial mapping of surpluses and deficiencies across the EU-27.
- Life cycle analyses and economic cost analyses of phosphorus recovery systems and comparison to the use of mineral fertiliser (Cordell *et al.*, 2011) Both types of analysis are context-dependent and may need to be done on a case-by-case basis.
- Development of indicators to measure the quality of recovered phosphorus for its use as a fertiliser, both in terms of ability to improve soil fertility and its environmental implications. These could also inform quality standards for the fertiliser industry.
- More research into the application of business models of available technology to instigate a market for waste products and recovered phosphorus products.

8. www.bordeaux-aquitaine.inra.fr/tcem/seminaires_et_colloques/colloques/designing_phosphorus_cycle_at_country_scale
9. www.wageningenur.nl/en/show/Outcomes-2nd-Scientific-European-Phosphorus-Workshop-2013.htm

investment, such as improving agricultural efficiency, more effective use of animal manure, and decreasing the amount of food waste. However, this needs to be combined with real life experience from agriculture and industry to be effective. New technologies will require economic investment that cannot be guaranteed a certain payback, at least not for some years. There may also need to be reliable intervention to create the necessary conditions and markets for phosphorus recovery to become a feasible option. Whatever combination of responses are chosen, the need for their continuous evaluation and feedback must be emphasised in order to, as much as possible, ensure responses are adaptive.

In the European Conference on Sustainable Phosphorus (March, 2013)[10] it was suggested that there are three main scales of research: macro, meso and micro (see Figure 25). The macro level looks at issues such as geopolitics, climate change and sociology using tools including scenario analysis and modelling. The outputs of this form of research inform risk assessments about the criticality of the situation and timescale for action. The meso level is more focused on industry and agriculture, looking at issues of phosphorus accumulation and deficiencies in agriculture, and losses in fertiliser and food production. Research on this level will often involve phosphorus flow analyses to inform more efficient use of phosphorus and loss minimisation, as well as the possible contribution of phosphorus recovery. Lastly the micro level of research is focused on products and processes with the aim of improving efficiency and optimising them for use and recovery of phosphorus, using appropriate indicators.

It is important to conduct research at all these levels but it is also important to integrate them into a coherent message and ensure that the policy decisions based on their findings are working towards the same goals. The development of feasible and coherent goals involves the participation of the wide range of stakeholders that are experienced and influential in this area. This could potentially involve a more social research element to gather and document views. Alongside this there is also a need to create overarching institutions to not only bring together the different voices in this important debate, but to ensure they are aware of the messages and evidence from the various levels of research (see Figure 26).

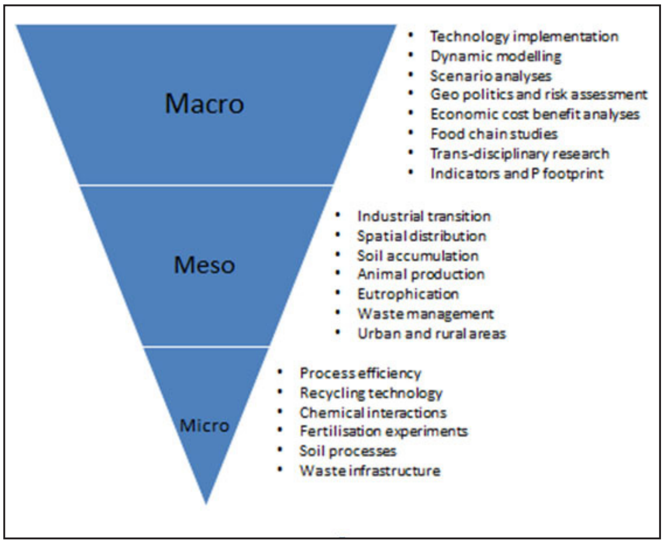

Figure 25: Defining a research agenda for sustainable phosphorus management by the integration of the macro, meso and micro scales of research and knowledge. (Pellerin *et al.*, 2013b).

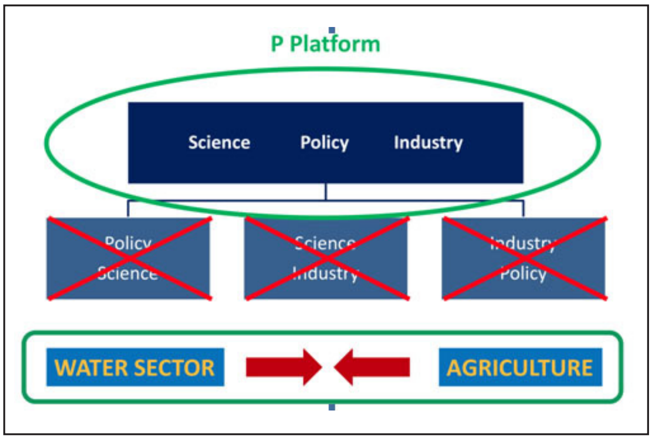

Figure 26: Phosphorus Platform (Kabbe, 2013)

10. http://www.phosphorusplatform.org/espc2013.html

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