The Legacy of Phosphorus: Agriculture and Future Food Security

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This research was conducted under the auspices of the C.T. de Wit Graduate School for Production Ecology and Resource Conservation (PE&RC).

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Thesis
submitted in fulfilment of the requirements for the degree of doctor
at Wageningen University
by the authority of the Rector Magnificus
Prof. Dr M.J. Kropff,
in the presence of the
Thesis Committee appointed by the Academic Board
to be defended in public
on Monday 13 October 2014
at 1:30 p.m. in the Aula.

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The Legacy of Phosphorus: agriculture and future food security.
210 pages.

PhD thesis, Wageningen University, Wageningen, NL (2014)
With references, with summaries in Dutch and English

ISBN 978-94-6257-094-8

Abstract

Growing global demand for food leads to increased concern regarding phosphorus (P), a finite and dwindling resource. Debate focuses on current production and use of phosphate rock rather than on the amount of P required to feed the world in the future. While the time scale of P depletion is debatable, a critical question beyond the physical availability of P is whether P resource depletion can be managed by sustainable consumption of P.

We quantified P demand for cropland as well as grassland in 2050 at global scale. Methods employed included an extensive analysis of historical, long-term P application and agricultural production data and a modeling approach that considers major P flows in a soil-crop system. We applied a simple two-pool soil P (DPPS) model to reproduce historical crop and grass P uptake as a function of P inputs from fertilizer and manure and to estimate P requirements for crop and grass production in 2050. Accounting for legacy P in estimation of the required fertilizer in cropland leads to a reduced fertilizer requirement compared with other studies that did not account for residual soil P. In Europe, Asia, Latin America and Oceania, crop production can benefit from the residual P. In contrast, in Africa more than a five-fold increase in P application is needed to achieve the target P uptake in 2050.

I conclude that the future P requirements from fertilizer in cropland increase less than crop production increases, whereas in grassland the opposite is true. This is because much of the P in animal manure spread in cropland originates from grassland. The transfer of (manure) P from grassland to cropland is not compensated with the transfer of P in livestock feed from cropland to grassland – resulting in soil P depletion of grassland.

To achieve the target crop and grass production in the next four decades a global cumulative P input from mineral fertilizer and manure of ca. 1200 and 1215 Tg is needed in cropland and grassland, respectively. The amount of mineral fertilizer P needed in cropland and grassland systems in total is estimated to be 1380 Tg until 2050, corresponding to 10700 Tg phosphate rock. This amount of phosphate rock is about 16% of the total phosphate rock reserves currently thought to be available on the planet. Thus, we will not immediately run out of P, but mineral fertilizer prices may increase, which may pose a serious challenge to regions with low soil P stocks.

Finally, to provide an example of potential solutions to the global P scarcity, China as a key player in P consumption and production is studied and the feasibility of efficient use of P in China's agriculture is discussed.

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1

Introduction

1.1. Background

Global demand for agricultural products is expected to increase by ca. 60% (Alexandratos and Bruinsma, 2012) in the next decades, with a substantial pressure on agriculture to increase production (Baudron and Giller, 2014). The demand for food is increasing due to the growing world population, while rising incomes might cause a shift from plant to animal proteins in human diets over the next 50 years (Steinfeld et al., 2006). Increasing food demand may cause food prices to rise and strongly affect the developing countries (Koning et al., 2008). In addition, biofuels are grown to meet increasing demand for energy and compete with food crops for land, water and nutrients (Keyzer et al., 2008). It is widely acknowledged that water and arable land are the key resources to meet the global needs for crop production. However, less attention has been paid to nutrients such as phosphorus (P) (Cordell et al., 2009; Smil, 2000).

Increasing crop production requires growth in fertilizer use. Meat and milk production increasingly rely on inputs of feed such as cereals. Phosphorus footprint is defined, as the amount of P required producing annual food consumption per capita. At present, meat and milk consumption are the most important factors accounting for the global average P footprint (Metson et al., 2012).

Phosphate rock (PR) is the primary source of P for fertilizer production. PR is a non-renewable resource that takes 10-15 million years to form from sea sediments (Cordell et al., 2009). Globally 90% of the extracted phosphate rock is used for food production. In the mid 1800s superphosphate was produced for the first time by treating apatite- a calcium phosphate mineral- with sulphuric acid (Van Kauwenbergh et al., 2013). Producing one tonne of phosphoric acid requires three tonnes of sulphuric acid and 3.5 tonnes of phosphate rock. Then phosphoric acid is used to produce both granular and fluid P fertilizers. Practically all P fertilizers today are made by a "wet extraction process", which produces phosphogypsum as a highly polluting by-product (Van Kauwenbergh et al., 2013).

Van Kauwenbergh (2010) defined two different terms for PR, i.e. "reserves" and "resources". "Phosphate rock that can be economically produced at the time of the determination using existing technology" is called Reserve. Resource is a term that encompasses "PR of any grade, including reserves, that may be produced at some time in the future".

The currently known P resource may be rapidly depleting by production of P fertilizer, laundry and dishwasher detergents, feed P supplements and industrial applications. Global PR reserves and resources are dynamic due to a wide variety of factors (Van Kauwenbergh, 2010). Estimates for PR reserves are subject to change with updated information, new discoveries, changes in economic and technology. The USGS estimate (USGS, 2010) for PR reserves reported in 2010 was 16,000 Tg, yet later the IFDC reported 60,000 Tg of reserves for the same year (Van Kauwenbergh, 2010). The USGS revised its estimate in 2011 by a factor of four, from 16,000 to 65,000 Tg PR. Most of the difference in reported PR reserves between 2010 and 2011 came from a revision of the PR available in Morocco (Van Kauwenbergh et al., 2013). Before 2003, China showed a relatively small PR reserve, but in 2003 it revealed the availability of more PR reserves than any other country. New discoveries and availability of information played a major role in the dramatic change in these reserves estimates. Reserve estimates in 2013 for the world's top ten sources of PR are shown in the Table 1-1.

Table 1-1 Reserve estimates for the world's top ten PR reserve holding countries (USGS, 2014).

Country	**Reserve 2013 (Tg)**	**World total (%)**
Morocco and Western Sahara	50,000	75
China	3700	6
Algeria	2200	3
Syria	1800	3
South Africa	1500	2
Jordan	1300	2
Russia	1300	2
United States	1100	2
Australia	870	1
Peru	820	1
Other countries	2170	3
World (rounded)	67,000	100

Currently known reserves are concentrated in a few countries, particularly Morocco and China. Morocco is estimated to have about 75% of the world's PR reserves, while China holds the second place with 6% of the world's PR reserves. Global PR production has increased sharply since 2007 and according to the latest USGS report PR production was 224 Tg in 2013.

Recent estimates indicate that the PR reserves may last 100-400 years (Cordell et al., 2009; Cordell and White, 2011; Smil, 2000; Steen, 1998; Van

Kauwenbergh, 2010). The exact timing of the P depletion is uncertain as it depends on different factors (Neset and Cordell, 2012). Using current reserve and production records, van Kauwenberg et al. (2013) estimated that the world has over 300 years of reserves and over 1400 years of resources. However, PR as a non-renewable natural resource should be managed and used efficiently. Apart from the resource depletion, the quality of PR is declining. The concentration of P_2O_5 in the remaining P-rich rocks is smaller, making it more costly to extract, and the concentration of heavy metals such as cadmium and radium is larger.

1.2. Phosphorus in agriculture

Phosphorus is a chemical element with atomic number 15 and atomic weight 30.97. This element is essential for all living organisms as it is an essential component of DNA and RNA. Neither proteins nor carbohydrate polymers can be made without P (Cordell et al., 2009; Smil, 2000). Furthermore, P is of vital importance in the supply of energy for the synthesis of complex molecules of life. Adenosine triphosphate (ATP) is a nucleotide in cells to transport energy (Smil, 2000; Syers et al., 2008; Tate, 1984). Dairy foods, meat, and cereals are the largest dietary sources of P (Smil, 2000).

In plants, phosphorus plays an essential role in photosynthesis, biological nitrogen fixation of legumes and crop maturation. The main nutrient that constrains nitrogen fixation in soils of the tropics is phosphorus (Giller, 2001). This nutrient is especially important for young tissues for flowering, fruiting and seed formation, root development of the lateral and fibrous rootlets, strength of straw in cereal crops and improvement of crop quality (Brady and Weil, 2008; Smil, 2000). In well fertilized plants, tissue P concentrations are about 0.4-1.5% of the dry matter (White and Hammond, 2008) in case of animals, even a smaller fraction is found in the meat and milk. Although plants need only a relatively small amount of P, without P there is no plant growth and production possible, because no other element can replace it in its vital role (Brady and Weil, 2008; Smil, 2000). That is one of the reasons why agriculture is the largest global consumer of P.

Agricultural soils, plant and livestock production systems are three main P stocks in agricultural P cycles (Beaton et al., 1995). Agricultural systems influence soil P status by P inputs through organic P manure, mineral P fertilizers and crop residues. Agricultural activities including the harvest of plant and animal products, cultivation and tillage and other agricultural-induced activities such as erosion and leaching (Hedley et al., 1995) can change soil P status. On the global scale, soil erosion is the major loss pathway for P from agricultural soils (Smil, 2000). Surface run-off is not only an important loss term, but also contributes to eutrophication of surface water. This undesirable process affects fresh and ocean waters in many parts of the world (Mackenzie et al., 1998; Smil, 2000). When P ends up in ocean's sediments or phosphate rocks, it remains there for millions of years. After uplifting from sediments, finally P is released again through weathering and the cycle starts again (Figure 1-1).

Arable farming depletes soil P through crop P uptake, soil erosion and leaching losses (Sharpley et al., 2001). The threat of P resource depletion and associated effects on fertilizer prices are essential reasons to increase the efficiency of P use in agriculture. In agricultural industry, P can be used in the form of mineral fertilizers, or organic manures (e.g., animal manure, green manure, compost, biosolids, and human excreta) as well as P soil reserves accumulated from past applications (residual P) and natural fertility. P has been used excessively in agricultural systems for decades in many industrialized countries (Smil, 2002). In intensive agricultural systems a long history of P application has built up soil P status, consequently crops show less response to fertilizer applications (Hedley et al., 1995). Availability of manure and cheap P fertilizer contributed to high soil P application rates locally. For instance, in Europe cumulative inputs of P fertilizer and manure for the period 1965-2007 grossly exceeded the cumulative P uptake by crops (FAO, 2011).

As discussed above, the P cycle covers a large number of processes in industry and in nature (Figure 1-1). In this study I focus on P application to soils and P uptake by plants. In the following sections of this chapter, I first discuss the agronomic and soil aspects of P availability to plant roots in soils. Next, I present the objectives of my PhD research, and a brief overview of focus of each chapter of this book. Finally, I give an overview of the tools and methods that I needed to achieve my objectives.

1.3. Soil P and its availability to plants

Availability of soil P to plant roots depends mainly on two factors, the concentration of phosphate ions in the soil solution and the ability of soil to substitute these ions when they are removed by plant roots (buffer capacity) (Syers et al., 2008). Nutrients move to the root by mass flow and diffusion. The main process for P uptake is diffusion, where ions move along a concentration gradient from a higher to a lower concentration. There is a balance between the demand by the plant and the supply from the soil, which influences the extent of depletion around the root area. This balance changes in time as plants grow and more roots are developed (Hinsinger and Donald, 1998; Syers et al., 2008) or when fertilizers are applied or erosion occurs.

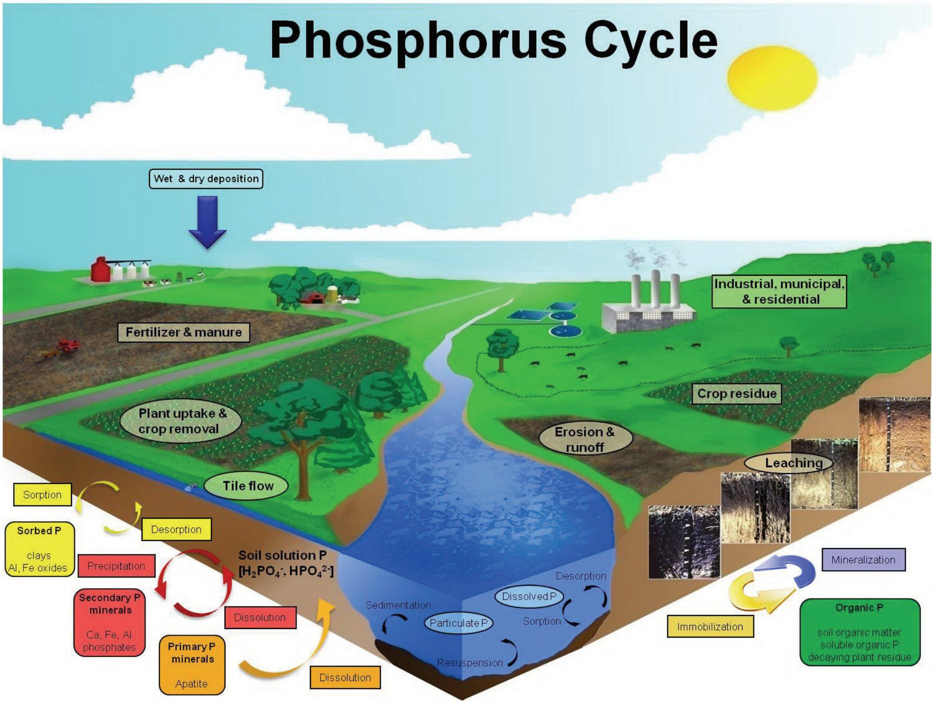

Figure 1-1 Phosphorus cycle (Adapted from Minnesota University; permission granted by Jeff Strock).

The P availability depends on several factors such as soil pH, soluble iron, aluminium, and manganese, presence of iron-, aluminium-, and manganese-containing minerals, available calcium and calcium minerals, amount of decomposition of organic matter and the activities of microorganisms (Brady and Weil, 2008; Nziguheba et al., 1998). There is an important effect of soil reaction (pH) on the solubility and thus the concentration of P in the soil solution. Soils with high soluble Fe and Al, clay minerals, or high Ca activity react with phosphate to form insoluble compounds that are largely unavailable to plants (Figure 1-2). At a higher pH (alkaline soil) P is fixed with calcium and at a lower pH (acidic soil) P is fixed with iron and aluminum (Brady and Weil, 2008). In agricultural soils the pH generally ranges between 4.0 and 9.0 (Figure 1-2). The optimum pH for P plant uptake is 6.0-6.5, as then the greatest amount of P is soluble and available for plant roots (Hoffland et al., 2011; Lindsay and Stephenson, 1959). Plants take up P in the form of $H_2PO_4^-$ and HPO_4^{2-} ions. However, these ions react very fast with the minerals in clays. In very acidic soils P is present as a $H_2PO_4^-$ ion and in very alkaline soils HPO_4^{2-} ions dominate.

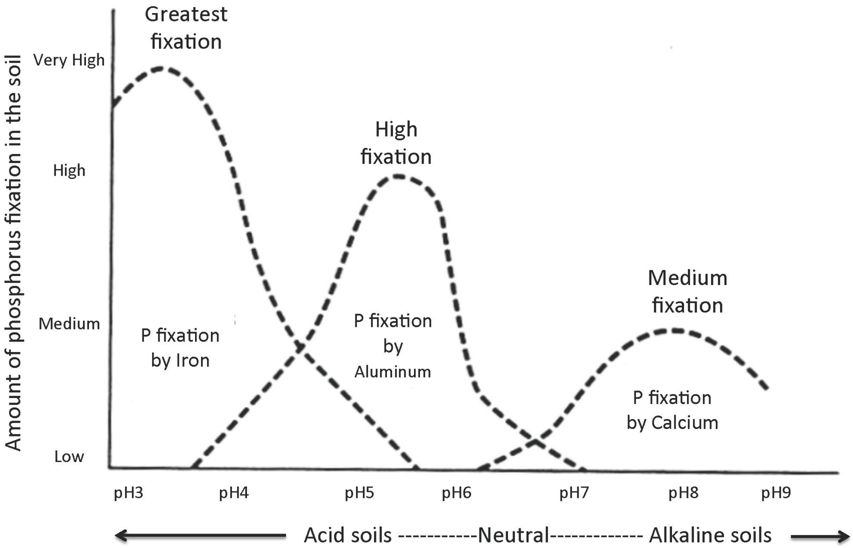

Figure 1-2 Phosphorus fixation at different pH values (Stevenson and Cole, 1999)

The P concentration is also influenced by sorption to minerals and organic matter. Ionic P ($H_2PO_4^-$ and HPO_4^{2-}) is removed from the soil solution by adsorption. A chemical bond is formed with clay, oxides or hydroxides of Fe and Al, calcium carbonates and organic matter. After this reaction the adsorbed P still is accessible for plant uptake, although the amount that is readily available is reduced. Desorption is the reversed process of adsorption. Soils differ significantly in their capacity to adsorb and desorb P. The capacity depends on their clay content, the presence of Fe/Al oxides, pH and the amount of P in the soil and also the way it is cultivated (Nziguheba et al., 1998). Precipitation and its opposite process dissolution are reactions between P and Ca, Al, and Fe. It differs from adsorption because the products of this reaction are less available for plant uptake than the production of adsorption. In soils with a high pH, calcium phosphates are formed, which can be converted to apatite in the long term.

The second factor that influences the P concentration in the soil solution is the ability of soil to replace the phosphate ions when removed by crop, and depends on the soil's buffer capacity. Crop uptake is determined by the effective root length and efficiency of P uptake by roots, which is co-determined by other growth factors. Due to the slow diffusion rates of P in soil, which lead to depletion zones around the root, many plants form a symbiosis with fungi called mycorrhiza. This symbiosis helps in P uptake as the fungal hyphae can function like a root (Hinsinger and Donald, 1998; Syers et al., 2008), causing about 10-fold increase in the effective root surface area and a

2-3 fold increase in the P uptake per unit root length compared to non-mycorrhizal plants (McNear Jr., 2013). Thus with increase of the contact between roots and soil, capacity of the plant for nutrient uptake is improved.

The efficiency of P uptake differs between plant species and genotypes. The composition of soil and its structure influence the buffer capacity (Hinsinger and Donald, 1998; Syers et al., 2008).

As noted above, P is found in soils both in organic and inorganic forms. The proportion of these two forms differs from soil to soil and depends on the history of management (fertilizer and manure application, type of crop, erosion soil losses, etc.). Frequently more than half of the P is in the organic form. The minerals are releasing more soluble P forms when they weather as a result of soil development in time, but this differs from soil to soil. Weathering is the breaking down of minerals, by physical and chemical processes. So, in weathered soils the calcium and other basic minerals, which bind the phosphates, are broken down and finally leached from the soil. This causes a decrease in pH of the soil solution, and enhances fixation of P with Fe and Al (Hoffland et al., 2011; Syers et al., 2008).

Fertilizer P applied to the soil has a direct effect on the available soil P. Generally, crops can take up 10-20% of the applied fertilizer P in a particular growing season, while a substantial amount of applied P accumulates in the soil (Syers et al., 2008; Wolf et al., 1987). Fertilizers may have effects beyond the applied season, which is called the residual value or residual effect of P in soil (Barrow and Campbell, 1972; Bolland et al., 1989). The residual P is the difference between P inputs (mineral fertilizer, manure, weathering and deposition) and P outputs (withdrawal of P in harvested products, and P loss by runoff or erosion) (Bouwman et al., 2009). Plant can use this source of P for many subsequent years that should be considered in substantial application of fertilizers. Low P recovery rates are related to soil properties. The soil type influences how easily the phosphate is converted in more insoluble forms.

Another process influencing P availability is mineralization. It is defined as the decomposition of organically bound nutrients into inorganic forms by microorganisms; the inorganic forms are released in the soil solution. Organic P comes from crop residues or organic manure. Immobilization is the conversion of inorganic forms to organic forms; this occurs when soil microorganisms absorb the soil P to use it as a nutrient (Brady and Weil, 2008; Hoffland et al., 2011; Stewart and Tiessen, 1987). Factors like temperature and moisture control the activity and growth of microorganisms. Therefore, these factors also control the balance between mineralization and immobilization (Tate, 1984).

1.4. Objectives and content

The potential impacts of limited P on global food security have been studied mostly based on consideration of the available P resources. However, the importance of sustainable P application at the global scale to mitigate the P global problem has not yet been highlighted. In my research I emphasize the sustainable consumption of P rather than the total available P sources. My overall goal is to assess the required P fertilizer in future global food production scenarios in different world regions and the entire globe in 2050. In my analysis, residual P and its role in sustainable P management is essential. The focus of my research is illustrated in Figure 1-3 showing the pools and flows of P in croplands and grasslands and transfer of P between these two systems. I pursued the following objectives to understand sustainable consumption of P in agriculture:

1. To assess the long-term P application through chemical P fertilizer and manure, historical P withdrawal by harvested crops and grazed grass and P transfer between cropland and grassland using long-term data for different world regions including Africa, Asia, Eastern Europe, Latin America, North America, Oceania, Western Europe and for the entire globe.

2. To calculate future global P fertilizer demand for food production in different world regions. This requires a model-based approach. Food production consists of different functional group of crops including cereals, legumes, vegetables, fruits or grasslands for feeding the animals. In contrast to the large number of existing models that simulate nitrogen and carbon dynamics, there are not many models to simulate the effect of soil P dynamics on agricultural production systems at global scale. Hence, an important objective of the work was to identify a suitable model for assessing global P requirements for food production.

3. To estimate the amount of P that can be saved through a more sustainable use of P. The key feature of this objective is to consider the contribution of residual P to the available P for plant uptake, now and in future.

This study is based on three major components, i.e. data analysis, model selection and development and finally scenario analysis for future food requirement trajectories. These three components were required to a) perform a sound analysis of the impact of different rates of P application on agricultural production and consequently food production, b) develop reliable scenarios for assessment of future food P demand, and c) optimize P use efficiency through development of new management strategies.

These objectives have been investigated in six chapters. In Chapter 2, I present a generalization of the QUEFTS model for different climate and soil environments. I show the challenges that may be encountered when using this model at global scale. Partly based on findings of Chapter 2 an alternative model for analysis of crop soil modeling for P applications is proposed in Chapter 3. Chapter 3 covers the long-term effect of residual soil P on crop production in arable land and its contribution to global food production. The same type of analysis is performed in Chapter 4 for grasslands. Based on findings of Chapters 3 and 4, I identify the crucial role of China as a major consumer and producer of P fertilizer in the world in Chapter 5. This chapter clearly shows how potentially effective sustainable P application policies at global scale can be. I highlight the fact that sustainable P application is achievable at global scale if policy makers improve legislations for fertilizer application, and among all countries, China can play a key role. Finally the thesis is wrapped up in Chapter 6, where concluding remarks and discussions are provided.

This research is the first study that estimates the global scale P demand in 2050, not only for croplands but also for grasslands based on a physical-based model. The analyses are supported by comprehensive analysis of long-term P application crop production data and quantitative analysis is provided of sustainable P application for feeding the world in 2050. To achieve these targets for estimation of future trends of P consumption, a proper model for crop-soil simulation and valid scenarios for future population growth are needed. These two major issues are discussed in the following sections and will be referred in other chapters of this thesis.

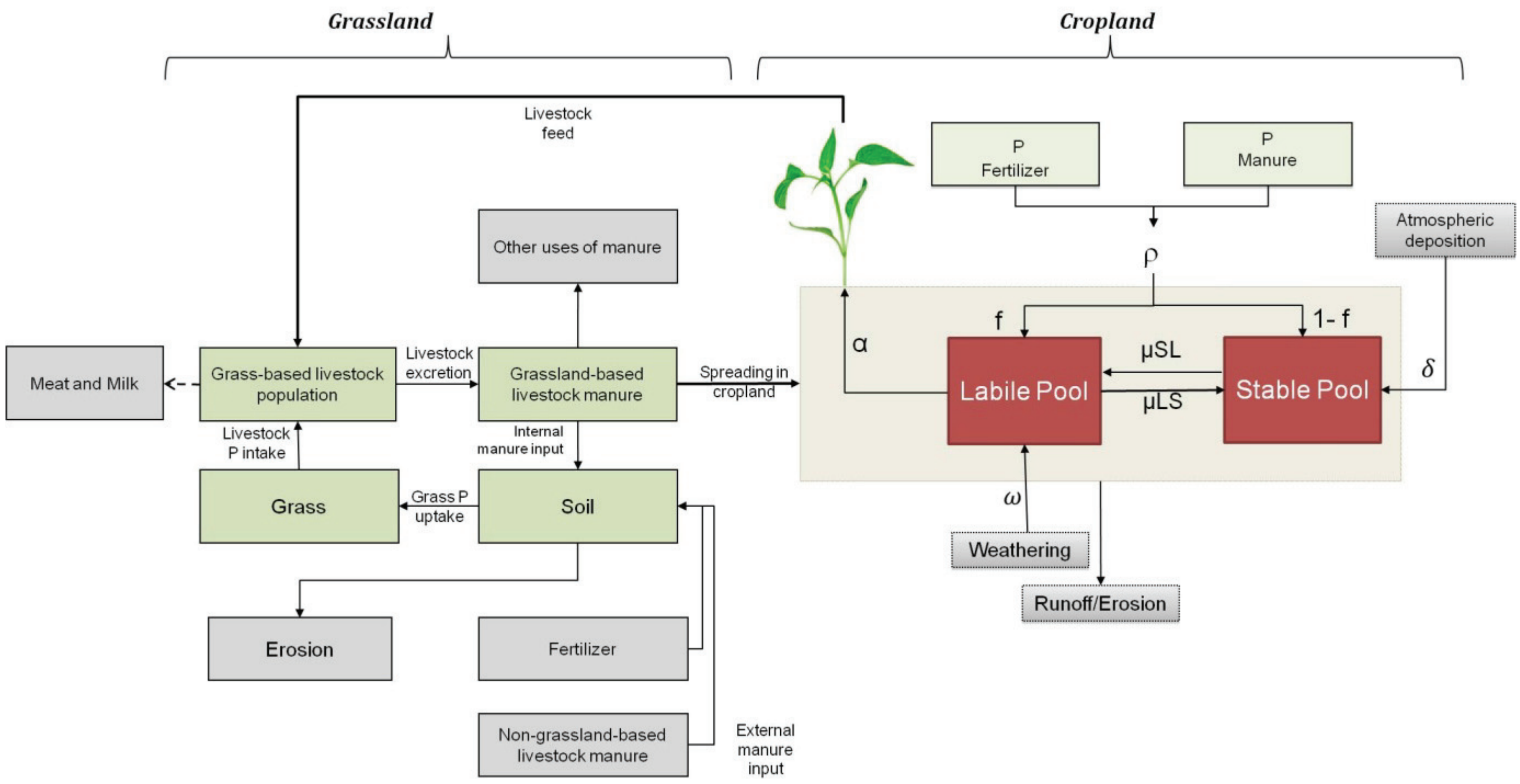

Figure 1-3 Flows and pools of P in croplands and grasslands as studied in Chapters 3, 4 and 5.

1.5. Modelling phosphorus cycling in soil-plant systems

Various models have been developed and described in the literature. Modelling of the dynamics of P in soil-plant systems in agricultural fields is an important component of this thesis. Table 1-2 presents an overview on major P-crop models including other additional information. A brief explanation of each model is also given in this chapter. These models can be classified on the basis of various criteria. Here, available P models are grouped based on three criteria:

a) **Input data requirement:** depending on the structure of the models (e.g. number of pools, components, etc.) and the modelling approach, models have different levels of required input data.

b) **Degree of complexity:** it includes different aspects such as dynamics of model, number of modeled processes and details of processes. Thus application of complex models requires several parameters to be measured or calibrated.

c) **Adaptability and application scale:** some models are specific to some climate or physico-chemical conditions for a given scale. An important quality is the adaptability of models to different conditions and different scales.

WOFOST

WOrld FOod STudies (WOFOST) is a mechanistic, and site-specific crop growth model (Boogard at al. 1998). The model has evolved over the past years and has been applied for different purposes (Van Ittersum et al., 2003 and references there in). WOFOST simulates the crop growth and production for given weather conditions (light and moisture), soil macronutrient data (N, P, K) based on the QUEFTS model (Janssen et al., 1990), crop characteristics and crop management. WOFOST can evaluate, for instance, effects of climate or climate changes to predict crop production and yield variability.

The required input data for climate, soil profile, soil physics, soil fertility and crop data include: rainfall, temperature, wind speed, global radiation, air humidity, soil moisture content, and data on rooting depth and saturated and unsaturated water flow. Data on crop management are also needed. WOFOST has a daily time step to calculate crop growth and soil water balance, and the nutrient uptake is calculated per growing season. The model was developed initially for tropical agriculture, but in principle it is applicable anywhere (Van Diepen et al., 1989).

PHOSMOD

PHOSMOD is a mechanistic model, developed for describing the effect of soil indigenous and fertilizer P on crop growth and P uptake in arable cropping systems (Greenwood et al., 2001a). This model can calculate the maximum amount of P that can diffuse to each root segment through the soil. The PHOSMOD model can simulate plant growth and nutrient uptake on a daily basis. The model was tested for a field with vegetables grown on a sandy loam soil (Greenwood et al., 2001b) as well as in a field fertilizer trial of spring barley under different soil and weather conditions in Norway (Kristoffersen et al., 2006). An adjusted model was able to predict the different responses to applied fertilizer in different soil types and the amount of P in plant dry matter. It was claimed that PHOSMOD can predict the effect of different P fertilization strategies and might be useful to improve fertilizer-planning programs, but it has been applied and tested in only a limited number of regions.

APSIM

The Agriculture Production system SIMulator (APSIM) is a modular modeling framework that simulates crop production in relation to climate, genotype, and soil and management issues in farming systems (Keating et al., 2003). APSIM is applicable to a diverse range of crops, pastures and trees. Soil processes have been considered in APSIM and modules for soil water, nitrogen, phosphorus, pH, soil erosion and a full range of management controls are included in APSIM. The SOILP and MANURE models have been implemented for describing the transformation of P in soil and handling the release of N and P from manure, respectively.

CENTURY/ DAYCENT

The first broadly used soil organic matter (SOM) model was developed by Jenkinson and Rayner (1977). This conceptual framework was used in the CENTURY model. The CENTURY, soil organic matter model was developed to simulate SOM dynamics in natural grassland in the North American Great Plain (Parton et al., 1987). Parton et al. (1987) added the impact of soil texture on SOM dynamics and developed detailed nutrient cycling sub-models that simultaneously simulate soil C and N, P and S dynamics (Parton et al., 1994; Parton et al., 1988). Parton et al. (1988) modified a P cycle model (e.g. Cole et al., 1977) to simulate P flows with a monthly time step. These flows include weathering, formation and solubilization of P, P fertilizer, P removal by plant grain and residues and erosion. Transformations of P in the CENTURY model are based on empirical relations and developed for N limiting soils (Parton et al., 1988). In N-limited soils, P dynamics has a minor effect on plan production level and rate of organic matter decomposition (Gijsman et al., 1996).

Gijsman et al. (1996) tried to apply this model to simulate C, N, P dynamics on highly weathered low-P soils in native savanna grassland. But the model behaviour was unreliable and the output of model was very sensitive to the input parameters. Gijsman et al. (1996) concluded that the CENTURY model - developed for temperate soils - has limited applicability for those tropical soils in which P is the major limiting nutrient relative to N.

The DAYCENT model, a terrestrial ecosystem model, is daily time step version of the CENTURY model. Most soil processes operate on a daily scale, while plant growth is simulated weekly.

DSSAT /Phosphorus module

Daroub et al. (2003) have developed a model to simulate P dynamics and P uptake by crops adapted to highly weathered acidic and alkaline calcareous soils. The soil P module was linked with two crop models within the DSSAT software, i.e. CERES and CROPGRO (Jones et al., 2003). DSSAT includes crop simulation models, databases for weather, soils, and crops. This package includes models of 16 different crops. The new DSSAT cropping system model (CSM) has one soil module, a Crop Template module, which can simulate different crops, a weather module, and a competition module for light and water among the soil, plants, and atmosphere. The P module developed by Daroub et al. (2003) comprises organic and inorganic P pools based on a daily time step. In addition to general data sets needed to run the general model, the rate, date and type of P fertilizer applied, and the initial value of soil P in each pool are needed. The authors stated that additional testing is needed to make sure the model is more adaptable to a wide variety of soils, weather, and crop conditions (Daroub et al., 2003).

FIELD

FIELD (Field-scale Interactions, use Efficiencies, and Long-term soil fertility Development) is a simplified crop-soil model (Tittonell et al., 2010). This model works based on the availability of light, water, nitrogen, phosphorus and potassium and their interactions. Simulation of crop production has been done using the general concept that crop production is equal to resource availability multiplied with resource use efficiency. In this model light determined yield and water limited yield are calculated first and the minimum value is used for calculating nutrient limited yield. The model uses seasonal time steps for simulating crop productivity and changes in nutrient stocks at field scale. If data are available for soil mineral nutrient parameterization, the supply of N and P follow C:N and C:P ratios of different C pools. Otherwise, empirical functions used in the QUEFTS model (Janssen et al., 1990) are employed to estimate soil N, P and K supply from soil analytical data. The model was calibrated and tested against experimental data for major crops grown in smallholder systems of sub-Saharan Africa (Tittonell et al., 2010).

Ecosys

A mathematical model was developed by Grand and Robertson (1997) within the ecosystem simulation model called Ecosys. Transformation and transport of inorganic and organic P were coupled to root growth and nutrient uptake models. The model avoids the use of site-specific parameters and is independent of P uptake data. So, Grand and Robertson (1997) have claimed that it could be possible to simulate P uptake under different soil and climate conditions given relevant soil properties. They tested seasonal estimates of soluble P concentration, root growth and P uptake from the combined models with data measured from barley under fertilized and unfertilized treatments in a long term P fertilizer experiment (Grant and Robertson, 1997).

EPIC

The Erosion-Productivity Impact Calculator (EPIC) model (Williams et al., 1983) was developed to simulate wind and water erosion and their effects on crop productivity in the United States. A simplified soil and plant P model to simulate N and P fertility (Jones et al., 1984) is embedded in the EPIC model. Since EPIC was originally developed for erosion simulation, it can model long-term processes such as weathering and N, P removal by erosion.

The P sub-model of EPIC comprises stable, active and labile inorganic P, fresh organic and stable organic P, grain, stover and root P. The model has a daily time step and simulates P uptake and transformation in up to 10 soil layers with variable depth. Crop P uptake is sensitive to crop P demand and amount of P labile, soil water, and roots in soil layer. The required soil data in the EPIC include soil chemical, physical and taxonomic data (Jones et al., 1985).

PARJIB

PARJIB was developed by Reid (2002) to describe the yield response to nutrient supply. This model simulates potential and maximum yield adjusted for plant population and water stress. Crops responses to N, P, and K are simulated separately, and then individual N, P and K yields are combined to calculate the final yield. The model can be calibrated against field measurements of crop performance. Calibration requires information on maximum or potential yield, fertilizer N, P, and K and irrigation applications, potential evapotranspiration, and soil information. Soil information requirement comprises estimates of available water capacity and soil chemical analyses such as N, Olsen P and exchangeable K. Other data inputs include soil exchangeable Mg, fertilizer Mg, plant population, soil pH, and bulk density.

Wolf et al. (1987) model

A simple model was designed to simulate the long-term response of crop production to P fertilizer by Wolf et al. (1987). In this model, it was assumed that the crop yields are limited only by P supply. The model includes both labile and stable pools of P with a yearly time step. The model can calculate P transfer between the two pools, the P uptake by crops, and the pool sizes. Input data to run the model include rate and type of applied fertilizer, total P uptake by fertilized and unfertilized crops during the first year of applying fertilizer, net input of P, and the time constant of transfer between labile and stable pools.

The model can be used for calculating the fertilizer requirement for a target yield. Wolf et al. (1987) mentioned that the time constant of P transfers between labile and stable pools can be estimated from field experiments. The results of this model have been compared with long-term result of field trials with maize, sorghum, and rice in Brazil, Australia, and Madagascar, respectively. Hereafter we refer to this model as the DPPS (Dynamic Phosphorus Pool Simulator) model.

QUEFTS

Quantitative Evaluation of Fertility of Tropical Soils "QUEFTS" was applied in Kenya and Surinam for studying the relationship between grain yield and nutrient supply. The model was originally used for calculating tropical maize yield as a function of N, P and K, which are available from soil and added fertilizer (Janssen et al., 1990). The model consists of four steps. First, the QUEFTS estimates the potential N, P, and K supply, based on organic N and C content, Olsen-P content, exchangeable K, and pH (H_2O) as independent variables. In the second step, the actual uptake of each nutrient is calculated as a function of the potential supply of two other nutrients. In the third step three yield ranges are calculated in terms of the actual uptakes of N, P, and K. In last step these yield ranges are combined in pairs, and the yields estimated for pairs of nutrients are averaged to obtain an ultimate yield estimate.

The QUEFTS model was successfully implemented for calculating required fertilizer for different crops such as maize, rice and wheat in different tropical and subtropical soils. QUEFTS was calibrated by Smaling and Janssen (1993) in Kenya for maize, Witt et al. (1999) for rice in tropical and subtropical Asian countries, Pathak et al. (2003) for wheat in tropical and subtropical regions of India, Haefele et al. (2003) for rice in West Africa, Liu et al. (2006) for wheat and maize in China and Das et al. (2009) in Eastern India for rice.

The listed models have been developed for different processes in crop-phosphorus interactions. However, regarding the complexities, applications of the models, objectives and scale of this study, not all of these models were applicable for my purposes. A more detailed discussion of different aspects of the model is presented below.

A) Input data requirement

DSSAT/P-module, APSIM, and the CENTURY models require a significant amount of data as inputs for P module and also data on phenology and a few more parameters for crop growth such as light interception and photosynthesis. The structure of the EPIC model is complex with significant data requirements. QUEFTS, DPPS and PHOSMOD are relatively simple models. In the latter two models P has been assumed as the only limiting nutrient, and N and K are considered to be sufficient for maximum growth, while QUEFTS requires NPK input data. The required data for the QUEFTS model can be collected in a relatively standard soil survey (Smaling and Janssen, 1993). DPPS only needs the amount of P application, which facilitates its application.

Among the reviewed models the limited data requirements of QUEFTS and DPPS is an obvious advantage of these models.

B) Complexity of models

The reviewed models can be divided into mechanistic and semi-mechanistic models. Usually mechanistic models are more complex than the semi-mechanistic models as they may simulate different processes. For application of these models many parameters should be measured or calibrated. Among different models, QUEFTS and DPPS have fewer calibration parameters that make them more feasible to apply at global scale.

QUEFTS combines empirical and theoretical approaches and has in total 12 parameters to be calibrated. In soil P models generally different pools are distinguished. DPPS distinguishes two soil P pools (labile, stable); this is because increasing the number of pools from two (labile, stable) to three (active, labile and fixed) was shown not to improve model predictions, and makes calibration more difficult (Bhogal et al., 1995).

C) Adaptability

Some of the reviewed models have been applied and calibrated only for specific locations (Table 1-2). For example, the CENTURY model that was originally developed for temperate soils has less applicability for P-limited soils in tropics. Or, the FIELD model was developed for field scale, and has been applied only in sub-Saharan Africa. PARJIB has been tested in a small number of case studies and there is no significant verification record in the literature. The QUEFTS model, as a semi-mechanistic tool, has been applied and calibrated for three major food crops (maize, wheat, and rice) in different parts of the world for tropical and subtropical conditions. The QUEFTS model simultaneously works with three major nutrients (N, P, K) and can calculate the final yield in terms of these three nutrients. DPPS has been tested for two staple foods: maize and rice as well as sorghum in South America (Brazil), Africa (Madagascar) and Australia.

Table 1-2 An overview of crop-phosphorus models

Model	Objective	Applied scale		Crop type
		Temporal	Spatial scale	
WOFOST(Van Diepen et al. (1989))	Crop growth under water limited, NPK limited	Daily; One growing season	Originally for tropics	Temperate and tropical crops
PHOSMOD (Greenwood et al. (2001a) ; (2001b))	Effect of soil P and P fertilizer on crop growth, total P uptake	Daily	Norway	Vegetable crops (Spring Barley)
PARJIB (Reid (2002))	Yield response to nutrient supply, adjusted for crop population and water stress	Seasonal	New Zealand	Maize, sweet corn, tomato and carrot
DPPS (Wolf et al. (1987))	P Uptake, transfer P between P pools, fertilizer requirement for a target yield	Yearly	Brazil, Australia, Madagascar	Maize, Sorghum, Rice
APSIM (Keating et al. (2003))	Water balance, N and P transformations, soil pH, erosion and a full range of Management controls	Daily	Australia, Zimbabwe, Netherlands, Kenya and Colombia	Crops, Pastures and trees
CENTURY (Parton et al. (1988))	Dynamics of soil organic matter –C, N, P, S dynamics	Monthly	Temperate, Tropics	Grasslands
DAYCENT (Parton et al. (1998))	C, N, P, S dynamics (Generalized model for N_2 ad N_2O)	Daily	Temperate, Tropics and global	
DSSAT (P-module)(Daroub et al. (2003))	P dynamics and P uptake by crops	Daily	Colombia, Syria, and Tanzania	Wheat, Sorghum, Bean, maize

Table 1-2 Continued.

Model	Objective	Applied scale		Crop type
		Temporal	Spatial scale	
QUEFTS (Janssen et al. (1990))	N, P, K	Seasonal	Topics and subtropics	Maize, Wheat, Rice
FIELD (Tittonell et al. (2010))	Crop productivity, soil nutrient stocks	Seasonal	Sub-Saharan Africa (field scale)	Maize, Napier, Sweet potato
Ecosys (P-module)(Grant and Robertson (1997))	P uptake in different soil and climate (Coupled to a root and mycorhizal growth model)	Seasonal	Canada	Barley
EPIC (Sub model)(Jones et al. (1985) ,(1984))	Org. C, total N, Org. P, crop demand and uptake of N, P	Daily	United State	Maize, Wheat

Most models discussed above require comprehensive input data sets that are not available at the larger scale. Furthermore, if the data are available, a complex calibration procedure is required for prospective application of the models. Among the reviewed models QUEFTS and DPPS seem to be potentially suitable for application in this study. In case of scarcity of data on NPK application, DPPS would be the preferred alternative due to very limited input data needed.

1.6. Scenarios for future global P requirement

The future is very uncertain. To explore possibilities for the future that cannot be predicted by extrapolation of past and current trends, scenario development is a proper option (Cork et al., 2005). According to Nakicenovic et al. (2000) a "*Scenario* is a comprehensive and plausible description of the future of the human-environment system, including a narrative with qualitative trends and quantitative projections relevant to development patterns".

To address Objectives 2 and 3, scenarios can be used to look at future P requirements but also to assess how P use can be made more efficient. This requires an integrated approach that combines land use, livestock and crop production together with the P models selected in the previous section for simulating crop-fertilizer response at the global scale.

There are different global scenarios, such as from the Organization for Economic Cooperation and Development (OECD) (OECD, 2012), Millennium Ecosystem Assessment (Alcamo et al., 2006), IPCC-SRES (Nakicenovic et al., 2000), and Rio+20 (van Vuuren et al., 2012). In this thesis, we need scenarios that focus on nutrients. This means that we need scenarios that include future food demand, production, and the nutrient uptake at scales that match the requirements of the selected model for this study. Particularly future production and consequently P uptake is needed as a target for selecting the appropriate scenarios. A few recent examples of scenarios that include projection of nutrient uptake are the Millennium Ecosystem Assessment (MEA) (Alcamo et al., 2006) and more recently the Rio+20 (van Vuuren et al., 2012) scenarios.

Future demand of P fertilizer in global croplands in Chapter 3 was calculated based on the target crop yields, given by four different MEA scenarios for 2050. Four MEA scenarios, i.e., Global Orchestration (GO), Order from Strength (OS), Techno-garden (TG) and Adapting Mosaic (AM)(Cork et al., 2005), for 2000 to 2050 describe contrasting future developments in agricultural land use under changing climate (Table 1-3) (Bouwman et al., 2009). The MEA scenarios differ in the projections for population, economic development, energy demand and production, industry and food, feed and biofuel demand and production, attitude towards environmental problems, globalization versus emphasis on regional collaboration and local solutions to environmental problems. These give a good basis to construct a range of scenarios for global P use and depletion. In the different world regions as mentioned in Objective 1, I used the Global Orchestration scenario (GO) for the target crop production and P uptake in 2050. This scenario predicts the highest increase in global crop production among the four MEA scenarios.

Recently, van Vuuren et al. (2012) developed new pathways (Rio+20) to achieve global sustainability goals for food, land and biodiversity, as well as for energy and climate by 2050. The Rio+20 study describes four scenarios, i.e., the Trend scenario and three challenge pathways. The Trend scenario describes possible trends in the absence of climate and sustainability policies. The three challenge pathways were designed to assess the potential to achieve sustainability goals.

In Chapters 4 and 5 I used the Rio+20 Trend scenario as a baseline scenario for simulations of future P demand in the global grassland analysis and China's croplands. Baseline scenarios represent a continuation of current trends, with no dramatic changes or shifts in production and management systems, attitude towards environmental problems, etc. The Rio+20 Trend scenario is a baseline or business-as-usual scenario, and comparable with the baseline scenario of the Environmental Outlook of the Organization for Economic Cooperation and Development (OECD) (OECD, 2012), the Global Orchestration scenario of the Millennium Ecosystem Assessment (Alcamo et al., 2006) and the A1 scenario of IPCC-SRES (Nakicenovic et al., 2000). These scenarios have similar assumptions on population growth and economic development pathways.

Table 1-3 Description of four Millennium Ecosystem Assessment scenarios (adapted from (Bouwman et al., 2009))

	Global Orchestration (GO)	**Order from Strength (OS)**	**Technogarden (TG)**	**Adapting Mosaic (AM)**
Brief description				
	Globalization, economic development, reactive approach to environmental problems	Regionalization, fragmentation security, reactive approach to environmental problems	Globalization, environmental technology, proactive approach to environmental problems	Regionalization, local ecological management with simple technology, proactive approach to environmental problems
General				
World population (billion)	Low	High	Medium	High
	2000: 6.1	2000: 6.1	2000: 6.1	2000: 6.1
	2050: 8.2	2050: 9.7	2050: 8.9	2050: 9.6
Income	High	Low	High	Medium
Global GHG emissions	High	High	Low	Medium

Table 1-3 Continued.

	Global Orchestration (GO)	**Order from Strength (OS)**	**Technogarden (TG)**	**Adapting Mosaic (AM)**
Global mean temperature increase	High	High	Low	Medium
Per capita food consumption	High, high meat	Low	High, low meat	Low, low meat
Agricultural trends				
Productivity increase	High	Low	Medium-high	Medium
Biofuels	4% of cropland area in 2050	1% of cropland area in 2050	28% of cropland area in 2050	2% of cropland area in 2050
Fertilizer use and efficiency	No change in countries with a surplus; rapid increase in N and P fertilizer use in countries with soil nutrient depletion (deficit)	No change in countries with a surplus; slow increase in N and P fertilizer use in countries with soil nutrient depletion (deficit)	Rapid increase in countries with a surplus; rapid increase in N and P fertilizer use in countries with soil nutrient depletion (deficit)	Moderate increase in countries with a surplus; slow increase in N and P fertilizer use in countries with soil nutrient depletion (deficit); better integration of animal manure and re-cycling of human N and P from households with improved sanitation but lacking a sewage connection.

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2

Crop yield response to soil fertility and N, P, K inputs in different environments: testing and improving the QUEFTS model

This chapter has been published as:
Sattari, S. Z., van Ittersum, M. K., Bouwman, A. F., Smit, A. L., and Janssen, B. H. (2014). Crop yield response to soil fertility and N, P, K inputs in different environments: Testing and improving the QUEFTS model. Field Crops Research 157, 35-46.

Global food production strongly depends on availability of nutrients. Assessment of future global phosphorus (P) fertilizer demand in interaction with nitrogen (N) and potassium (K) fertilizers under different levels of food demand requires a model-based approach. In this paper we tested use of the QUEFTS model (Quantitative Evaluation of Fertility of Tropical Soils) for assessing crop yields in response to N, P and K application in different environments. QUEFTS was initially developed to simulate interactions between N, P and K for tropical soils under maize crop. We performed an extensive model analysis of crop yields in relation to soil and fertilizer nutrients for six field data sets with maize, rice, and wheat crops grown in tropical and temperate regions. The model equations had to be adapted to broaden the model applicability beyond the original boundary conditions of pH, rain-fed cropping systems, optimum harvest index and temperature. Recalibration and modification resulted in a good agreement between model predicted and observed yields. Our results indicate that the adjustments increased the applicability of the model. However, for application in global studies QUEFTS is data demanding and, also, further testing (and probably improvement) is needed, since various processes (e.g. inputs of other nutrients than N, P and K, sub-soil properties and water supply) are ignored in the model, but may differ dramatically across the globe.

2.1. Introduction

Global food demand is on a trajectory to increase 70% by 2050. Increased food demand and a truly sustainable agriculture must be met by natural resources such as water, land and nutrient resources (Brown, 2003; Koning et al., 2008). Besides water and land, three key nutrients Nitrogen (N), Phosphorus (P) and potassium (K) are playing a major role in global food production. Of these, phosphorus (P) is an important nutrient due to its low recovery rate and finite availability (Smil, 2000; Steen, 1998). When addressing the need for plant nutrients with regard to critical global food problems, it is essential to know the quantities of nutrients required for target crop production and the supply of nutrients by the soils used for crop growth.

Exploring the consequences of possible future shortage of nutrient inputs and their interactions can be done with model-based approaches. Crop production models can be characterized as empirical and mechanistic (process-oriented) models. Empirical models directly employ a relationship between model variables and model outputs without description of fundamental (physical) processes. These models usually are site-specific (Smaling and Janssen, 1993). Mechanistic models are often more complex as they describe known physical and biological processes in crop and soil. In contrast to the large number of existing models that simulate N and carbon (C) dynamics, there are not many models to simulate the effect of soil P and K in interaction with N on cropping systems in different soil types.

Yield responses to NPK fertilizer applications in relation to soil fertility have been studied in a model, Quantitative Evaluation of the Fertility of Tropical Soils (QUEFTS), which is based partly on empirical, and partly on theoretical relationships (Janssen et al., 1990). The model needs minimum input data and has a relatively small number of calibration parameters. In addition, it has the possibility to assess NPK limitations in relation to the target yield.

QUEFTS has been applied in a variety of countries, for different sites with a range of soil types, and crops in tropical areas, but it has rarely been tested in temperate regions.

Witt et al. (1999) applied the QUEFTS model for irrigated lowland rice between 1994 and 1997 from 15 tropical and subtropical sites in six Asian countries. Given 381 observations, Haefele and Wopereis (2005) verified the QUEFTS parameters determined for rice by Witt et al.1999 for irrigated rice in Sahelian West Africa. They proposed that the model could be used to develop strategies to provide sufficient fertilizers in different agro-ecological zones (Haefele and Wopereis, 2005). Furthermore, QUEFTS was calibrated by Das et al. (2009) for site-specific, balanced fertilizers of 20 sites in Eastern India for rice. QUEFTS has been used also to estimate the nutrient requirements for target wheat yields (Liu et al., 2006; Pathak et al., 2003).

The overall objective of this research is to test the suitability of QUEFTS as a generic model with global applicability, and to improve the model when needed and possible. As a result of extensive data collection, six sets of experimental data on different crops and different environments across the world were obtained from supportive colleagues. Based on these datasets, we performed a systematic analysis of the various model components to identify major challenges for its performance in different circumstances, to explain possible differences between simulated and measured yields, and to extend the applicability of the model at global scale.

2.2. Materials and methods

2.2.1. QUEFTS – model description

QUEFTS was designed as a tool for land evaluation by combining the assessment of tropical soil fertility with the assessment of fertilizer requirements. The model was originally developed for maize under rain-fed conditions. However, it can be modified for other crops as long as the relationship between nutrient supply and yield is known (Janssen et al., 1990).

In the QUEFTS model it is assumed that N, P and K are the growth-limiting factors, i.e. these are more limiting than other factors such as water availability, limited root penetration and poor crop husbandry practices. Soils should be deep and well drained. The values of the diagnostic soil properties should lie within the ranges for which QUEFTS was tested: pH (H_2O) between 4.5 and 7.0; organic C less than 70 g kg^{-1}; P-Olsen less than 30 mg kg^{-1}; exchangeable K less than 30 mmol kg^{-1}.

The modeling procedure in the QUEFTS model consists of four steps: supply of available nutrients, actual nutrient uptake, yield ranges, and yield estimate.

2.2.1.1. Step 1. Assessment of the amounts of available nutrients

Available nutrients may be supplied by soil and by inputs. QUEFTS uses purely empirical, linear as well as nonlinear multiple regression equations to estimate the potential soil supplies of available N, P, and K, based on organic C content, Olsen-P, exchangeable K (K_{exch}), and pH (H_2O), and optionally organic N and Total P content, as independent variables. All variables, parameters and their units in the QUEFTS equations are given in Table 2-1. The equations were derived for non-flooded field situations.

For the estimation of the supply of available nutrients by inputs, usually in the form of fertilizers, QUEFTS requires fertilizer recovery fractions for different nutrients. Only a part of the fertilizer nutrients added to the soil is taken up by the crop, other parts may be misplaced by leaching, volatilization and erosion or can accumulate in soil. The maximum fraction of a certain nutrient that is taken up depends on availability of the other nutrients. This maximum fraction of the applied nutrient that enters the crop is considered to be immediately available nutrient, the other fraction of the applied nutrient is not (immediately) available (Janssen, 1999). Default values for maximum recovery fractions of N, P and K in QUEFTS are 0.5, 0.1 and 0.5. In case of availability of appropriate data these values can be calculated in each case.

The "difference method" (Syers et al., 2008) was used for the assessment of the default values in QUEFTS. This method is based on the difference in experimentally assessed nutrient uptakes between crops grown with and without fertilizer, divided by the amount of applied fertilizer to calculate the recovery fraction.

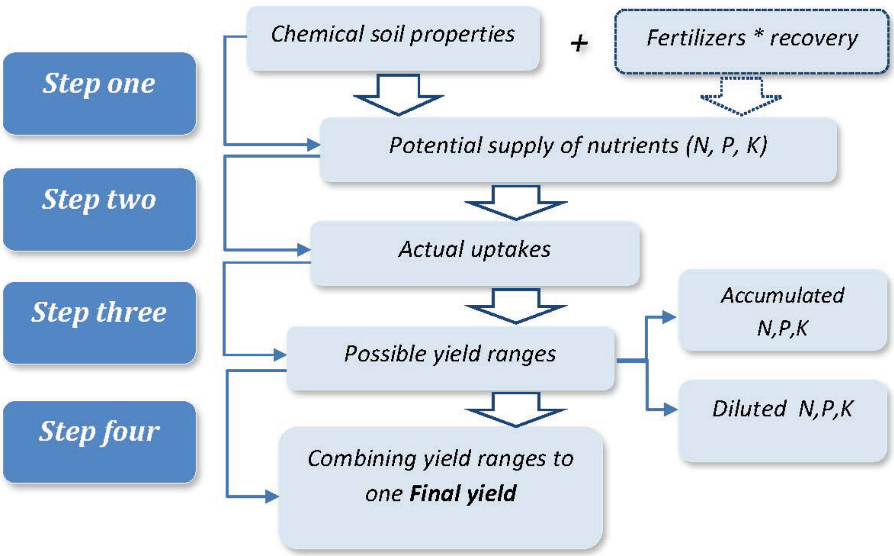

Figure 2-1 Four steps of the simulation procedure in QUEFTS.

A crucial requirement for the assessment of the maximum supply of an available nutrient, from soil as well as from input, is that all other growth factors, including the availability of the other nutrients than the one under study, are at optimum level. For that assessment the following relations are used:

$$S_N = \alpha_N f_N C_{org} + I_N R_N \tag{2-1}$$

$$S_P = \alpha_P f_P C_{org} + \beta_P P_{Olsen} + I_P R_P \tag{2-2}$$

$$S_K = (\alpha_K f_K K_{exch})/(\gamma_K + \beta_K C_{org}) + I_K R_K \tag{2-3}$$

Table 2-1. Abbreviations used in QUEFTS, units and assigned values.

Symbol	**Definition**	**Units**	**Assigned value * or equation**
N_{org}	Soil organic nitrogen	g kg^{-1}	Input data ≤ 7
C_{org}	Soil organic carbon	g kg^{-1}	Input data ≤ 70
P_{Olsen}	Soil phosphorus (Olsen P)	mg kg^{-1}	Input data ≤ 30
P_{Tot}	Total soil phosphorus	mg kg^{-1}	Input data; 25 × C_{Org}
K_{Exch}	Soil exchangeable potassium	mmol kg^{-1}	Input data ≤ 30
pH	pH(water)	-	4.5 -7
I_i	Application of fertilizer *i* (i = N, P, K)	kg ha^{-1}	Input data
R_i	Recovery fraction of nutrient *i* (i = N, P, K)	-	R_n=0.5;R_p=0.1;R_k=0.5
S_i	Potential supply of nutrient *i* (i = N, P, K)	kg ha^{-1}	Eqs. (2-1)-(2-5)
α_i	Parameter (i = N, P, K) in Eqs. (2-1 to 2-3)	-	$\alpha_N = 6.8; \alpha_P = 0.35; \alpha_K = 400$
β_i	Parameter (i = P, K) in Eqs. (2-2-2-3)	-	$\beta_P = 0.5; \beta_k = 0.9$
γ_i	Parameter (i = K) in Eq.2-3		$\gamma_k = 2$
α_{NN}	Parameter in Eq. 2-4	-	$\alpha_{NN} = 68$
q_P	Parameter in Eq. 2-5	-	$q_P = 0.014$
f_i	pH correction factor of nutrient *i* (i = N, P, K)	-	Eqs. (2-6)-(2-8)
U_i	Actual uptake of nutrient *i* (i = N, P, K)	kg ha^{-1}	Eq. (2-9)
a_i	PhE_i at maximum accumulation for nutrient *i,* (i = N, P, K)	kg kg^{-1}	a_N=30;a_P=200;a_K=30
d_i	PhE_i at maximum dilution for nutrient *i,*	kg kg^{-1}	d_N=70;d_F=600;d_K=120
r_i	Minimum nutrient *i* (i = N, P, K) uptake to produce any grain	kg ha^{-1}	r_N=5;r_P=0.4;r_K=2
Y_i^a	Yield with maximum accumulation of nutrient *i* (i = N, P, K)	kg ha^{-1}	Eq. (2-10)
Y_i^d	Yield with maximum dilution of nutrient *i* (i = N, P, K)	kg ha^{-1}	Eq. (2-11)
Y_{ij}	Yield for the pair of nutrients *i, j*	kg ha^{-1}	Eq. (2-12)

Table 2-1 Continued.

Symbol	**Definition**	**Units**	**Assigned value * or equation**
$Y_{\max}$	Potential (grain) yield	kg ha^{-1}	10000 at 12% moisture
Y_U	Ultimate yield	kg ha^{-1}	Eq. (2-13)
DM	Dry matter production	kg ha^{-1}	Eqs. (2-16) and (2-17)
χ_{gi}	Mass fraction of nutrient *i* in grain nutrient *i* (i = N, P, K)	g kg^{-1}	Eqs. (2-16)-(2-18)
χ_{si}	Mass fraction of nutrient *i* in straw nutrient *i* (i = N, P, K)	g kg^{-1}	Eqs. (2-16)-(2-18)
PhE_i	Physiological efficiency	kg kg^{-1}	Eq. (2-18)

* All the values are adopted from the original QUEFTS model (Janssen et al., 1990) for maize. Recalibrated values are presented in Table 2-4.

where S_N, S_P and S_K are supplies of crop-available N, P and K, respectively; α, *β* and γ are empirical parameters; I_N, I_P, and I_K refer to N, P, and K inputs to the system, f_i is a pH dependency coefficient (Eqs. 2-6 to 2-8)and R_N, R_P, and R_K refer to the maximum recovery fraction of each fertilizer.

When data on organic nitrogen are available, the soil supply of available nitrogen can be calculated by Eq. (2-4). The value of α_{NN} is 10 times that of α_N in Eq. (2-1), assuming that the C:N ratio of the organic matter is 10.

$$S_N = \alpha_{NN} f_N N_{org} + I_N R_N \tag{2-4}$$

Eq. (2-5) calculates S_P with Total P (P_{Tot}) is:

$$S_P = q_P f_P P_{tot} + \beta_P P_{Olsen} + I_P R_P \tag{2-5}$$

The default values of α_P in Eq. (2-2) and q_P in Eq. (2-5) are 0.35 and 0.014, and hence their ratio is 25. This suggests that the ratio of P_{tot}/C_{org} is also 25, when P_{Tot} is expressed in mg kg^{-1} and C_{org} in g kg^{-1}. Such a value was found indeed as an average for this ratio in areas where no fertilizer P had been applied (Janssen and Guiking, 1990). Once farmers start to apply inorganic fertilizer P the ratio increases, and then it is recommended to use only Eq. (2-5).

The coefficient f_i (i = N, P, and K) in Eqs. (2-1)-(2-5) is used to describe the pH-dependency of soil organic matter mineralization, P solubility (Figure 2-2) and K exchangeability, as discussed in more detail in the original QUEFTS paper (Janssen et al., 1990).

$$f_N = 0.25(pH - 3) \tag{2-6}$$

$$f_P = 1 - 0.5(pH - 6)^2 \tag{2-7}$$

$$f_K = 0.625(3.4 - 0.4pH) \tag{2-8}$$

When pH (H_2O) is outside the range of 4.6-7.4, f_P (Eq.(2-7)) has a negative value, and when pH (H_2O) is >8.5, f_K is negative (see Section 2.3.2.2). In soils with high pH and high soil organic matter content, the effective cation exchange capacity (CEC) is relatively high, which causes at a given content of exchangeable K a relatively low availability of K to the plant because K saturation of CEC is then relatively low (Mowo et al., 2006).

2.2.1.2. Step 2. Relation between supply of available nutrients and actual uptake

The actual uptake of each nutrient is calculated as a function of the potential supplies of that and the other nutrients. For example, $U_i(j)$ refers to the uptake of i in relation to j. If i = N, j may be P or K. In other words, two values for N uptake can be calculated based on the potential supplies of P and K. The lower uptake value of the two is used for the final estimation of nutrient uptake (U_i).

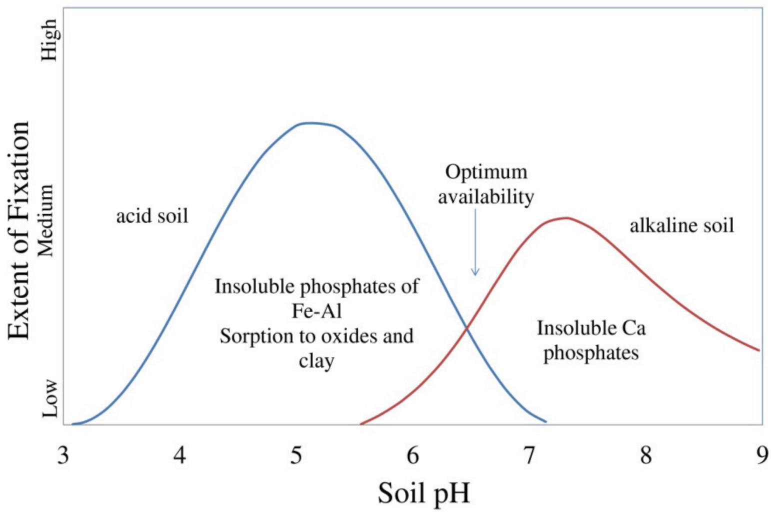

Figure 2-2 Schematic presentation of the effect of pH on fixation of P in two different soil types. Optimum availability occurs at pH equal to 6.5 (Redrawn from Stevenson and Cole (1999)).

The relationship between supply and actual uptake of a nutrient is a theoretical relation consisting of three zones (Figure 2-3). In the initial zone (I), the actual uptake of nutrient *i* (U_i) is equal to and hence changing linearly with the supply of nutrient *i* (S_i). At large values of S_i, further increase of supply does not lead to further increase of nutrient uptake (zone II). The latter situation may occur when the other nutrient is yield limiting. In the intermediate ranges of $S_{i,}$, there is a nonlinear behavior (zone III). The relationship between uptake and supply in zone III is assumed to be parabolic (III) (Janssen et al., 1990). The following equations are used to calculate actual uptake. Subscripts refer to N, P and K. All parameters and symbols are explained in Step 3 and in Table 2-1.

$$U_i(j) = \begin{cases} S_i, & \text{if } S_i < r_i + (S_j - r_j)(a_j/d_i)(\text{I}) \\ r_i + (S_j - r_j)(d_j/a_i), & \text{if } S_i > r_i + (S_j - r_j)(2d_j/a_i - a_j/d_i)(\text{II}) \\ S_i - 0.25\frac{(S_i - r_i - (S_j - r_j)(a_j - d_i))^2}{(S_j - r_j)(d_j/a_i - a_j/d_i)}, & \text{else (III)} \end{cases}$$

$$i, j = N, P, K \quad i \neq j \qquad (2\text{-}9)$$

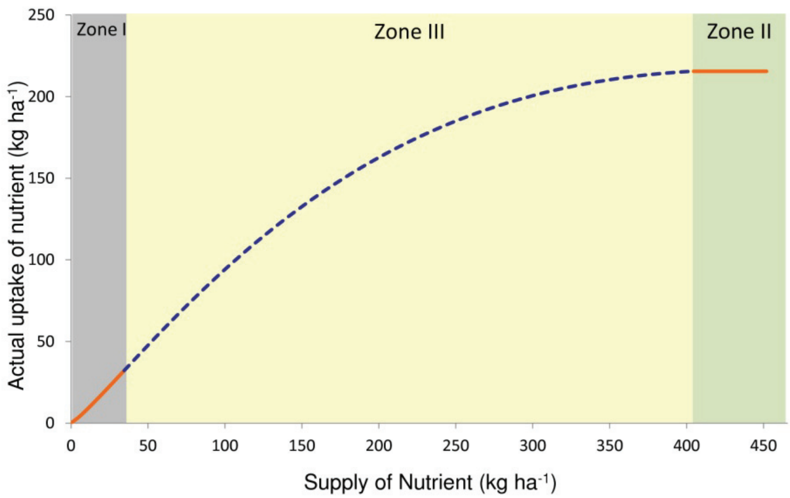

Figure 2-3 Actual uptake versus supply of nutrient *i*, considering three zones I, II, and III.

2.2.1.3. Step 3. Relation between actual uptake and yield ranges

When the uptake of a nutrient is very small compared to that of the other nutrients, maximum dilution of this nutrient occurs within the plant. When other nutrients are strongly limiting, the nutrient under study starts to accumulate in the plant up to a maximum mass fraction. These principles are used in the third step of QUEFTS, where yield ranges are calculated between Y_i^a, yield at maximum accumulation (*a*) and Y_i^d, yield at maximum dilution (*d*), as functions of the actual uptake (U_i) and the minimum uptake required to produce any grain (r_i):

$$Y_i^a = a_i(U_i - r_i), i = N, P, K \tag{2-10}$$

$$Y_i^d = d_i(U_i - r_i), i = N, P, K \tag{2-11}$$

The unit of a and d is kg grain yield per kg uptake, and those of Y_i^d, U_i and r_i are kg ha^{-1}. Eqs. (2-10) and (2-11) are only meaningful when the harvest index (ratio of grain to total above-ground biomass) of maize is about 0.4 to 0.5 (Smaling and Janssen, 1993); if not, it is recommended to apply equations which relate physiological efficiency to harvest index, as will be explained later in Eqs. (2-17) and (2-18). Physiological efficiency (PhE) is the ratio of the yield of the economic plant components (e.g. grains, tubers) to uptake by the whole crop. It is sometimes named conversion efficiency (Chikowo et al., 2010), but more often internal utilization efficiency (Witt et al., 1999).

2.2.1.4. Step 4. Combining yield ranges to ultimate yield estimate

In the fourth step, the yield ranges are combined for pairs of nutrients, and the yields estimated for pairs of nutrients are averaged to obtain an ultimate yield estimate. The following equation is used to calculate Y_{ij}, the yield for the pair of nutrients i and j.

$$\begin{aligned} Y_{ij} = Y_j^a &+ 2\frac{(\min\{Y_j^d, Y_k^d, Y_{max}\} - Y_j^a)(U_i - r_i - (Y_j^a/d_i))}{\min\{Y_j^d, Y_k^d, Y_{max}\}/a_i - Y_j^a/d_i} \\ &- \frac{(\min\{Y_j^d, Y_k^d, Y_{max}\} - Y_j^a)(U_i - r_i - (Y_j^a/d_i))^2}{\min\{Y_j^d, Y_k^d, Y_{max}\}/a_i - Y_j^a/d_i} \\ i, j, k &= N, P, K \quad i \neq j \neq k \end{aligned} \tag{2-12}$$

Y_{max} stands for the maximum yield that is possible given the levels of irradiance, water availability under rain-fed condition and genetic crop properties, cf. the water-limited yield level (van Ittersum and Rabbinge, 1997). In Eq. (2-12) it is avoided that Y_{ij} becomes greater than Y_{max}, or than Y_k^d the yield at maximum dilation of the third nutrient k, thus considering interaction between three nutrients.

The final or ultimate yield (Y_U) is calculated as the mean of the values for all pairs of nutrients.

$$Y_U = (Y_{NP} + Y_{NK} + Y_{PN} + Y_{PK} + Y_{KN} + Y_{KP})/6 \tag{2-13}$$

2.2.2. Case studies

We focused on maize, rice and wheat as the main staple food crops. We collected the secondary data sets from different case studies covering tropical and temperate zones with appropriate data for QUEFTS. The responses to fertilizer applications were analyzed for maize in eastern Africa (Kenya), and USA (Nebraska), for rice in western Africa (Senegal) and Asia (China and Philippines) and for wheat in Europe (UK). The main characteristics of these sites are presented in Table 2-2 and chemical characteristics of their top-soils in Table 2-3.

2.2.2.1. Tropical zones

Rice (Senegal)

The data were obtained from irrigated rice farms in Sahel and Savanna regions of western Africa (Haefele et al., 2003). The data presented here are from the main wet growing season (July-December 1995) of 36 irrigated cases of the Thiagar site in Senegal. This site is situated in Senegal river delta and the soil has developed in fluvial deposits with fine to very fine texture; dominated by clay loam texture. The treatments in this case study included no-fertilizer application (control), NP and NPK applications.

Rice (The Philippines)

Rice data from 26 farms located within a radius of 15-25 km around a research station in the Philippines were obtained from IRRI (1997-1998) (Dobermann et al., 2002; Witt et al., 1999).

On-farm experiments located in Maligaya-Philippines are representative for a large area (≥ 100000 ha) with similar soils and cropping systems. Vertisols and Inceptisols are dominant soil types and common soil textures are clay loam to clay. The area has a humid or sub-humid tropical climate and is located in large inland plains and basins or river deltas with flat topography (Dobermann et al., 2002; Witt et al., 1999). The fertilizer application includes four treatments: fully fertilized treatment (NPK) and three treatments (NP, NK, PK) missing the nutrients K, P and N, respectively.

Maize (Kenya)

The datasets included three sites in western Kenya: Aludeka division in Teso district (0°35′ N; 34°19′ E) in 2002, Emuhaya division in Vihiga district (0°4′ N; 34°38′ E) in 2003 and Shinyalu division in Kakamega district (0°12′ N; 34°48′ E) in 2003.

The dominant soil types in Aludeca are Acrisols, Luvisols, Lixisols, Vertisols, in Emuhaya are Ferralsols and Nitosols and in Shinyalu is Nitosols. Soil texture in Aludeka and Emuhaya sites is sandy clay loam, but in Shinyalu is clay loam (Tittonell et al., 2008; Tittonell et al., 2005). Each site included six farms and in each farm there were three fields including homefield, midfield and outfield. Each field had five treatments i.e. no-fertilizer application (control), three treatments with one of the nutrients missing (NP, NK and PK), and a fully fertilized treatment (NPK).

2.2.2.2. Temperate zones

Rice (China)

The data were obtained from long-term experiments under the rice mono-culture system on a calcareous soil from 1978 to 1991 (Shen et al., 2004). The trial site was located at Hangu Farm in the rice-growing region of Hebei province, northern China. The soil type at the field site is classified as Cambisol and soil texture is silt loam. Five treatments were examined, i.e. no-fertilizer application (control), one treatment with only N application, two treatments with either P or K missing, and one fully fertilized treatment (so control, N, NK, NP and NPK).

Wheat (Broadbalk-UK)

The wheat production data from Broadbalk-UK on silt loam soil were obtained from the Rothamsted Archive (e-RA). Broadbalk is one of the oldest, long-term agronomic experiments in Rothamsted. This field has a long history as arable cropping land; the first experimental winter wheat was sown in 1843. The effect of different combinations of inorganic fertilizer N, P, K, Na and Mg on crop yield was compared with farmyard manure (FYM) and a control treatment of zero manure or fertilizer inputs (Rothamsted, 2006). The experiment comprises different rotation sections. We obtained the data for years 1987, 1992, 1997 and 2000 of two sections of one field with the continuous wheat cropping system. Total P data was only available in year 2000. In the specific wheat cropping system used in this paper, FYM, Na and Mg were not considered, so that only four treatments of two fields were examined, i.e. no-fertilizer application (control), N, NP, and NPK in 2000.

Table 2-2 Geography, climate, soil texture and NPK experimental data of the case studies.

Crop	Country/Site	Rainfall [a] (mm y^{-1})	Temperature [a] (°C)	Soil texture	NPK treatment (kg ha^{-1})
Rice	Senegal/ Thiagar	200	27	Clay loam	N(138,180) P(26) K(31)[(1)]
Rice	Philippines /Central Luzon	1800	28	Clay loam to clay	N(150) P(30) K(90)[(2)]
Maize	Kenya/Aludeka	1460	22.2	Sandy clay loam	N(100) P(100) K(100)[(3)]
	Kenya /Emuhaya	1850	20.4	Sandy clay loam	N(100) P(100) K(100)[(3)]
	Kenya /Shinyalu	2145	20.8	Clay loam	N(100) P(100) K(100)[(3)]
Rice	China/Hangu Farm	600[(4)]	10.5[(4)]	Silt loam	N(213) P(32) K(87)[(5)]
Wheat	UK/Broadbalk	704[(6)]	9.2[(6)]	Silt Loam	N(96) P(35) K(90)[(7)]
Maize	USA/Cairo	718[(8)]	10.5[(8)]	Silt Loam	N(196) P(20) K(40)[(9)]
	USA/Paxton	718[(8)]	10.5[(8)]	Silt loam	N(196) P(20) K(40)

[a] Values for the rainfall is annual and for the temperature is annual average.

(1) (Haefele et al., 2003); (2) (Dobermann et al., 2002); (3) (Tittonell et al., 2008) ; (4) Average between 1978 and 1987; (5)Average 1978-1991 (Shen et al., 2004); (6) Average rainfall between 1997 and 2000, average temperature between 1978 and 1990; (7) e-RA; Rothamsted Archive; (8) Mean precipitation and temperature are based on WMO Climatological Normals (CLINO) for the period 1961-1990; (9) (Setiyono et al., 2010; Wortmann et al., 2009).

Table 2-3 Soil chemical properties (0-20 cm) in different locations

Region	**Crop**	**Country**	**Site**	**C_{org}**	**N**	**P_{Olsen}**	**K_{Exch}**	**pH**
				($g\ kg^{-1}$)	**($g\ kg^{-1}$)**	**($mg\ kg^{-1}$)**	**($mmol\ kg^{-1}$)**	**(H_2O)**
Tropics	**Rice**	**Senegal**	Thiagar (1995)	7.9	1.2	11.7	7.4	5.6
		Philippines	Central Luzon (1997-98)	12.7	1.1	4.3	2.1	6.2
	Maize	**Kenya**	Aludeka (2002)	9.5	0.6	4.4	4.5	5.7
			Emuhaya (2003)	12. 9	1.3	4.2	4.7	5.9
			Shinyalu (2003)	17.1	1.7	3.9	3.7	5.3
Temperate	**Rice**	**China; Hebei Province**	Hangu Farm(1978-1991)	15.7	0.9	7.2	10	7.8
	Wheat	**UK**	Broadbalk (2000)	10.5	1.0	44.6	3.3	7.7
	Maize	**USA-Nebraska**	Cairo (2002-04)	14.5	1.5	11.1	13.9	6.9
			Paxton (2002-03)	12.5	1.3	4.2	10	7.2

Maize (Nebraska-USA)

The data include 34 site-year combinations across Nebraska from 2002 to 2004, covering a major irrigated maize area across Nebraska (Setiyono et al., 2010; Wortmann et al., 2009). These locations represent different environments, soil types and cropping practices in the Western Corn Belt. Nebraska soils are mostly Mollisols developed in loess. These soils are widespread in semi-arid regions in Central Asia and North American Great Plains. Mollisols are deep calcareous soils with high levels of organic matter especially in the deep surface horizon, which are typically 60 and 80 cm thick.

Cropping systems included maize followed by maize, soybean and dry bean. For this study we focused on maize followed by maize only (13 site –years). Among these 13 sites only Cairo had three seasons (2002-2004) and Paxton two seasons (2002-2003) data. Soil texture varied from silt loam in Cairo and loam sand in Paxton. From a great number of fertilizer treatments we chose five treatments, i.e. no-fertilizer application (control), NP, NK, PK and fully fertilized treatments (NPK).

2.3. Model modifications and recalibration of the parameters

2.3.1. Necessity for modifications and recalibration

The major boundary conditions of the original QUEFTS are (i) modern crop varieties are used with optimum harvest index. Harvest index (*HI*) of the crops should be constant or vary within a narrow range. *HI* is supposed to vary between 0.4 and 0.45 for maize (Smaling and Janssen, 1993), 0.4 or higher for rice (Haefele et al., 2003; Witt et al., 1999) and 0.5 for wheat (Kemanian et al., 2007); (ii) the cropping system is rain-fed; (iii) soil nutrients are mainly derived from topsoil; (iv) soil pH is within the range $4.5 < pH(H_2O) < 7.4$.

The equations used in QUEFTS are partly based on experimental data (Steps 1 and 3) and partly on theoretical considerations (Steps 2 and 4). For other crops than maize and other soil and climate conditions it should be tested whether the (default) values of the model parameters can still be applied.

R_N was estimated from the difference in uptake of N between treatments NPK and PK, to be sure that the response to N was not limited by inadequate supplies of P and K. Similarly, R_P was estimated from the difference in uptake of P between treatments NPK and NK, and R_K from the difference in uptake of K between treatments NPK and NP.

QUEFTS estimates crop production that can be obtained when the yield is not limited by the availability of other nutrients. Drought, erosion, pests and diseases should be absent, and crop management is assumed to be good.

In this study, initially the original version of QUEFTS (Janssen et al., 1990) was used to estimate the response of crops (maize, rice and wheat) to different fertilizers applications and soils fertility levels.

2.3.2. Soil supply of available nutrients

Table 2-4 presents the default and recalibrated values, as well as the expressions of the parameters used in the equations relating soil supply of available nutrients to chemical top-soil properties in Step 1 of the model.

2.3.2.1. Alternative relations

For calculation of the soil supply of N, Eq. (2-1) or Eq. (2-4) was applied. It was found that the higher value calculated by these two equations gave a better fit with measured results. Similarly, the higher value calculated by either Eq. (2-2) or (2-5) gave a better fit with measured results than the lower value of Eq. (2-2) or (2-5). The highest values were used in the next model calculations.

In Eq. (2-5), Total P is used. When Total P had not been assessed, a good estimate for it is:

$$\mathrm{P}_{Tot} = 95P_{Olsen} \tag{2-14}$$

The value of 95 was found in areas where no or only little fertilizer P had been applied. Once farmers start to apply inorganic fertilizer P, the ratio of P_{Tot}/*P-Olsen* decreases and may finally be only 10-20% of the original value (Rothamsted, 2006; van der Eijk et al., 2006), making it unjustified to use Eq. (2-14).

2.3.2.2. Temperature

The availability of soil N depends on mineralization of soil organic matter (Janssen, 1996). Because the mineralization rate is strongly related to temperature, the availability of N is greater at higher temperature for a given level of soil organic N. On the other hand the crop development cycle is shortened by higher temperature. Since these two processes cancel each other out within the tropical temperature range from 20 to 27 ^{o}C, no temperature correction was introduced in the original model equations for N. When the model is used in temperate regions with temperatures around 10 ^{o}C in the growing season, it is essential to take temperature effects into account.

The modified Eqs. (2-1) and (2-4)(Table 2-4) take the temperature into account, using an expression that has been applied earlier as well (Smaling and Janssen, 1993; Yang and Janssen, 2002). The parameter α_{NN} is related to the relative mineralization rate of soil organic N. This rate depends on temperature (*T*, in ^{o}C). A practical temperature correction factor is $2^{(T-9)/9}$ (Janssen, 1996), indicating that the relative mineralization rate is doubled with each 9 ^{o}C increase in temperature. The value of $2^{(T-9)/9}$ is 3.4 when *T* =25 ^{o}C and 1 when *T* = 9 ^{o}C. So at 25 ^{o}C, α_{NN} is 3.4 times larger than at 9 ^{o}C. It follows that α_{NN} at 9 ^{o}C is 68/3.4 = 20, and can be described by 20*$2^{(T-9)/9}$ at any temperature (between 9 and 27 ^{o}C). Similarly, α_{N} can be described by 2*$2^{(T-9)/9}$. This broadens the applicability of QUEFTS to areas outside the tropics.

2.3.2.3. pH

In alkaline and calcareous soils the pH will generally exceed 7.0. For such soils the pH correction factors (Eqs. (2-6)-(2-8)) had to be modified, as shown in Table 2-4. The pH range was widened and subdivided into 2-5 sections. For pH between 6.7 and 8, f_P is calculated with an equation that was proposed earlier (Smaling and Janssen, 1993):

$$f_P = 1 - 0.25(pH - 6.7)^2 \qquad (2\text{-}15)$$

When pH is between 6.0 and 6.7 the value of f_P is set at 1 showing a plateau. At pH < 6, the original equation is used. When pH is very low (<4.7) or very high (>8) all pH correction factors get fixed values (Table 2-4).

The modified Eqs. (2-6)-(2-8)(Table 2-4) allow for using QUEFTS for soils outside the original pH boundaries. The equation for f_K has changed from linear to curvilinear, following the procedure used in the Netherlands for clayey soils (van Dijk and van Geel, 2010).

2.3.2.4. Flooding

As mentioned in Section 2.2.2, QUEFTS is meant for rain-fed cropping and soils should be deep and well drained. In some of the case studies, however, irrigated cropping was practised. Under flooded conditions soil chemical and biological processes are considerably different than in aerobic soils as already described (De Datta and Gomez, 1975). Sometimes sub-soil may play a role in the absorption of other nutrients as well. Then, the default parameter values for Eqs. (2-1)-(2-8) are not valid anymore. Therefore, it was tried to adapt Eqs. (2-1)-(2-5) by introducing a ‘flooding’ factor f_F to the parameters.

Table 2-4 Default and recalibrated values of model parameters (Step 1)

Equation	Parameter	Default	Recalibrated	Remarks
2-1	α_N	6.8	$2 * 2^{(T-9)/9}$	Temperature
2-4	α_{NN}	68	$20 * 2^{(T-9)/9}$	T in ^oC
2-3	α_K	400	500	
2-5	Total P	Measured	95 *P-Olsen* [a]	
			0.4	$pH < 4.7$
2-6	f_N	$0.25\ (pH\text{-}3)$	$0.25(pH\text{-}3)$	$4.7 < pH < 7$
			1	$pH > 7$
			0.02	$pH < 4.7$
		$1 - 0.5(pH - 6)^2$	$1 - 0.5(pH - 6)^2$	$4.7 < pH < 6$
2-7	f_P		1	$6 < pH < 6.7$
			$1 - 0.25(pH - 6.7)^2$	$6.7 < pH < 8$
			0.57	$pH > 8$
			1	$pH \leq 4.5$
2-8	f_K	$2.21 - 0.25\ pH$	$6.1 pH^{(-1.2)}$	$4.5 < pH < 6.8$
			0.6	$pH > 6.8$

[a] only for soils in Kenya that did not receive chemical fertilizer P

2.3.3. Yield uptake affected by harvest index (*HI*)

When there is a large variation in *HI* within one data set, the following approach instead of Eqs. (2-10) and (2-11) is appropriate. The uptake of Nutrient *i* (U_i) is the sum of nutrients in grain and straw:

$$U_i = DM_g.\chi_{gi} + DM_s.\chi_{si} \tag{2-16}$$

where DM_g and DM_s stand for dry-matter production (kg), χ_{gi} and χ_{si} for the mass fraction (g kg^{-1}) of nutrient *i* in grain and straw, respectively.

The physiological efficiency (PhE_i) is DM_g/U_i (in kg kg^{-1}) and the proportion DM_g/DM_s is *HI/(1-HI)*. If *DM* denotes the sum of DM_g and DM_s, it holds:

$$U_i = 1000.(HI.DM.\chi_{gi} + (1 - HI).DM.\chi_{si}) \tag{2-17}$$

$$PhE_i = 1000.HI/(HI.\chi_{gi} + (1 - HI)\chi_{si}) \tag{2-18}$$

So, PhE_i increases with increasing *HI* less than proportionally when $\chi_{gi} > \chi_{si}$ (as is the case for N and P), and more than proportionally when $\chi_{gi} < \chi_{si}$ (as is the case for K). For maximum PhE_i (*d*), χ_{gi} and χ_{si} should be minimum, and for minimum PhE_i (*a*), χ_{gi} and χ_{si} should be maximum.

For the accumulation and dilution parameters a_i and d_i in the empirical Eqs. (2-10) and (2-11) used in Step 3 of the model, the values presented in Table 2-5 were used, unless indicated else in the following sections. Sometimes there were reasons to assume that the harvest index did not have the default values, and then Eq. (2-18) was introduced in Eqs. (2-10) and (2-11). The values of r_i were set at zero.

$$Y_i^a = a.U_i \quad \text{where } a = 1000.HI/(HI.\chi_{gimax} + (1 - HI).\chi_{simax}) \tag{2-19}$$

$$Y_i^d = d.U_i \quad \text{where } d = 1000.HI/(HI.\chi_{gimin} + (1 - HI).\chi_{simin}) \tag{2-20}$$

The subscripts *imin* and *imax* of χ_g and χ_s refer to the minimum and maximum mass fractions (g kg^{-1}) of nutrient *i* in grain (G) and straw (S).

Recalibration of a_i and d_i (see Eqs. (2-10) and (2-11)) with Eq. (2-18) is only possible when *HI* is known.

Table 2-5 Measured and calibrated accumulation (a) and dilution (d) parameters (kg grain per kg nutrient).

Author/year	**Location**	**Crop**	**N**		**P**		**K**	
			a	*d*	*a*	*d*	*a*	*d*
Janssen et al. (1990)	Kenya, Surinam	Maize	30	70	200	600	30	120
Smaling & Janssen (1993)	Kenya	Maize	30	80	160	600	30	120
Liu et al. (2006)	China	Maize	21	64	126	384	20	90
Setiyono et al. (2010)	USA/Indonesia/Vietnam	Maize	40	83	225	726	29	125
This study	*USA/Nebraska/Cairo*	*Maize*	*32*	*75*	*155*	*595*	*39*	*170*
This study	*Kenya (after HI correction)*	*Maize*	*26*	*60*	*180*	*540*	*24*	*96*
Liu et al. (2006)	China	Wheat	25	56	171	367	24	67
Pathak et al. (2003)	India	Wheat	27	60	162	390	20	59
Maiti et al. (2006)	India	Wheat	35	100	129	738	17	56
This study	*Broakbalk/UK*	*Wheat*	*30*	*70*	*135*	*500*	*25*	*70*
Witt et al. (1999)	Six Asian counties	Rice	42	96	206	622	36	115
This study	*Philippines*	*Rice*	*48*	*96*	*206*	*589*	*36*	*102*
Das et al. (2009)	India	Rice	31	87	192	678	33	81
Haefele et al. 2003	Sahelian west Africa (Senegal)	Rice	48	96	206	589	36	102
This study	*China/ Hebei Province*	*Rice*	*48*	*96*	*206*	*589*	*36*	*102*

2.4. Results

2.4.1. Rice (Senegal)

There are no significant differences between observed and model calculated yields (Figure 2-4). Although the original model gave good results, and hence there was no reason to recalibrate the model parameters, the model was also run with the recalibrated values of Table 2-4. In the original paper (Haefele et al., 2003) R_N, R_P and R_K were set at 0.50, 0.26 and 0.53; in the recalibrated model only R_P had to be changed. The best fit was obtained with R_P set at 0.20. In the modified Eq. (2-1) (see Table 2-4), 26.5 ^{o}C was substituted for temperature (T). Values of a_i, (Eqs. (2-9) –(2-12)) were found (Haefele et al., 2003) to be 48, 206 and 36 for i = N, P and K, respectively. The values of d_i proved to be 96, 589 and 102, while r_i was 0 for all three nutrients (Table 2-5).

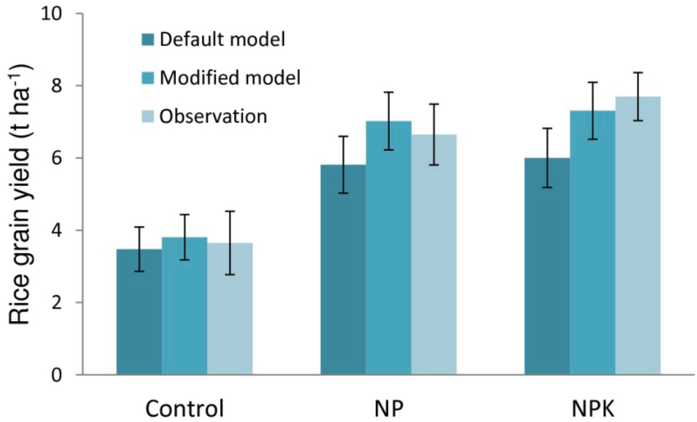

Figure 2-4 Rice in Senegal: comparison of the outcomes of the default model and the recalibrated model with the observations for three fertilizer treatments. Each column represents the average yield of 12 fields (error bars refer to the standard deviation value in each data series).

2.4.2. Rice (The Philippines)

For the Philippines case study, there were no measured yield data available for the control treatment (Figure 2-5). Calculated yields with the default model in treatments NPK and PK matched better with the observations than the NP and NK treatments. The model underestimated the effect of applied N at low K and P values, respectively. Differences between simulated and observed yields might be explained by physical processes not included in the model such as delivery of nutrients to crops from nutrients supplied by rainwater, irrigation water and sedimentation. Also, the mobility of added nutrients improves under wet conditions.

All such processes were taken into account by allocating a factor f_F to the parameters in Eqs. (2-2) and (2-3) (Section 2.3.2.4). Their values were calibrated by curve fitting. In Eq. (2-2), α_P and β_P were multiplied by f_F=1.6, and α_K in Eq. (2-3) was multiplied by f_F =1.2. The maximum recovery fractions of N and P were found to be 0.70 and 0.20, respectively. The recovery fraction of K was set at 0.50, but it is of less importance because K was not strongly limiting as seen from the difference in observed yields between NPK and NP (Figure 2-5). Using the calibrated parameters, the agreement between calculated and measured yields is better than in case of the original QUEFTS parameter values (Figure 2-5). For α_N in Eq. (2-1), the original value of 6.8 was kept, corresponding to a temperature (T) of 25 ^{o}C.

2.4.3. Maize (Kenya)

For the calibration of QUEFTS a selection was made from the data obtained by the authors (Tittonell et al., 2008). Data from Shinyalu were excluded, because this region has steep slopes where soil erosion may have affected yields. Furthermore, homefields were excluded, as they often are heterogeneous because of spot-wise application of household wastes. Yields calculated with the default model were somewhat higher than observed yields for treatments PK, NP and NPK, and lower for the Control and NK, but the average value across all treatment of the default model was not far from the observed average (Figure 2-6a, Table 2-6).

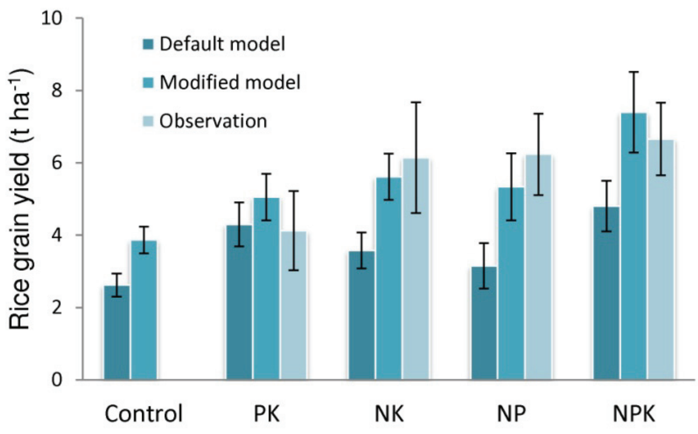

Figure 2-5 Rice in Philippines: comparison of the outcomes of the default model and the recalibrated model with the observations for five fertilizer treatments. Each column represents the average yield of 42 fields (error bars refer to the standard deviation value in each data series).

The data set included values of the harvest index (*HI*), with an average value of 0.34. According to Eq. (2-16), using data of χ_{gi} and χ_{si} presented by Nijhof (1987), PhE_i for N, P and K at *HI*=0.34, is approximately 0.85, 0.90 and 0.80 times the values found at *HI*=0.40, required for QUEFTS (Smaling and Janssen, 1993). So, it was necessary to take the influence of the harvest index into account. After introduction of the effect of the harvest index, the average yield was about 14% lower than that of the default model, and 12% below the observed average value (Table 2-6). Then, the recalibrated pH correction factors shown in Table 2-4 were introduced in the model, resulting in a slight increase of the yields of all treatments (Table 2-6). Furthermore, the calculations of soil available nutrients were adopted as was done in the other sites: for N the higher result of Eqs. (2-1) and (2-4) was taken, for P the higher value of Eqs. (2-2) and (2-5), and for K the parameter α_K was set at 500. This resulted in an overestimation of the yield, especially for the NP and NPK treatments (Table 2-6). The average model yield was practically equal to the average observed yield when the recovery fraction was set to 0.32 instead of 0.5.

Table 2-6 Stepwise recalibration of QUEFTS for the Kenya maize data (t ha^{-1}).

Model version and introduced parameters	**Control**	**PK**	**NK**	**NP**	**NPK**	**Average**
Observation	2.1	2.5	2.8	3.6	3.9	2.9
Default model	1.6	2.7	2.3	3.9	4.7	3.0
Harvest index	1.4	2.3	2.0	3.3	4.1	2.6
HI and pH	1.6	2.3	2.2	3.4	4.2	2.7
HI, pH, S_N, S_P,S_K	1.9	2.9	2.5	3.9	4.6	3.2
HI, pH, S_N, S_P,S_K , R_N	1.9	2.9	2.4	3.5	4.1	2.9

Finally, yields of the homefields of Aludeka and Emuhaya, and of all fields of Shinyalu were calculated with the recalibrated model. Figure 2-6b shows that the average of the calculated yields of the homefields is practically equal to that of the observed yields, while the calculated yields of Shinyalu are higher than the observed yields. This is caused by the difference in harvest index, which was 0.22 in Shinyalu, while it was 0.34 in Aludeka and Emuhaya (experimental data from (Tittonell et al., 2008)). As a result the recalibrated model overestimated the yields in Shinyalu. Although the averages of calculated and observed yields of the home-fields are equal, temporal and spatial heterogeneity created a large variability around the 1:1 line.

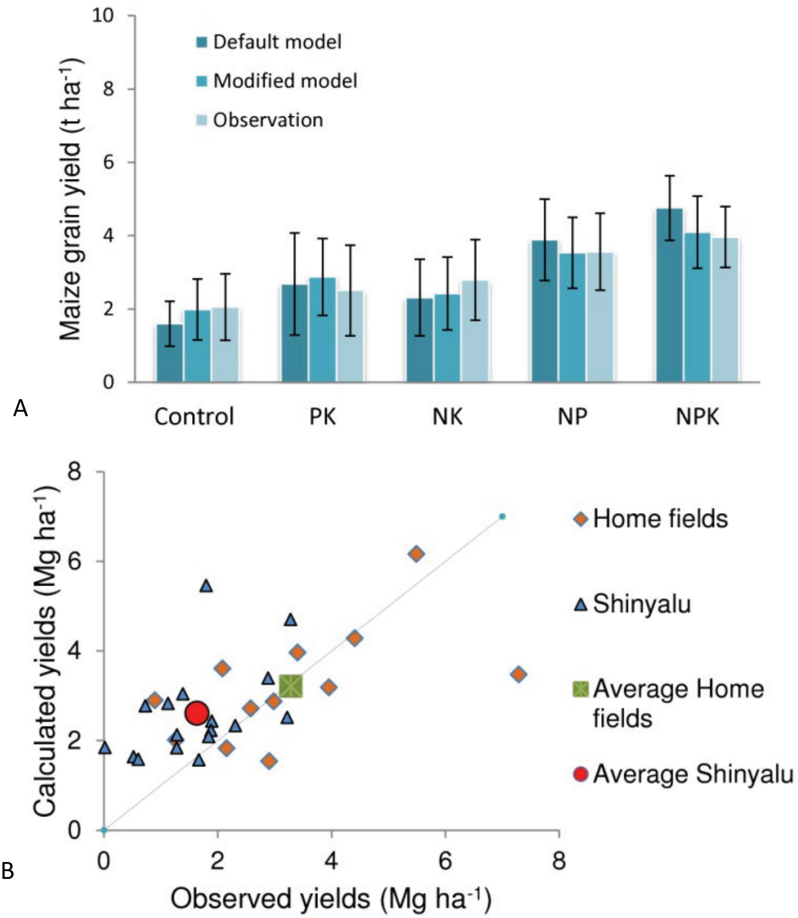

Figure 2-6 (a) Maize in Kenya: midfields and outfields of Aludeka and Emuhaya. Comparison of the outcomes of the default model and the recalibrated model with the observations for five fertilizer treatments. Each column represents the average yield of 23 fields (error bars refer to the standard deviation value in each data series). (b) Maize in Kenya: Homefields of Aludeka and Emuhaya, and all fields of Shinyalu. Comparison of the outcomes of the recalibrated model with the observations.

2.4.4. Rice (China)

The original QUEFTS model was applied to the rice-growing region of Northern China in Hebei Province. For this experiment the nutrient uptake data to calculate the accumulation and dilution coefficients were not available. Therefore we used the same values for *a* and *d* parameters as in Senegal and the Philippines, i.e. 48 and 96 kg grain kg^{-1} N, 206 and 589 kg grain kg^{-1} P and 36 and 102 kg grain kg^{-1} K, respectively (Table 2-5).

As soils were sampled and analyzed each year, QUEFTS calculations were performed for the annual soil data. The long term data from 1978 to 1991 show that the rice grain yields do not show any response to K and a moderate response to P. Average fertilizer application rates are presented in Table 2-2 and soil data in Table 2-3. Details of soil data, fertilizer applications and yields are shown in the original paper (Shen et al., 2004).

With the default values of the model parameters simulation results of rice grain yield were far below the observed yields (Figure 2-7). A major problem was the high pH resulting in a negative value of f_P (-0.62) and low value of $f_{K.}$ (0.175) resulting in low supplies of P and K, i.e. ca 1.5 and 35 kg, respectively. After introduction of the recalibrated model parameters in Step 1 (Table 2-4), the model simulation results were in better agreement with observations, but still too low (not shown). Taking into account that under irrigation nutrient availability (particularly P) increases, the soil P parameters of Eq. 2-2 (α_P and β_P) were multiplied by 1.2 and P recovery was set at 0.20, the same as in Senegal. Further improvement of the model was obtained by setting the recovery fraction of N at 0.60. The differences between the yields calculated with the recalibrated model and the observed yields were not significant (Figure 2-7).

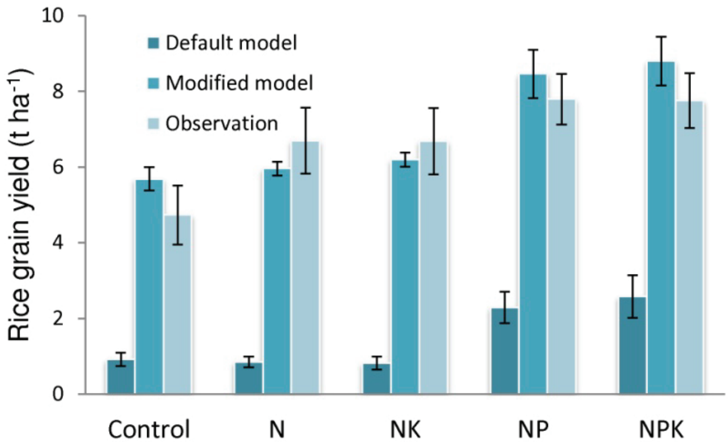

Figure 2-7 Rice in China: comparison of the outcomes of the default model and the recalibrated model with the observations for five fertilizer treatments. Each column represents the average yield of 14 years-fields (error bars refer to the standard deviation value in each data series).

2.4.5. Wheat (Broadbalk, UK)

So far, the QUEFTS model has only been applied in tropical and subtropical regions for the wheat crop (Liu et al., 2006; Maiti et al., 2006; Pathak et al., 2003). Using the data of some Chinese case studies, and after calculations with Eqs. (2-19) and (2-20) in which minimum and maximum nutrient mass fractions from literature (Nijhof, 1987) were substituted, accumulation (*a*) and dilution (*d*) values were set at 30 and 70 kg grain kg^{-1} N, 135 and 500 kg grain kg^{-1} P and 25 and 70 kg grain kg^{-1} K, respectively. Using these values and the default QUEFTS parameter values, wheat yield was calculated for the Broadbalk case study.

For the calculation of the soil supply of P, Eq. (2-2) could not be used because of the long history of fertilizer P application. Instead we used Equation (2-5), but unfortunately, Total P was only analyzed in year 2000 and not in the other years. Figure 2-8 shows that QUEFTS with default parameter values simulated wheat yield fairly well only in case of complete fertilization (NPK). Because pH exceeded 7.4, *f P* was negative and the soil did not supply any P, resulting in zero or almost zero yields when no P was applied. In all treatments (except NPK) the differences between the default model results and observed yields were large, even when P was applied (Treatment NP). Obviously, the supply of K was underestimated because of the high pH. This high pH also explains why the estimated soil supply of N is relatively high, but because of the low P and K only a small part (about 20%) of the available N is actually taken up, according to Step 2 calculations. To solve the problems created by the pH well above the boundary value of 7, the expressions for *fP* and *fK* were recalibrated (Table 2-4). Using these formulations, the calculated soil supplies of P and K and uptake of available N increased considerably. The estimated N supply was also too high because the default value of 68 of α_{NN} in Equation (2-4) is valid in tropical situations with an average temperature of about 25 ^{o}C, but not in the UK with average temperatures of ca. 9^{o}C. Figure 2-8 shows that the yields calculated with the recalibrated model are in good agreement with the observed yields.

2.4.6. Maize (Nebraska, USA)

Calibration of Step 1 of the model was done for the only site with data for three seasons (2002-2004) with full fertilizer treatments and maize-maize cropping system (Cairo Figure 2-9).

The default model strongly underestimated yields. Likely, the measured yields were so high because crops were irrigated, resulting in higher availability of soil as well as of fertilizer nutrients. To take improved availability into account, an irrigation factor (f_F) was introduced, as in the Philippines.

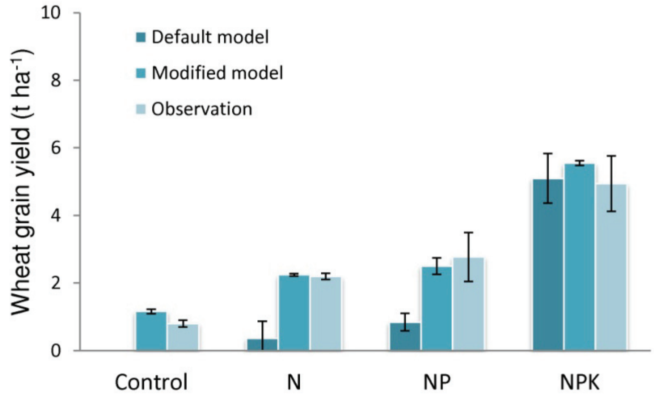

Figure 2-8 Wheat in Broadbalk: comparison of the outcomes of the default model and the recalibrated model with the observations for four fertilizer treatments. Each column represents the average yield of 2 fields (error bars refer to the standard deviation value in each data series).

It proved necessary, however, to assume higher values for f_F in Eqs. (2-2) and (2-3) for Nebraska than for the Philippines, and also to multiply α_N and α_{NN} in Eqs. (2-1) and (2-4) with a factor f_F. This may be a consequence of the relatively high values of C_{org}, P-Olsen and exchangeable K in Nebraska's sub-soils. By curve fitting the values of f_F in Eqs. (2-1), (2-2) and (2-3) were found to be 1.75, 1.70 and 1.35, respectively. The recovery fractions of N, P and K were estimated at 0.60, 0.20 and 0.50, respectively, also by curve fitting. The yields calculated with the recalibrated model did not significantly differ from the observed yields in Cairo (Figure 2-9).

Next, it was verified whether the 'Cairo model' could also be used for the other area in Nebraska (Paxton). Figure 2-10 shows that the yield for the Paxton site calculated with the Cairo model were closer to the observed yields than the yields calculated with the default model, but still differed significantly from the observed yields. Further adaptation of f_F was needed to arrive at a good agreement with the observed yields; in the Paxton model f_F is 3.38 and 3.37 in Eqs. (2-1) and (2-2), respectively, while for the estimation of soil available K in Eq. (2-3) no factor f_F was required.

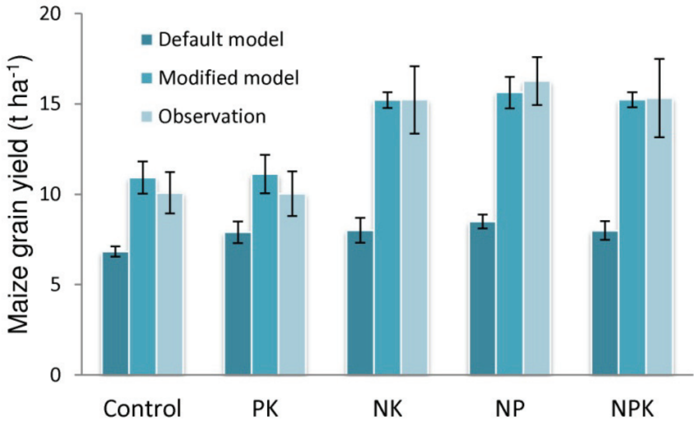

Figure 2-9 Maize in Nebraska (Cairo): comparison of the outcomes of the default and recalibrated models with the observations. Each column represents the average yield of three year-field combinations (error bars refer to the standard deviation value in each data series).

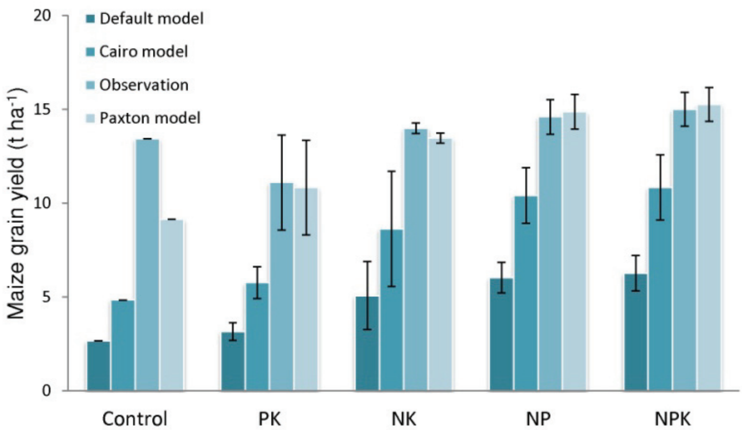

Figure 2-10 Maize in Nebraska (Paxton): comparison of the outcomes of the default and two recalibrated models with the observations. Each column represents the average yield of two year-field combinations, except for the control.

2.5. Discussion

2.5.1. Performance of QUEFTS

Modeling crop production in response to soil fertility and nutrient inputs at the global scale is challenging, mainly due to the complexity of the dynamics of nutrients in soils and the relation with plant uptake. Heterogeneity of climatic conditions, soils, crops and crop management makes it almost impossible to develop simple, yet adequate models for this broad range of conditions.

The original QUEFTS model was based on results of field experiments in Suriname and Kenya with rain-fed maize as test crop. Although its use under other ecological conditions repeatedly required recalibration of some model parameters, there was no need to modify the structure of QUEFTS, consisting of assessment of supply of available nutrients, uptake of nutrients as a function of proportions of available N, P and K, and crop production in response to nutrient uptake. The most precarious part of the model remains the relationship between chemical soil data and the supplies of available soil nutrients (Eqs. (2-1)–(2-8)), as many local environmental factors may interfere. The other parts of the model (Eqs. (2-9)–(2-20)) have a generic character, and depend on crop type.

It is impossible, however, to deal with heterogeneity in (unmeasured) soil properties in the field, and with variation in weather conditions from season to season. Weather conditions have a distinct effect on the recovery fractions of fertilizer nutrients, making it sometimes necessary to adapt the value of R_N, R_P and R_K in Eqs. (2-1)-(2-5) to the prevailing weather conditions. Once the values for the parameters of Eqs. (2-1)-(2-8) have been adapted, and the crop nutrient requirements are known, application of QUEFTS leads to good results. Major problems encountered in the use of QUEFTS were related to availability of data on soils and fertilizer field trials, to irrigation and to extreme soil pH values. Below we discuss issues related to the data availability, the dynamics of soil P pools and residual soil P, and modifications required for QUEFTS application in flooded systems, subsoil nutrients and high pH soils.

2.5.2. Data availability

Although it was the intention of the developers of QUEFTS to minimize the number of required soil data, data availability was a large challenge for this study. For a proper testing of QUEFTS also information on crops (*a, d,* HI) is requested. Four chemical topsoil parameters (C_{org}, P-Olsen, exchangeable K, and pH (H_2O)) should be known and preferably also N_{org} and Total P.

In an exploratory study done by ISRIC at the global level, of the 11000 soil profiles in ISRIC WISE database only 1147 data points have P-Olsen included (Batjes, 2010). These data cover only 15 different countries (967 are from Botswana and Yemen) and hence refer to a limited range of global soil profiles. To make the present set of QUEFTS parameters representative for extrapolation to the global level it is recommended to calibrate the model also for other soil P analyses than P-Olsen, e.g. P-Bray I, P-AL, P-water. When

QUEFTS was developed in the 1980s, however, P-Olsen proved better related to soil P supply than the other ‘available P’ indices (Janssen et al., 1990).

The minimum demand for data of cereal crops is grain yield, but the possibilities for testing become much greater if also straw yield is known as well as the nutrient mass fractions in grain and straw. If so, the accumulation (*a*) and dilution (*d*) of nutrients can be calculated, and the experiment-based Steps 1 and 3 of QUEFTS can be directly investigated instead of indirectly via comparison of observed and estimated yields.

Our study also supports the recommendation (Janssen, 2011) of the use of 2^3 factorial fertilizer trials (treatments: Control, N, P, K, NP, NK, PK, and NPK) for testing the performance of QUEFTS, and for (re)calibration of its soil and crop related parameters. Unfortunately such experimental data are rarely available. Second best are experiments with the treatments NP, NK, PK, and NPK and preferably also a control without any NPK inputs.

2.5.3. Residual soil P

QUEFTS is a static model that does not capture, by definition, the dynamics of soil P pools. It is meant to be valid for the first growing season following soil chemical analysis. Generally, crops can take up 10-20% of the applied fertilizer P in a particular growing season, while a substantial amount of applied P accumulates in the soil as “residual P” (Sattari et al., 2012; Syers et al., 2008; Wolf et al., 1987). Plants can use this source of P for many subsequent years. This residual effect is, however, not captured by QUEFTS. In some cases, observed high recovery of fertilizer P could be explained by and corrected for P applications in preceding years, resulting in more realistic estimates of first-season recovery (Janssen, 2011; Smaling and Janssen, 1993), but such corrections were not possible in the present study.

2.5.4. Flooded and irrigated systems

Differences between QUEFTS-simulated and observed yields might be explained by physical processes not included in the model such as delivery of nutrients to crops from nutrients supplied by rainwater, irrigation water and sedimentation. The amount of nutrients in irrigation water may be considerable, compared to the low levels of inherent soil nutrients (Dobermann and Oberthür, 1997). K input in rain and irrigation water could account for about one quarter of K uptake in Southeast Asian rice cropping systems (Hoa et al., 2006) and thus can influence observed yields. So far, only two sources of nutrients have been included in QUEFTS, i.e. inherent soil fertility (in the original version only for the top 20 cm soil layer) and fertilizer applications (also manure can be included).

Step 1 of QUEFTS was originally calibrated for rain-fed cropping conditions. Parameter values of Step 1 cannot be directly applied to model flooded systems, where under oxygen-deficient conditions several soil chemical processes change by which the solubility of especially P increases. Anaerobic conditions cause reduction of ferri phosphates into ferro phosphates of which the latter have a higher dissolution product and hence dissolve better. Further, the movement of nutrients from soil to root is easier in submerged soils than in soils at field capacity or drier conditions.

As far as we know, the issue of irrigation has so far not explicitly been addressed in studies using the QUEFTS model. In studies at IRRI the difficulties of the assessment of soil nutrient supplies on the basis of chemical soil data was circumvented by using simple field trials to estimate the indigenous N, P and K supplies (Dobermann and Fairhurst, 2000). We have incorporated the effect of flooding by introducing a ‘flooding’ factor of which the value depends on the intensity of flooding, maintaining the original set-up of QUEFTS to estimate soil nutrient supply on the basis of soil chemical properties.

2.5.5. Subsoil nutrient supply

Major problems were encountered with the Nebraska data. There, the average maize production is 15 t ha^{-1}, which is much higher than in the other experiments. Nebraska is known to be a high-yielding region, i.e., deep fertile sub-soils, irrigation and high irradiation together create an environment that is more favorable for crop growth than the environments for which QUEFTS has been developed. Nutrient uptake in QUEFTS is calculated based on the topsoil (0-20 cm) properties. In soils with deep A horizons, however, roots may take up nutrients also from below 20 cm depth. The mentioned ‘flooding’ factor (f_F) was supposed to capture the contribution of deeper soil layers to nutrient uptake too. That the factor f_F had to take all these beneficial factors is seen as an ‘emergency’ solution.

Quantitative relationships between chemical soil properties of the sub-soil and potential supply of nutrients by such soils could not be established in the present study, because of the confounding with irrigation. This makes application of standard parameter values in the equations of Step 1 of the QUEFTS model practically impossible in situations with irrigation and fertile sub-soils. It must be concluded that QUEFTS is a too simple model to simulate the soil-input-crop relationships in Nebraska.

2.5.6. High pH soils

Because QUEFTS was originally developed in the humid tropics where most soils have a rather low pH, the pH correction factors used in Step 1 had a weak validity at higher pH, and were not at all applicable at pH above 7. It was a challenge in our exercises to adapt QUEFTS' pH correction factors (f_N, f_P, f_K) to soils with a high pH. The resulting equations and boundaries of pH ranges shown in Table (2-4) illustrate the complexity of pH effects on soil nutrient availability. Further testing remains desirable.

2.6. Conclusions

To our knowledge, this is the first time that a study is presented on application of the QUEFTS model to different climatic conditions. Two types of modifications were made: (i) change of parameter values by recalibration; (2) change in model equations. The model structure was not changed. The model equations were adapted to broaden the model applicability beyond the original boundary conditions of pH, rain-fed cropping, tropical temperatures and optimum harvest index.

As a result, the adjusted QUEFTS model performed fairly well under the conditions studied in this paper. The underlying idea of the model, which considers the proportions of three nutrients jointly to determine the final yield proved appropriate. Although the original model has been developed for tropical soils, our results indicate that proper modification of the parameters via a temperature correction factor makes it applicable to temperatures prevailing in other environments.

Since various processes (e.g. availability of other nutrients than N, P and K, sub-soil properties and irrigation) are ignored in the model, but may differ dramatically across the globe, QUEFTS requires verification and sometimes recalibration of the parameter values before it can be applied.

Major limitations of QUEFTS application are soil heterogeneity and annual variability of weather conditions. QUEFTS cannot take such variations into account, and its results should be considered as yield estimates for average soil properties and average years. The modifications and recalibration presented in this paper increase the applicability of the model, which is useful because QUEFTS can estimate the soil fertility and response to nutrients (and thus also the requirements of nutrient inputs) for the three major cereal crops (wheat, maize and rice) in both tropical and temperate locations. However, the empirical parts of the model do not allow QUEFTS to be become a generic model with global applicability, capturing all combinations of soil, climate and management conditions.

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Residual soil phosphorus as the missing piece in the global phosphorus crisis puzzle

This chapter has been published as:
Sheida Z. Sattari, S. Z., Bouwman, A. F., Giller, K. E., and Van Ittersum, M. K. (2012). Residual soil phosphorus as the missing piece in the global phosphorus crisis puzzle. Proceedings of the National Academy of Sciences 109, 6348-6353.

Phosphorus (P) is a finite and dwindling resource. Debate focuses on current production and use of phosphate rock rather than on the amounts of P required in future to feed the world. We applied a two-pool soil P model to reproduce historical continental crop P uptake as a function of P inputs from fertilizer and manure and to estimate P requirements for crop production in 2050. The key feature is the consideration of the role of residual soil P in crop production. Model simulations closely fit historical P uptake for all continents. Cumulative inputs of P fertilizer and manure for the period 1965-2007 in Europe (1115 kg P ha^{-1} of cropland) grossly exceeded the cumulative P uptake by crops (360 kg ha^{-1}). Since the 1980s in much of Europe, P application rates have been reduced while uptake continued to increase. This is due to the supply of plant available P from residual soil P pool. We estimate that between 2008 and 2050 a global cumulative P application of 700-790 kg ha^{-1} of cropland (in total 1070-1200 Tg P) is required to achieve crop production according to the various Millennium Ecosystem Assessment (MEA) scenarios. We estimate that average global P fertilizer use must change from the current 17.8 to 16.8-20.8 Tg yr^{-1} in 2050, which is up to 50% less than other estimates in the literature that ignore the role of residual soil P.

3.1. Introduction

Significant improvements in agricultural productivity and efficiency of resource use are required to secure food production for the projected world population in 2050 (Koning et al., 2008). Food production needs to grow faster than the global population due to changing human diets, with increasing per capita consumption of meat, and production of biofuels. This challenge can be managed through significant improvements in agricultural productivity and P fertilizer use efficiency.

P is essential for plant growth and often a major limiting nutrient in agriculture(Koning et al., 2008; Smil, 2000; Syers et al., 2008). Many studies have raised concern about rapid depletion of the world's P reserves (Cordell et al., 2009; Dery and Anderson, 2007). Recently, it was suggested that global P production will peak by 2033 (Cordell et al., 2009). In contrast, other studies conclude that almost half of the currently available P resources will be depleted by 2100 (Van Vuuren et al., 2010), or that P rock reserves will be available for the next 300-400 years (Van Kauwenbergh, 2010).

Resource use efficiency, including strategies such as recycling of human P sewage sludge and other waste materials containing P, and reducing runoff and erosion, will be important to improve the sustainability of human P cycle (Van Vuuren et al., 2010).

Readily available P in the soil solution provides most of the plant-available P. The two main factors that control the availability of P to plant roots are the concentration of phosphate ions in the soil solution and the P-buffer capacity, i.e. the ability of the soil to replenish these ions when plant roots remove them (Syers et al., 2008). Soils differ in their buffering capacity and in the extent to which they fix P in insoluble compounds that are unavailable for uptake. P fixation depends on the amount of iron and aluminium oxides (properties that are particularly relevant in strongly weathered tropical soils) or high calcium activity (Fairhurst et al., 1999; Sanchez, 1976). Following a traditional misconception that P fixation is dominant and irreversible, P has been used excessively in agricultural systems for decades in many industrialized countries (Smil, 2002).

A critical concentration of readily available P must be maintained to obtain good crop yields (Syers et al., 2008). Crop uptake is generally only 10-20% of the P fertilizer applied in the first year, but a substantial part of applied P accumulates in the soil as "residual P" (Syers et al., 2008; Wolf et al., 1987). The residual P is the difference between P inputs (mineral fertilizer, manure, weathering and deposition) and P outputs (withdrawal of P in harvested products, and P loss by runoff or erosion) (Bouwman et al., 2009). The residual value of P refers to P that can be taken up by crops for many years (Nuruzzaman M, 2005; Syers et al., 2008) depending on soil P fixation capacity, soil pH, crop species, and P application time (Sanchez, 1976). In an ideal situation, when adequate P is present in the readily available pools, annual P inputs from fertilizer equal to the plant P uptake may be adequate to maintain good crop yields (Syers et al., 2008). Where the amount of readily available P is below a critical level, the rate of P release from residual P is insufficient to sustain optimal crop yields.

Here, an analysis is presented of the historical and future demand of P in global crop production systems. The key feature of this study is the contribution of residual P to the available P for crop uptake at the continental and global scale. A Dynamic Phosphorus Pool Simulator (DPPS) - a simple two-pool P- model (Wolf et al., 1987) including labile and stable pools and long-term P input and output data are applied in this analysis to simulate the P transformations in soil, the build-up of residual soil P pool and crop P uptake (Figure 3-1).

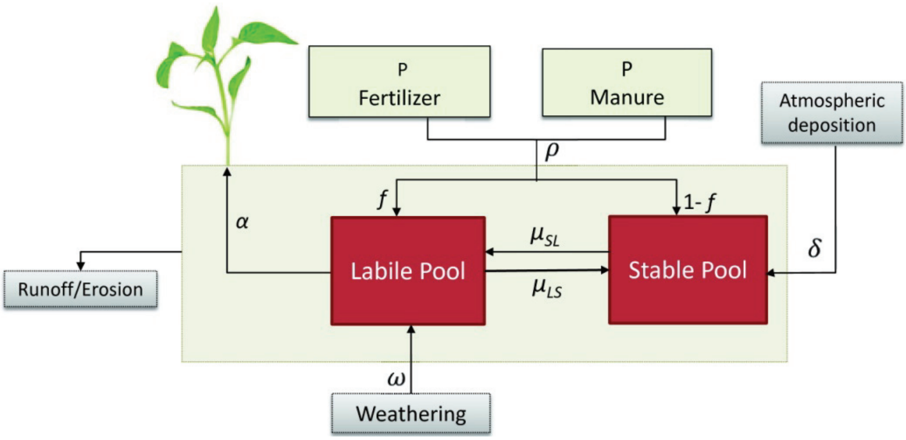


Figure 3-1 Scheme of the DPPS model. The model includes two dynamic pools of P; the labile (P_L) and the stable (P_S) pools, comprising both organic and inorganic P. Four inputs of P to the system are defined; fertilizer, manure, weathering and deposition. The coefficients *f* and 1- *f* refer to the fraction of ρ that transfers to the P_L and P_S, respectively. Coefficient α represents the crop P uptake fraction from P_L. Parameter's ω and δ are weathering and deposition inputs to the P_L and P_S, respectively. μ_{LS} and μ_{SL} denote to the transfer rate of P from the P_L to the P_S and from P_S to the P_L, respectively (redrawn from (Wolf et al., 1987)).

3.2. Material and Methods

3.2.1. Model structure

In soil P models generally different pools are distinguished. In our continental and global analysis, we used the DPPS model, which distinguishes two soil P pools (labile, stable). This is because increasing the number of pools from two (labile, stable) to three (active, labile and fixed) was shown not to improve model predictions, and makes calibration more difficult (Bhogal et al., 1995).

The DPPS model was used to simulate the long-term historical P uptake by crops for time series of P inputs, and to estimate the future P inputs for a specific future target P uptake. Different sources of P input to the system have been defined in the model, i.e. fertilizer, manure, weathering and atmospheric depositions (Figure 3-1). Withdrawal of P in harvested crops (uptake) and runoff (erosion) are two outflows from the system.

P inputs (excluding the runoff loss) are allocated to two dynamic P pools, namely the stable (P_S ; 20%) and the labile P pools (P_L ; 80%). The model simulates the P transfers between the pools, the uptake of P by the crop and the size of both pools (Figure 3-1). To calculate the dynamics of P in these two pools, two differential equations are used:

$$\mathrm{dP}_L/dt = f\rho + \mu_{SL}P_S - \mu_{LS}P_L - \alpha P_L + \omega \qquad (3\text{-}1)$$

$$dP_S/dt = (1-f)\rho + \mu_{LS}P_L - \mu_{SL}P_S + \delta \qquad (3\text{-}2)$$

The rates of P transfer from P_L to P_S and vice versa are denoted by μ_{LS} and μ_{SL}, respectively (yr^{-1}). The coefficient ρ refers to the total P input (mineral fertilizer and manure) after subtracting runoff loss of P. The coefficients f and 1-f refer to the fraction of ρ that transfers to P_L and P_S, respectively. Coefficient α represents the crop P uptake fraction from P_L. Parameters ω and δ are weathering and deposition inputs to P_L and P_S, respectively. A large μ_{LS} makes P_L less available for plant uptake and a large μ_{SL} indicates that the stable pool acts as a buffer that replenishes the labile pool.

The model can also be formulated in a target-oriented approach (Van Ittersum and Rabbinge, 1997), in which the (future) P uptake is a model input and the P application a result assuming no change in cropland area.

In Western Europe, Asia and Latin America, the rapidly increasing P inputs did not lead to a similar increase of P uptake (Figure 3-2a, d and e). Therefore, with increasing rate of P application and growing size of the P_L, a coefficient to limit the uptake fraction was imposed (Janssen et al., 1987).

We accounted for the dynamics of cultivated land as follows. Each year, the initial conditions (with no fertilizer history) are assigned to the new, additional area and the crop P uptake in each segment is calculated based on the history of that segment and finally weighted by its corresponding fractional area. This allows us to have differentiated productivity and soil contents of residual P for different parts of the total area.

3.2.2. Data used

Long-term crop yield, annual P fertilizer consumption and areas of arable crops (1965-2007) were obtained from FAO (FAO, 2011), including different world regions.

We distinguished globally 161 different crops. P contents for each crop were obtained from different sources (Bouwman et al., 2005a; USDA, 2006, 2010). Harvested P was calculated as the production times the P content of the harvested product.

Total P in manure production within pastoral, mixed and industrial livestock production systems was computed from the animal stocks within these systems and their P excretion rates, based on various sources (Service, 1985; Sheldrick et al., 2003; Smith, 1991; Van Horn et al., 1996; Wilkerson et al., 1997). Animal manure available for application to crops and grassland includes all stored or collected manure and excludes excretion in grazing land and animal manure used for other purposes (fuel, building material), or manure not used at all (such as manure from confined animal feeding operations, CAFOs, collected in lagoons).

We used global values from Liu et al. (2008) for P supply from weathering (1.6 Tg yr^{-1}) and atmospheric deposition (0.4 Tg yr^{-1}).These values were more conservative compared with 2 Tg yr^{-1} (weathering) and 1-2 Tg yr^{-1} (atmospheric deposition) estimated by Smil (2000). From the global values we calculated the average P inputs per hectare (1 and 0.25 kg ha^{-1} yr^{-1} for weathering and deposition, respectively).

For runoff we used data from Bouwman et al. (2009) who estimated that 10% of the total P input (fertilizer and manure) is lost from croplands, assuming that most of fertilizer P inputs are on flat terrains, such as river floodplains. Close to 80% of global arable land is on flat to gently undulating (<8% slopes) terrain, and close to 50% has slopes <2% (Bouwman et al., 2006; National_Geophysical_Data_Center, 2012). Since in DPPS fertilizer and manure directly enter the labile and stable pools, the calculated P loss by runoff is actually taken from the labile and stable pools in a ratio of 4:1.

3.2.3. Scenario analysis

Future needs of P fertilizer are calculated based on the target crop yields, given by four different MEA scenarios for 2050: Global Orchestration (GO), Order from Strength (OS), Techno-garden (TG) and Adapting Mosaic (AM) (Table S3-4). Here we use the total P uptake for 2050 for the reactive Global Orchestration scenario (GO) in different world regions. This scenario predicts the highest increase in global crop production among the four MEA scenarios, and the largest P uptake, i.e., 11.8 kg ha^{-1} yr^{-1} versus 10.7 in Adapting Mosaic, 10.9 in Order from Strength and 11.4 in Technogarden scenarios.

Table 3-1 Cropland area, P application, target P uptake, and cumulative P input in different world regions and the entire globe.

	Cropland (10^6 ha)		**P Application rate (Fertilizer and manure) (kg ha^{-1} yr^{-1})**			**P Uptake rate (kg ha^{-1} yr^{-1})**		**Target P uptake rate[a] (kg ha^{-1})**	**Cumulative P input (kg ha^{-1})**	
Year	**1965**	**2007**	**1965**	**Max**	**2007**	**1965**	**2007**	**2050**	**1965-2007**	**2008-2050**
Western Europe	107	94	23.8	33.8	17.2	4.9	9.9	10.4	1115	600
Eastern Europe	231	199	6.1	18.7	4.7	2.6	3.9	4.4	430	225
North America	230	225	8.7	12.4	11.4	3.9	8.8	12.7	465	630
Latin America	112	170	4.4	20.8	20.8	3.1	8.9	13.2	480	840
Asia	446	541	6.4	27.5	27.3	3.5	10	15.8	690	1130
Africa	173	247	1.9	4.4	4.1	1.8	3.1	8.3	160	580
Oceania	41	46	14.8	18.3	16	1.1	2.5	6.5	560	690
World	1390	1520	7.6	16.6	16.6	3.2	7.6	11.8	550	790

[a] Based on the Global Orchestration scenario of the Millennium Ecosystem Assessment (Alcamo et al., 2006) in a recent implementation to assess nutrient cycling (Bouwman et al., 2009).

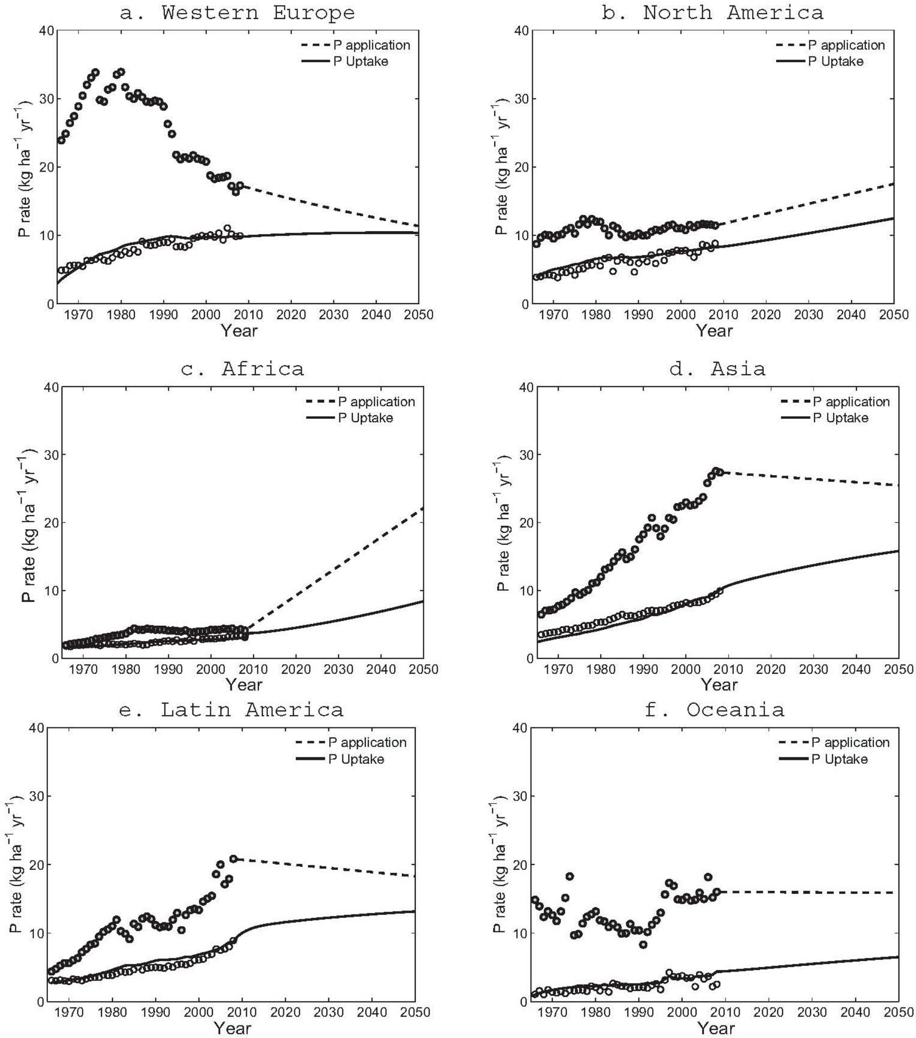


Figure 3-2 Trends of annual P application and P uptake in cropland for the period 1965 to 2050 according to GO scenario in a) Western Europe, b) North America, c) Africa, d) Asia, e) Latin America, and f) Oceania. Long-term FAO data (FAO, 2011) and simulation results are illustrated by circles and lines, respectively. Shaded and open circles refer to P application and P uptake rates, respectively. Dashed and solid lines refer to P application and P uptake rates, respectively. The regions are based on those defined by FAO (FAO, 2011). The R^2 values for calculated versus observed P uptake (1965-2007) range from 0.62 for Oceania, 0.81 for W-Europe, 0.83 for N-America, 0.84 for Africa, 0.93 for L-America, and 0.98 for Asia.

3.2.4. Sensitivity analysis

We analysed the sensitivity of DPPS to variation of fixed model parameters μ_{LS}, μ_{SL}, f, and the percentage of the total P input lost due to runoff (for more information see SI document). The implications of the uncertainty of the transfer time from the labile to the stable pool and from the stable to the labile pool are examined by increasing or decreasing the standard values by 50%.

We assessed the uncertainty of runoff loss of P by assuming larger fractions of the P inputs being lost. While the standard case is based on a global average loss rate, in many countries runoff and erosion loss may be more important. We therefore increased runoff P loss by a factor of 2 and 3 (20 and 30% of total inputs instead of the standard 10%).

The uncertainty in the fraction of total P input (mineral and manure) that transfers to labile (f) and stable pools (1-f) was assessed by assuming 90 and 70% instead of the standard 80% for f and consequently 10 and 30% for 1-f instead of the standard 20%.

3.3. Results

3.3.1. Historical P application through fertilizer and manure (1965-2007)

Past trends of fertilizer P use vary markedly across the different regions of the world. Annual P application in Western Europe was 24 kg P ha^{-1} in 1965, peaked at 34 kg ha^{-1} in the 1980s and then gradually decreased to 17 kg P ha^{-1} in 2007 (Figure 3-2a). Oceania's annual P fertilizer use fluctuated between 8 and 18 kg ha^{-1}, with rates in early 2000s returning to those of 1965 (Figure 3-2f). Annual application rates in North America remained fairly constant around 9-12 kg P ha^{-1} from 1965 to 2007 (Figure 3-2b), while the rates of P used rose rapidly from 4 to 21 kg ha^{-1} in Latin America (including Central and South America) and from 6 to 28 kg ha^{-1} in Asia (Figure 3-2 e, d). In Africa, annual P application rates remained low (<4.5 kg ha^{-1}) throughout the entire period (Figure 3-2c). North America, Western Europe, East and South Asia accounted for over 80% of global P fertilizer use in 1979-1999 (Bruinsma, 2003), despite comprising less than 55% of global cropland.

Aggregated global P application in croplands in the form of fertilizer and manure from 1965 to 2007 show a sharp increase in global fertilizer application between 1965 and 1990, while global manure P application in croplands was more stable in that period and increased gradually after 1990 (Figure 3-3a). At the continental and regional scale these trends are completely different. In Africa annual applications of inorganic P fertilizer and manure P in 1965 are the same (0.2 Tg yr^{-1} for each). While annual inorganic P fertilizer use doubled (up to 0.4 Tg), manure P showed a three-fold increase (reaching to 0.6 Tg) between 1965 and 2007. In Asia annual fertilizer P application increased 12.5-fold (from 0.8 Tg in 1965 to 10 Tg in 2007) and manure P more than doubled from 2 to 4.5 Tg in the same period.

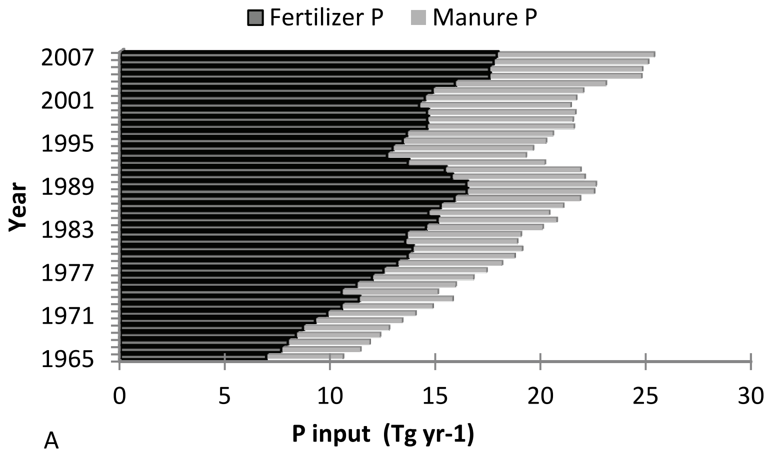
3.3.2. Cumulative P application and uptake between 1965 and 2007

Cumulative inputs of P fertilizer and manure (Bouwman et al., 2009; FAO, 2011) in Oceania (560 kg P) and Western Europe (1115 kg P) per hectare of cropland for the period 1965-2007 were much greater than the cumulative crop P uptake (100 and 350 kg P $ha^{-1}$for Oceania and W-Europe, respectively). Over the same period, cumulative P input in Asia was close to 700 kg P ha^{-1}, 500 kg P ha^{-1} in North America, Eastern Europe and Latin America, but only 160 kg P ha^{-1} in Africa. The cumulative P uptake was much smaller in Africa (105 kg ha^{-1} P uptake) than in Asia, North and Latin America (250 kg P ha^{-1} uptake). At the global scale, less than half of the applied P between 1965 to 2007 (550 kg P ha^{-1}) was taken up by harvested crops (225 kg P ha^{-1}).

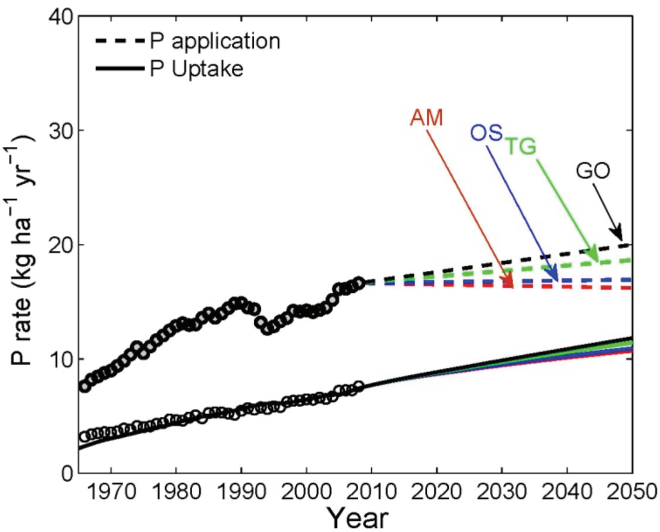
3.3.3. P demand between 2008 and 2050

We used the DPPS model to calculate P application rates (fertilizer and manure) based on target crop production from the Global Orchestration (GO) scenario of the Millennium Ecosystem Assessment (Alcamo et al., 2006) (Table 3-1). The GO scenario projects a rapid increase in annual crop production and P uptake in Africa (from 3.1 to 8.3 kg P ha^{-1}) and Oceania (from 2.5 to 6.5 kg P ha^{-1}) between 2008 and 2050, while in Western Europe annual P uptake will increase by only 5% (from 9.9 in 2007 to 10.4 kg P ha^{-1} yr^{-1} by 2050). With a faster increase in crop production (13%) in Eastern Europe, P target uptake rate will reach 4.4 kg P ha^{-1} in 2050 (Figure S3-6). In North America, Latin America and Asia, P uptake needs to increase by 40-60% between 2007 and 2050.

The DPPS model simulates the historical patterns of uptake as a response to the application rates remarkably well in all continents (Figure 3-2). R^2 values for calculated versus observed P uptake (1965-2007) for all continents and the world range between 0.81 and 0.98, and 0.62 for Oceania. During the past decades, Oceania experienced prolonged drought periods resulting in important fluctuations in crop production (Figure 3-2f), which is probably the reason for the poorer fit in Oceania than in other continents. Based on the DPPS model, we estimate that to achieve the 2050 target production, the cumulative P application between 2008 and 2050 amounts to 1130 kg P ha^{-1} in Asia (Figure 3-2d), 840 kg P ha^{-1} in Latin America, 690 kg P ha^{-1} in Oceania and 630 kg P ha^{-1} in North America.



A



B

Figure 3-3 A) Global annual applications of inorganic P fertilizer and manure P between 1965 and 2007. B) Trends of annual P application and P uptake in cropland for the period 1965 to 2050 in the entire globe according to the four MEA scenarios. Long-term FAO data (FAO, 2011) and simulation results are illustrated by circles and lines, respectively. Shaded and open circles refer to P application and P uptake rates, respectively. Dashed and solid lines refer to P application and P uptake rates, respectively. The R^2 value for calculated versus observed P uptake (1965-2007) is 0.96.

Compared with these continents, the cumulative P inputs are smaller in Western Europe (600kg P ha^{-1}) and Africa (580 kg P ha^{-1}) (Figure 3-2). The GO scenario shows a 55% increase in target P uptake rate in global agricultural systems from 2007 to 2050, from 7.6 to 11.8 kg P ha^{-1} yr^{-1}, resulting in a global cumulative P input around 790 kg P ha^{-1} between 2008 and 2050. The calculated global cumulative P input between 2008 and 2050 for the other MEA scenarios is 700 (AM scenario), 720 (OS) and 760 (TG) kg ha^{-1} for a target P uptake of 10.7, 10.9 and 11.4 kg ha^{-1} yr^{-1}, respectively (Figure 3-3b).

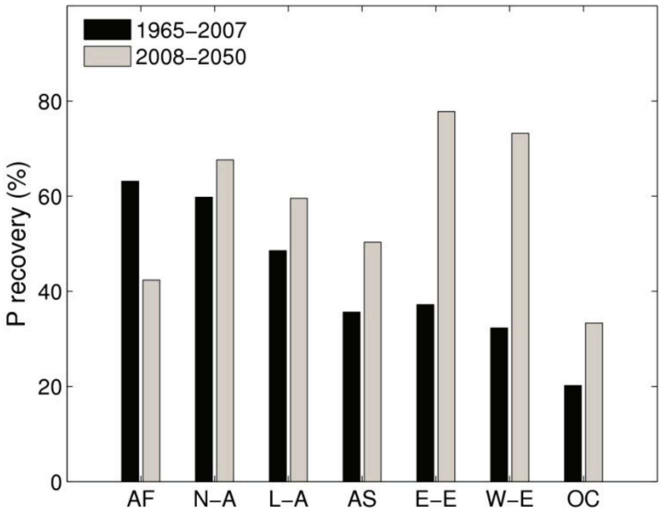
3.3.4. Sensitivity analysis

When assuming P runoff to be 20 or 30% of P input (instead of 10% as a default value) cumulative P application between 2008-2050 was, respectively 2-9% and 4-18% higher in different continents and the entire globe.

The full range from the sensitivity analyses (including transfer times, runoff loss and fraction of input entering the labile pool) for the required P input (2008-2050) in global croplands under the GO scenario is 1000 to 1440 Tg. In this case the total mineral P fertilizer, required in the year 2050 will range between 14.6 and 28 Tg (around the standard 20.8). Even the highest projections are still 20% less than future estimates found in the literature (Bouwman et al., 2009; Bouwman et al., 2011).

3.3.5. Phosphorus recovery (efficiency)

Accounting for the contribution of residual P in future P demand, the average of P recovery - defined as P in the crop yield over the P input - between 2008 and 2050 shows an increase relative to P recoveries calculated for 1965-2007 in all world regions (except for Africa) (Figure 3-4a). Global P recovery shows a steady increase from 30% in 1965 to close to 40-50% in the period 2000-2007 and close to 60% in 2050 (Figure 3-4b).



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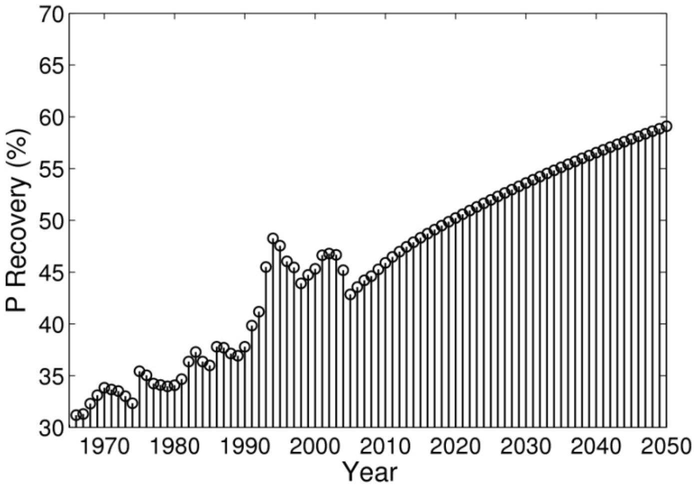

B

Figure 3-4 A) P recovery from actual historical data (1965-2007) and future simulated data (2008-2050). AF, Africa; N-A, North America; L-A, Latin America; AS, Asia; E-E, East Europe; W-E, West Europe and OC, Oceania. B) Modeled global P recovery from 1965 till 2050.

3.4. Discussion

3.4.1. Residual P and hysteretic crop uptake

Our analysis shows that including residual P in the estimation of the fertilizer required to achieve the target crop yields leads to a reduced fertilizer requirement compared with other studies that did not account for residual P. In most world regions, soil P status has been improved over the past decades by applying P fertilizer and manure. For instance, since the 1980s, P application rates have declined in many European countries while uptake continued to increase. This is possibly due to the continued supply of plant available P from the residual soil P pool (Nuruzzaman M, 2005).

Obviously the residual P pool must be maintained, so the current decreasing trend simulated for Europe (Figure 3-2a) needs to be balanced by P input equal to crop removal (Figure S3-7). By contrast, in many developing countries in Africa, Asia and Latin America, soils have been continually depleted over the years due to the low rate of P input. A global P balance study recently reported that almost 30% of global cropland area showed P deficiency in 2000 (MacDonald et al., 2011).

Our results show a hysteretic behaviour in crop P uptake versus fertilizer application. This behaviour is well pronounced in regions with large P applications (e.g. in Western Europe, Figure 3-5), where for the same P application rate, two different P uptake rates are observed at different points in time. The high uptake rate is the result of the residual P that has accumulated in the soil. Build-up of soil P fertility as a result of substantial past inputs of organic and mineral P fertilizer in several world regions has reduced the P inputs required, although P uptake by crops has stabilized or increased. This has also been observed at smaller scales. Experimental data at field, farm and country scales (Kamprath, 1967; Mishima et al., 2010; van Keulen et al., 2000; Verloop et al., 2010) support our findings that when soil available P is adequate, crop yields can increase with decreasing or even halting P application. For example a farm-scale study showed that long-term (1989-2006) equilibrium P fertilization did not lead to reduced crop yields in Dutch sandy soils (Verloop et al., 2010). A recent national-level study on P fertilizer in Japan indicated that crop yields remained constant or even increased, despite a decline in use of P fertilizer and manure between 1985-2005 (Mishima et al., 2010). Even in high P-fixing soils a large initial application of P (around 600 kg P ha^{-1}) can be adequate for cultivating maize for 7 to 9 years due to the effect of residual P (Kamprath, 1967).

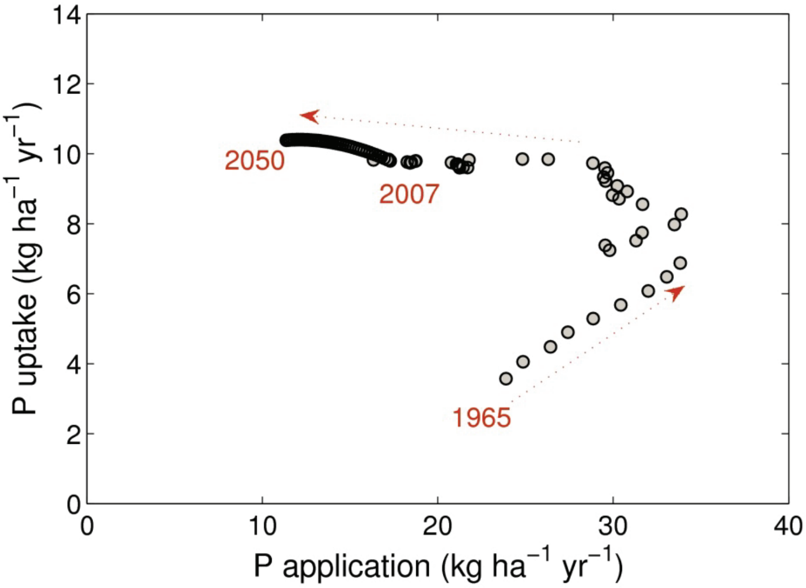

Figure 3-5 Hysteresis in P uptake versus P application between 1965 and 2007 (Western Europe). With the same amount of P application, there are two rates of P uptake that show the contribution of the residual P to the crop production.

3.4.2. Projection of future P requirement

There are only a few global studies that estimate the amount of P input – output in croplands in 2050 (Bouwman et al., 2009; Bouwman et al., 2011; Steen, 1998). Steen (1998) estimated a 2.5% annual growth in P consumption over the long term due to the 2-2.5% increase in crop yield per year. With this rapid growth, global annual consumption would be 26-31 Tg P in 2050. Total P use in 2050 calculated by Bouwman et al. (Bouwman et al., 2009; Bouwman et al., 2011) ranges between 23 and 33 Tg yr^{-1}, depending on the scenario for crop production and P use efficiency. These projections ignored the effect of residual soil P.

Our results show that accounting for the role of residual soil P leads to lower projections of P demand for the period 2008-2050. Global P application rates per hectare will increase, but less than proportional to the required increases in production and yield (Table 3-1). However, the situation differs among continents. In Europe, Asia, Latin America and Oceania crop production can benefit from the residual P accumulated due to past P fertilizer and manure use (Table 3-1), and P uptake can increase even with a reduction in P application rates between 2008 and 2050. In contrast, target P uptake rates can be achieved in North America with a slight increase in P application rates.

Due to the minimal P application rates in the past decades, in Africa more than five-fold increase from 4 kg P ha^{-1} in 2007 to ca 23 kg ha^{-1} in 2050 is needed to achieve the target P uptake. This result is consistent with conclusions of Steen (Steen, 1998) who suggested that 30-50% more P fertilizer than crop requirements must be applied for almost 30-50 years to restore soil P in depleted soils.

Our estimate of the global P input required in global cropland of 1200 million tonnes P for the 2008-2050 period includes P from both mineral fertilizer and animal manure. The total amount of P use in 2050 is smaller than recent projections in all regions except Africa (Bouwman et al., 2011). Accounting for the relative contribution of P from animal manure in global croplands under the GO scenario (32%)(Bouwman et al., 2009), global inorganic P fertilizer use must be 20.8 Tg in 2050. This is 10-40% less than estimated in other studies (Bouwman et al., 2009; Bouwman et al., 2011; Steen, 1998). The demand is even less for the other scenarios (Figure 3-3b). However, a large part of the P in animal manure that is recycled in cropland originates in grasslands. This transfer of P from grasslands to cropland is particularly important in developing countries (Bouwman et al., 2009), and contributes to the build-up of residual soil P in cropland. Given the increasing future demand for grass (Bouwman et al., 2005b), additional fertilizer P will be required to maintain soil fertility in the world's soils under grassland.

Our results suggest that residual soil P can contribute to crop production with a considerable lag time. The model results provide important information on where and how much P is needed to achieve food security in 2050. Since more than 80% of global P reserves is used in agricultural products (Smil, 2000), our results provide a firm basis to analyse depletion of global P reserves.

3.5. Supplementary Information

The data used in this work are available at
http://models.pps.wur.nl/content/data-residual-soil-phosphorus

3.5.1. SI Material and Methods

3.5.1.1. Data used

Annual P fertilizer consumption, harvested crop yield and areas of arable crops (1965-2007) were obtained from FAO(FAO, 2011), including Western Europe, Eastern Europe, North America, Latin America, Asia, Africa, Oceania and the world. Eastern Europe includes Belarus, Bulgaria, Czech Republic, Hungary, Poland, Republic of Moldova, Romania, Russian Federation, Slovakia and Ukraine; Western Europe is the rest of Europe; Latin America includes South America, Central America (including Mexico) and the Caribbean; North America is Canada and United States of America; Asia excludes the Russian Federation (part of Eastern Europe).

Globally 161 different crops were distinguished including 150 crops in Asia, 142 in Africa, 124 in North and Latin America, 120 in Europe and 110 in Oceania.

3.5.1.2. Estimation of Manure

The global amount of manure production is estimated to be between 15 to 24 Tg (Bouwman et al., 2009; Potter et al., 2010; Sheldrick and Lingard, 2004; Smil, 2000). All of these studies estimate the potential nutrients available from manure production. Here we used estimates of P in manure spreading in cropland lands, which is much less than total manure P production including grazing systems.

In most industrialized countries, 50% of the available animal manure from storage systems is assumed to be applied to cropland and the remainder to grassland (Lee et al., 1997). In most developing countries, 95% of the available manure is assumed to be applied to cropland and 5% to grassland, thus accounting for stubble grazing and manure excretion in croplands, and the lower economic importance of grass compared with crops in developing countries (Seré and Steinfeld, 1996). This system implies that where grazing animals are confined part of the time, and the manure collected during that period is spread in cropland, there is a transfer of P from grassland to cropland. Such a transfer is particularly important in many developing countries.

For EU countries we used maximum application rates of manure equivalent to 170-250 kg N ha^{-1} yr^{-1} based on existing regulations; since the amounts of N and P from manure are related (depending on the animal category from which the manure stems) this also leads to maximum P rates. Data for manure P inputs to cropland 1950, 1970 and 2000 (Bouwman et al., 2009) were the starting point. First, a linear change was assumed to obtain estimates for 1950-1960, 1960-1970 and 1970-2000. Years in between 1960 and 1970, 1970 and 2000 and between 2000 and 2007 were obtained by applying the annual variability of total manure P production calculated from total animal stocks for dairy cattle, non-dairy cattle, pigs, poultry and sheep and goats from FAO (FAO, 2011) and P excretion rates (Bouwman et al., 2009; Bouwman et al., 2011).

3.5.1.3. Parameter values used for the simulations

Parameters include the fraction of P inputs allocated to P_L or P_S, μ_{LS} and μ_{SL}, initial P uptake by fertilized and unfertilized crops during the first year of fertilizer application, weathering and deposition (Figure 3-1). The fractions of P inputs allocated to P_L or P_S depend on fertilizer type. However, we assume that all P inputs behave as superphosphate, with the fraction of P entering P_L (80%) and the P_S (20%)(Wolf et al., 1987). In all cases, the μ_{LS} was set to 0.2yr^{-1} (Janssen et al., 1987). However, a different μ_{SL} is considered in different continents depending on the major soil type for that continent. In most tropical and subtropical regions such as Africa, Oceania and Latin America, μ_{SL} is 0.03 yr^{-1}(Janssen et al., 1987), while in other regions μ_{SL} is 0.04 yr^{-1}.

The model uses initial crop P uptake from fertilized α and unfertilized soils β. In each continent the calculated P uptake(FAO, 2011) for 1965 is chosen as initial uptake value for fertilized soil. Assuming initial recovery of P fertilizer equal to 10% (Smaling and Janssen, 1993), the initial P uptake of unfertilized soil can be calculated by subtracting 10% of P input (mineral and manure P fertilizer) from the initial P uptake of fertilized soil. In Figure 3-2; α and β are respectively: a) 4.0; 2.0 b) 3.2; 2.5 c) 1.65; 1.5 d) 3.0; 2.5 e) 2.8; 2.3 f) 1.15; 0.06 and g) 2.9; 2.3.

3.5.1.4. Sensitivity analysis

The sensitivity of DPPS was tested by using a range of ±50% around the standard values of μ_{LS} and μ_{SL} (Table S3-2) and assuming larger P losses by runoff (20% and 30% of P inputs compared with 10% in the standard case) (Table S3-3). When μ_{LS} has a larger transfer rate from the labile to the stable pool (0.4 yr^{-1} versus the standard 0.2 yr^{-1}) and μ_{SL} has a smaller transfer rate from the stable to the labile pool (0.025 yr^{-1} versus the standard 0.04 yr^{-1}), the plant available P will be less than in the standard case. As a consequence, the required global cumulative P input is 1440 Tg P between 2008 and 2050 for the GO scenario, compared with 1200 Tg in the standard case. Plant available P will be higher than in the standard case by assuming a larger value for μ_{SL} (0.08 yr^{-1})

and a smaller value for μ_{LS} (0.1 yr^{-1}), resulting in a cumulative P input of 1070 Tg for 2008-2050 under the GO scenario.

Varying the fraction of input to the labile pool around the standard value leads to -16 to +4 % variation in the global P demand for 2008-2050 (Table S3-2).

Table S3-2 Sensitivity of the DPPS model to variation in μ_{LS}, μ_{SL} and f.

Model Case	**Model Parameters**			**Total fertilizer 2008-2050**	
	μ_{LS} (y^{-1})	μ_{SL} (y^{-1})	f	kg P ha^{-1}	Tg P
Standard μ, f	0.2	0.04	0.8	790	1200
Variable μ	0.4	0.025	0.8	950	1440
	0.1	0.08	0.8	700	1070
Variable f	0.2	0.04	0.9	660	1000
	0.2	0.04	0.7	820	1240

Increasing runoff and consequently decreasing the residual P in soil leads to larger required cumulative P application between 2008 and 2050 to meet the target P uptake in 2050 (1280 Tg P for 20% runoff loss, and 1380 Tg P for 30% loss, compared with 1200 Tg P in the standard case of 10% runoff loss) (Table S3-3).

Table S3-3 Sensitivity of the DPPS model to variation of P loss by runoff.

	Cumulative P input, 2008-2050					
	Standard Runoff (10% total P input)		**Runoff (20% total input)**		**Runoff (30% total input)**	
Region	kg ha^{-1}	Tg	kg ha^{-1}	Tg	kg ha^{-1}	Tg
Global	790	1200	840	1280	910	1380
W-Europe	600	56	660	62	730	68
N-America	630	142	680	155	750	168
Africa	580	142	590	144	630	154
Asia	1130	615	1200	650	1300	700
Oceania	690	32	710	33	720	34
L-America	840	109	890	115	1000	124

Regarding the results of the sensitivity analysis, increasing the fraction of input to the stable pool (1-f) leads to a smaller deviation from the standard case than decreasing this fraction. This is due to the long transfer time of P from the stable to the labile pool.

The full range resulting from the sensitivity analyses (including transfer times, runoff loss and fraction of input entering the labile pool) for the required P input in global croplands under the GO scenario is 1000 to 1440 Tg in the period 2008-2050. In this case the total mineral P fertilizer required in 2050 will range between 14.6 and 28 Tg (around the standard 20.8).

3.5.2. Additional analysis of P application and uptake in Europe

3.5.2.1. Eastern Europe

Cropland in Eastern Europe showed a 30% decrease in 1992 in the FAO statistics. Since the area reduction cannot be reliably included in the model, only area increase has been included. We assumed (Figure S3-6), that the change of the cropland area in Eastern Europe does not play a major role.

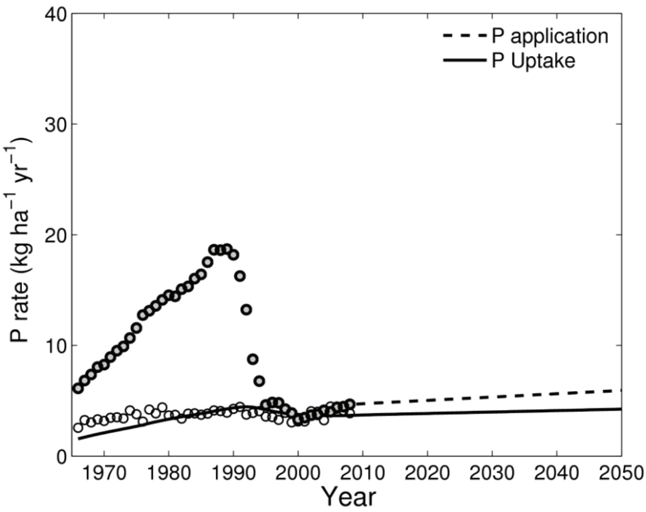

Figure S3-6 Trends of P application and P uptake for the period 1965 to 2050 in Eastern Europe assuming constant cropland area after 2007. Circles and lines illustrate long-term FAO data and simulation results, respectively. Shaded and open circles refer to P application and P uptake rates, respectively. Dotted and solid lines refer to P application and P uptake rates, respectively.

3.5.2.2. Western Europe in 2100

As mentioned before, to maintain soil productivity the trend of gradual depletion of residual P stock during 2008-2050 (Figure 3-2a) in Western Europe should be followed by input rates equal to or exceeding crop uptake (P-equilibrium fertilization). A simulation for a longer time horizon (until 2100) shows this argument as shown in Figure S3-7.

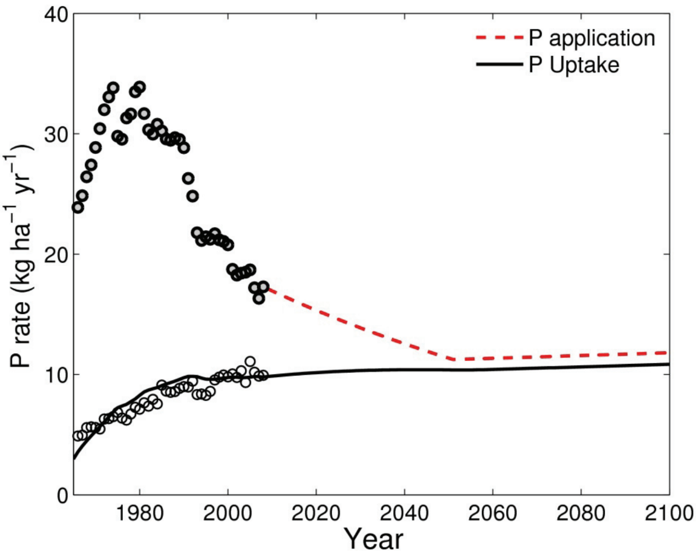

Figure S3-7 Trends of P application and P uptake for the period 1965–2100 in Western Europe. Circles and lines illustrate long-term FAO data and simulation results, respectively, as in Figure S3-6

3.5.2.3. Description of MEA scenarios

Table S3-4 Description of Millennium Ecosystem Assessment scenarios.

	Global Orchestration (GO)	**Order from Strength (OS)**	**Technogarden (TG)**	**Adapting Mosaic (AM)**
Brief description	Globalization, economic development, reactive approach to environmental problems	Regionalization, fragmentation security, reactive approach to environmental problems	Globalization, environmental technology, proactive approach to environmental problems	Regionalization, local ecological management with simple technology, proactive approach to environmental problems
General trends	Low	High	Medium	High
World population (billion)	2000: 6.1 2050: 8.2	2000: 6.1 2050: 9.7	2000: 6.1 2050: 8.9	2000: 6.1 2050: 9.6
Income	High	Low	High	Medium
global GHG emissions	High	High	Low	Medium

Table S3-4 Continued.

	Global Orchestration (GO)	**Order from Strength (OS)**	**Technogarden (TG)**	**Adapting Mosaic (AM)**
Global mean temperature increase	High	High	Low	Medium
Per capita food consumption	High, high meat	Low	High, low meat	Low, low meat
Agricultural trends				
Productivity increase	High	Low	Medium-high	Medium
Biofuels	4% of cropland area in 2050	1% of cropland area in 2050	28% of cropland area in 2050	2% of cropland area in 2050
Fertilizer use and efficiency	No change in countries with a surplus; rapid increase in N and P fertilizer use in countries with soil nutrient depletion (deficit)	No change in countries with a surplus; slow increase in N and P fertilizer use in countries with soil nutrient depletion (deficit)	Rapid increase in countries with a surplus; rapid increase in N and P fertilizer use in countries with soil nutrient depletion (deficit)	Moderate increase in countries with a surplus; slow increase in N and P fertilizer use in countries with soil nutrient depletion (deficit); better integration of animal manure and re-cycling of human N and P from households with improved sanitation but lacking a sewage connection.

Table adapted from (Bouwman et al., 2009).

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4

Massive transfer of phosphorus to croplands threatens sustainability of grasslands

Sattari, S. Z., Martinez Rodríguez, R., Van Ittersum, M. K., Beusen, A. H. W., and Bouwman, A. F. (2014). Massive transfer of phosphorus to croplands threatens sustainability of grasslands. To be submitted.

Population growth will increase food demand and economic growth is expected to cause a shift towards more meat and milk in human diets. Future demand of meat and milk will increase the pressure on grasslands to provide grass and fodder for livestock. That implies the relevance of maintaining fertility of grassland soils. Here we show that a large part of the phosphorus (P) in animal manure that originates from grassland is spread in cropland, while up to today very little mineral fertilizer has been applied in grasslands in most of the world's regions. We estimate that over the 1970-2005 period, globally only 63% (332 Tg) of manure remained within the grasslands and the rest left the grasslands for spreading in croplands as organic fertilizer as well as other uses, resulting in a very significant loss of soil fertility in grasslands. To avoid expansion of grassland areas that would lead to an increase in deforestation, productivity of existing grassland must be increased which will only be possible if the P status of grassland's soils is improved. We applied a two-pool soil P model to estimate P requirements for grass production towards 2050. To maintain the soil P status in grasslands and meet the target grass production in 2050 as estimated by the Rio+20 scenario, the cumulative amount of P application through manure and mineral fertilizer has to reach ca.1215 Tg P between 2005 and 2050. Added to the amount needed in the period 2008-2050 for croplands (1200 Tg P), a total amount of 1380 Tg mineral P is needed, corresponding to 10,700 Tg phosphate rock. We argue that a massive transfer of fertility is taking place from grasslands to croplands, and that better manure management, complemented with mineral fertilizer is crucial for the sustainability of the world's grasslands.

4.1. Introduction

Global food demand is expected to rise rapidly in the coming decades with a shift towards more meat and milk in human diets. Currently, meat consumption per capita in United States and Europe exceeds the global average by a factor of two (Westhoek et al., 2011) and global production of meat and milk are projected to increase two times in 2050 (Steinfeld et al., 2006).

Grasslands, a major part of the global agro-ecosystem, are a significant contributor to world's food security, primarily by supplying proteins, energy and fiber to ruminants. The global permanent pasture area is covering 26% of the land area. In addition, 33% of the global arable land is dedicated to feed crop production (Steinfeld et al., 2006). Historically, human populations have converted the most suitable land into arable fields, leaving more marginal lands to pastures. Much of the world's grasslands is in poor condition due to the overgrazing and consequent soil erosion, weed encroachment and desertification (Suttie et al., 2005). These problems are likely to worsen given that livestock production is projected to increase while global grassland area will remain almost constant (Bouwman et al., 2005). Grasslands have, in addition to their socio-economic importance, a relevant role in the context of climate change and its mitigation. Pastures, rangelands and dry lands provide key ecosystem services such as food and fiber production, water cycle regulation and carbon sequestration (Lal et al., 2007; O'Mara, 2012; Smith et al., 2007). These services may be threatened unless management systems change to halt overgrazing and land degradation. In addition, livestock is the most important driver of anthropogenic acceleration of N and P cycles, primarily due to the low nutrient conversion efficiency (Bouwman et al., 2013).

A large part of the P in animal manure that is applied in cropland actually originates from grassland (Sattari et al., 2012). The transfer of nutrients from grassland to cropland is particularly important in developing countries, where it helps to satisfy crop requirements and to build-up residual soil P in cropland areas. Yet, to date there is no comprehensive global study that quantifies the historical soil P budget for pastures and the P transfers between grasslands and croplands.

We calculated the P dynamics of P budgets in grassland, which is an essential step in development of global strategies to cope with the global P crisis. Then we estimated the future P requirements of grassland and of grassland plus cropland.

We distinguish two categories of grassland, i.e. grasslands in mixed and landless livestock production systems (which we group under the term intensive systems) and in pastoral systems. A general conceptual framework was elaborated that focuses on agricultural grassland systems including the key P inflows, outflows and distinguishing four compartments within the grassland system boundary: grassland-based livestock population, grassland-based livestock manure, soil and grass (Figure 4-1,Table S4-2).

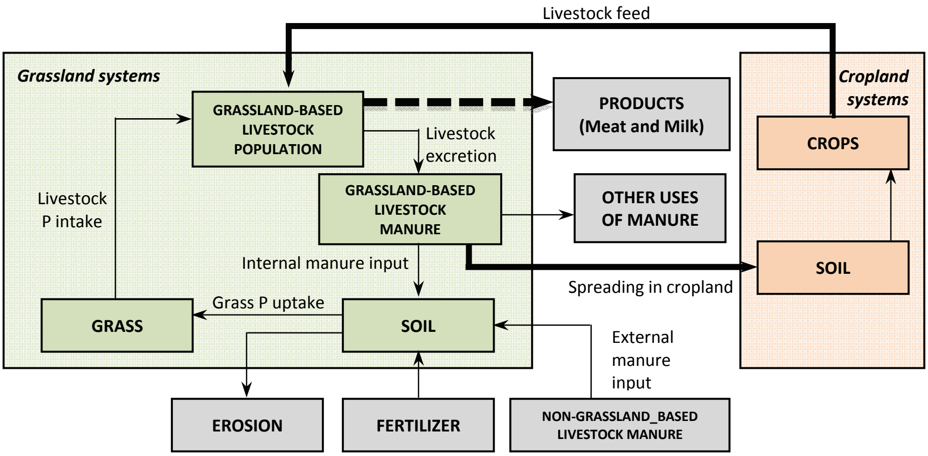


Figure 4-1 General scheme of the model. The model comprises four compartments within grassland systems (grassland based livestock population, grassland based livestock manure, Soil and Grass) and five compartments outside grassland boundaries (Products, Other uses of manure, Erosion, Fertilizer and Non-grassland based livestock manure). P transfers between grasslands and croplands are represented by *Livestock Feed* and *Spreading in cropland* flows (thick arrows). The thick, dotted arrow represents the P flow leaving the grassland systems via animal products, mainly meat and milk. Most relevant P flows from, to and within grasslands are described in Table S4-2.

For calculation of historical soil P budgets for soils under grassland, internal and external manure spreading in grassland and mineral fertilizer P are considered as inflows and grass uptake and erosion are regarded as outflows. We then calculate the ruminant production system and transfer of P between croplands and grasslands over the period 1970-2005. Phosphorus is imported through livestock feed from cropland to grassland, while manure is exported from grassland to cropland.

Using the DPPS –Dynamic Phosphorus Pool Simulator- model we calculated the amount of P needed for meeting the global grass requirement in intensive and pastoral systems according to the Rio+20 scenario (van Vuuren et al., 2012) until 2050. Finally the global P demand, comprising crop and grass production is projected for the next 45 years.

4.2. Material and Methods

4.2.1. Model description

The Dynamic Phosphorus Pool Simulator (DPPS) was used -a simple two-pool P-model (Wolf et al., 1987) including labile and stable pools and long-term P input and output data (Sattari et al., 2012). The DPPS model reproduces historical grass uptake as a function of P inputs (fertilizer and manure). This model considers the essential P fluxes between grass and soil. It includes both labile and stable pools of phosphorus with a yearly time step. The model can calculate P transfer between different pools, the P uptake by grass, and the pool sizes. The model can also be applied for calculating the fertilizer requirement for a future target yield. The DPPS model successfully simulated the historical patterns of crop P uptake as a response to the application rates in all continents and the entire globe as shown by Sattari et al. (2012). The model was used to calculate future P fertilizer and manure application rates in grasslands based on target grass productions in 2050 derived from the Rio+20 Trend or baseline scenarios (van Vuuren et al., 2012). Details of the model, and its application to estimate future P requirements are given in Sattari et al. (2012) and Wolf et al. (1987).

4.2.2. Data Used

Total manure production within pastoral and intensive systems was computed from animal stocks and P excretion rates. We used P excretion rates per head for dairy and nondairy cattle, buffaloes, sheep and goats, pigs, poultry, horses, asses, mules and camels based on various sources (Service, 1985; Sheldrick et al., 2003; Smith, 1991; Van Horn et al., 1996; Wilkerson et al., 1997). We used constant excretion rates per head, so that the P excretion per unit of product decreases with increasing milk and meat production per animal. For each country, animal stocks and P in the manure for each animal category were spatially allocated across intensive and pastoral systems. For the period 2005-2050, the distribution over these systems is provided by the Rio+20 study (van Vuuren et al., 2012).

Liu et al. (2008) assumed that cropland's weathering and atmospheric deposition is a fraction of the total land's, based on its area fraction of the total land. Employing the same approach, we have estimated about 4.0 $TgPyr^{-1}$ as weathering and atmospheric deposition in grasslands.

Phosphorus inputs to the grassland and grassland's soil systems are animal feed, grassland-based and non-grassland-based manure and mineral P fertilizers (Figure 4-1). Phosphorus outputs from the grasslands include milk, meat and livestock by-products, manure application to cropland and the other use of manure. Livestock by-products include adipose tissue, skeleton, viscera, blood, skin, hair and digestive content. Phosphorus outputs from grassland soils include grass uptake and soil erosion (Figure 4-1). More details on livestock production systems, animal categories, the manure calculations and manure use as well as all the grassland P inflows and P outflows calculations are provided in the SI Material and Methods.

4.2.3. Scenario for the period 2005-2050

We used the Rio+20 (van Vuuren et al., 2012) Trend scenario for simulating future P requirement in grasslands. Similar to the baseline scenario of the Environmental Outlook of the Organization for Economic Cooperation and Development (OECD) (OECD, 2012), the Global Orchestration scenario of the Millennium Ecosystem Assessment (Alcamo et al., 2006) and the A1 scenario of IPCC-SRES (Nakicenovic et al., 2000), the Rio+20 Trend scenario is a baseline or business-as-usual scenario with similar assumptions on population growth and economic development pathways. Baseline scenarios such as the Rio+20 Trend scenario represent a continuation of current trends, with no dramatic changes or shifts in production and management systems and attitude towards environmental problems. Apart from the Trend scenario, the Rio+20 study (van Vuuren et al., 2012) describes three challenge pathways, which were designed to assess the potential to achieve sustainability goals. In the Rio+20 Trend scenario, the world population is projected to grow from 7 to 9 billion people during the period 2010-2050.

4.3. Results

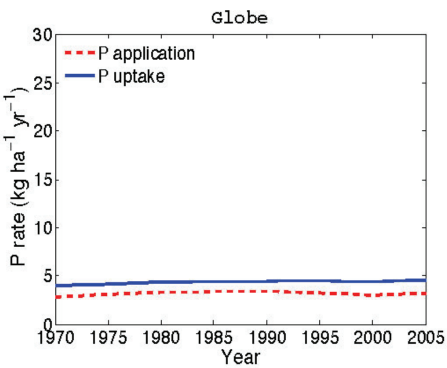
The results, obtained at the country scale, were aggregated to world regions (Africa, Asia, Eastern Europe, Latin America, North America, Oceania and Western Europe,), as well as globally.

4.3.1. Grassland soil P budget resulting from P application and grass P uptake

The historical rates of P application (manure plus fertilizer) and grass P uptake in soils under grassland were relatively steady for the period 1970-2005 in most regions of the world (around or less than 5 kg ha^{-1} yr^{-1}) except for Europe (Figure 4-2). The sharp decline in both P application and uptake in the region Eastern Europe in the early 1990s reflects the collapse of the USSR and political changes in Eastern European countries (D), when the use of fertilizer and animal numbers were severely reduced. In Western Europe, the rate of P application declined after 1980 in response to the increasing awareness of environmental problems associated with oversupply of nutrients in several countries and emerging European legislation (H). The P application through fertilizer and manure and P uptake were in balance in North America, and P uptake in grasslands exceeded P application in Africa, Asia, Latin America and Oceania and the entire globe between 1970 and 2005. In Eastern and Western Europe, P application exceeded uptake due to widespread and intensive use of mineral fertilizers compared with other regions.

Annual P application through mineral fertilizer was systematically lower than manure application in all regions (Table 4-1). Annual use of mineral fertilizer in Africa, Asia, Latin America and Oceania was negligible (≤0.1 kg P ha^{-1} yr^{-1}) throughout the entire period. Annual use of fertilizer in Eastern Europe peaked at 4.0 kg P ha^{-1} yr^{-1} in 1990, then drastically dropped in the next 7 years to levels prevalent in the 1970s and then decreased slowly to 0.1 kg P ha^{-1} yr^{-1} in 2005. In Western Europe P fertilizer rate was 3.2 kg P ha^{-1} yr^{-1} in 1970, had its maximum at 6.1 kg P ha^{-1} yr^{-1} in 1990 and then decreased gradually to 1.7 kg P ha^{-1} yr^{-1} in 2005 (Table 4-1).

Manure was the most important source of P for grassland soils worldwide (Table 4-1). Globally grassland soils received 3.0 kg P ha^{-1} yr^{-1} on average between 1970 and 2005. Africa´s annual P application through manure grew steadily from 1.6 kg P ha^{-1} yr^{-1} in 1970 to 2.6 kg P ha^{-1} yr^{-1} in 2005. In Africa the maximum contribution of manure from non-grassland-based animals to the total manure application was 0.4% in 2005. In Asia, manure application was 1.8 kg P ha^{-1} yr^{-1} in 1970, and increased to its peak in 2005 with an annual application of 2.4 kg P ha^{-1}. Relative contribution from non-grassland-based animals for Asia was 5% on average, with a maximum share of 8% in 2005.



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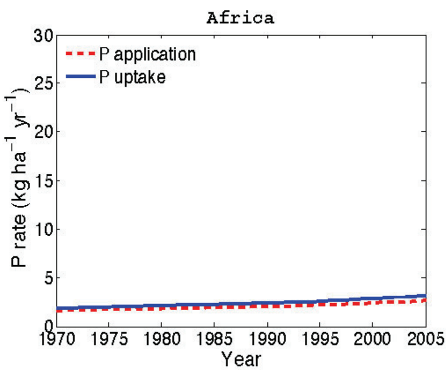

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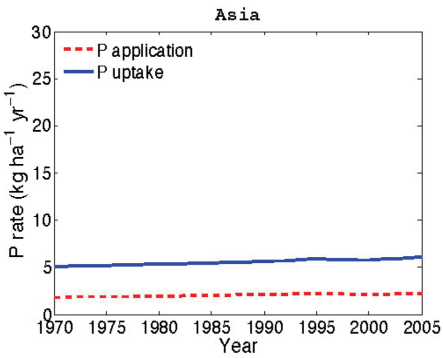

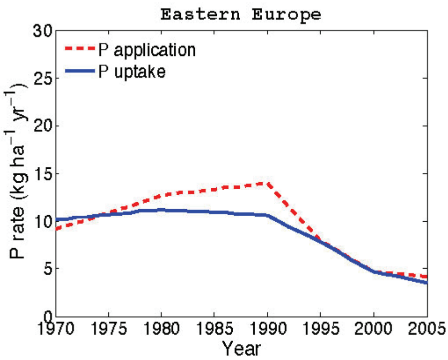

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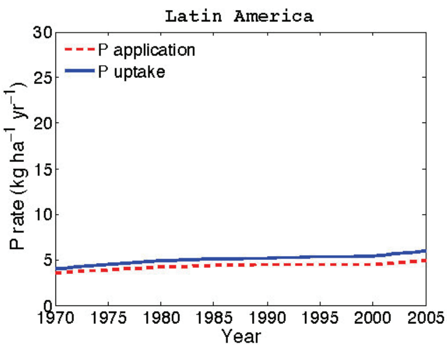

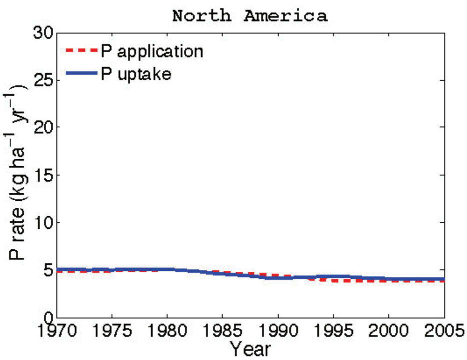

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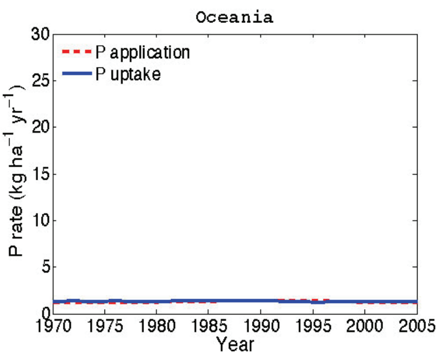

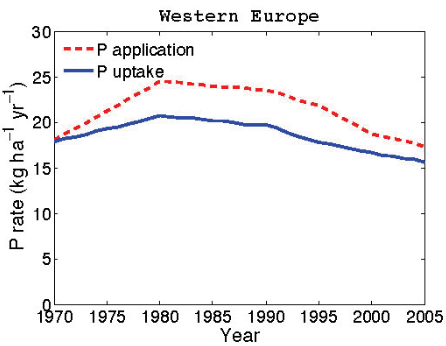

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H

Figure 4-2 Historical trends of annual P application (manure plus fertilizer) and P uptake in grassland systems (intensive and pastoral) for the period 1970-2005 in (A) World, (B) Africa, (C) Asia, (D) Eastern Europe, (E) Latin America, (F) North America, (G) Oceania and (H) Western Europe. These regions were also used in (Sattari et al., 2012). Dashed and solid lines represent P application and P uptake, respectively.

The importance of manure from non- grassland-based animals in both Eastern and Western Europe is significantly higher than for the other regions. In Eastern Europe, 15% of the total P applied to grasslands through manure came from non- grassland-based animals in 2005, while the maximum share in Western Europe was 16% in the same year.

The total manure application rate for Eastern Europe changed from 8.7 kgPha^{-1} yr^{-1} in 1970 to its peak in 1980 with 10.4 kg P ha^{-1}·yr^{-1} and dropped to 4.0 kg P ha^{-1} yr^{-1} in 2005. Western Europe showed a similar pattern, peaking in 1980 at 18.5 kg P ha^{-1} yr^{-1} while in 1970 and 2005 the application rates were 14.9 and 15.7 kg P ha^{-1} yr^{-1}, respectively. Oceania showed the lowest P input through manure application, with a maximum of 1.2 kg P ha^{-1} yr^{-1}.

Latin America showed a gradually increasing manure application from 3.6 to 4.9 kg P ha^{-1} yr^{-1} from 1970 to 2005, while North America's P application through manure fluctuated between 3.9 kg P ha^{-1} yr^{-1} in 2005 and 4.9 kg P ha^{-1} yr^{-1} 1970 with a maximum of 5.1 kg P ha^{-1} yr^{-1} in 1980.

Turning to grass P uptake, we also see large differences across world regions (and Table 4-1). In Africa, Asia, Latin America and Oceania, P uptake increased gradually between 1970 and 2005. In Africa P uptake increased from 1.9 to 3.1 kg P ha^{-1} yr^{-1}, in Asia from 5.1 to 6.4 kg P ha^{-1} yr^{-1}, in Latin America from 4.0 to 6.0 kg P ha^{-1} yr^{-1} in 1970 and 2005, respectively. Annual soil P uptake for Eastern Europe started at 10.2 kg P ha^{-1} yr^{-1} in 1970, peaked at 11.2 kg Pha^{-1}yr^{-1} in 1980 and then decreased rapidly to 4.0 kg P ha^{-1} yr^{-1} in 2005. In Oceania, uptake remained fairly constant around 1.3 kg P ha^{-1} yr^{-1} throughout the entire period. Annual P uptake in Western Europe was highest of all world regions, i.e. 18 kg P ha^{-1} yr^{-1} in 1970 and then decreasing to 15.7 kg P ha^{-1} yr^{-1} in 2005 with a maximum uptake rate of 20.7 kg P ha^{-1} in 1980.

4.3.2. Cumulative soil P budget in grasslands

Over the 1970-2005 periods, cumulative inputs of P through fertilizer and manure in Eastern Europe (38 Tg P) and Western Europe (56 Tg P) were slightly larger than the cumulative grass uptake of 34 and 49 Tg P, respectively. In contrast, cumulative P inputs of P in Africa (65 Tg P), Asia (72 Tg P) and Latin America (82 Tg P) were less than their grass P uptake (76, 191, 96 Tg P, respectively) for the same period (calculations based on Figure 4-2 and grassland area shown in Table 4-1). Of all the regions, Asia showed the highest negative P budget (-119 Tg P) while North America and Oceania had a balanced P budget.

Table 4-1 Global and regional grassland area and P inputs and outputs

Region*	**Area (10^6 ha)**		**Manure (kg/ha)**		**Mineral P (kg/ha)**		**Total P application (kg/ha)**		**Grass P uptake (kg/ha)**		**Imported Feed (kg/ha)**		**Export manure (kg/ha)**	
	1970	2005	1970	2005	1970	2005	1970	2005	1970	2005	1970	2005	1970	2005
Africa	882	904	1.6	2.6	<0.1	<0.1	1.6	2.6	1.9	3.1	0.0	0.1	0.3	0.5
Asia	874	971	1.8	2.4	<0.1	<0.1	1.8	2.4	5.1	6.4	0.2	0.7	1.2	1.8
East Europe	107	115	8.7	4.0	0.4	0.1	9.1	4.2	10.2	4.0	2.6	2.2	3.1	1.4
Latin America	487	546	3.6	4.9	<0.1	<0.1	3.6	4.9	4.0	6.0	0.2	0.2	0.5	1.0
North America	263	256	4.9	3.9	0.0	0.0	4.9	3.9	5.1	4.1	1.2	1.9	1.2	1.7
Oceania	462	407	1.2	1.2	<0.1	0.1	1.2	1.3	1.3	1.4	0.0	0.1	0.1	0.1
West Europe	79	67	14.9	15.7	3.2	1.7	18.1	17.4	18	15.7	2.1	4.0	4.5	3.6
World	3150	3270	2.7	3.1	0.1	0.1	2.8	3.2	4.0	4.6	0.3	0.6	0.8	1.1
World Intensive	560	537	8.2	9.9	0.6	0.4	8.8	10.3	13.2	16.1	1.9	3.5	4.0	5.8
World Pastoral	2590	2730	1.6	1.8	0.0	0.0	1.6	1.8	2.0	2.4	0.0	0.0	0.1	0.2

* The complete regional breakdown can be found in Table S4-4.

Globally, grassland soils received an estimated 374 Tg of P between 1970 and 2005 through mineral P fertilizer and manure applications. The intensive and pastoral systems received 209 and 165 Tg P, respectively (Table 4-1; Figure S4-7A; Figure S4-8A). Over the same time period, global cumulative grass P uptake and erosion from both systems were 509 and 33 Tg of P, respectively, which caused a negative P budget of -168 Tg for grassland soils between 1970 and 2005, 60% of which was in intensive and 40% in pastoral systems.

4.3.3. Phosphorus transfers between grasslands and croplands

Phosphorus is imported from cropland to grassland systems through livestock feed and P is exported from grassland systems through manure application to cropland. Feed use varied between different world regions (Figure 4-3). Africa and Oceania had a maximum rate of feed use of only 0.1 kg P ha^{-1} yr^{-1} in 2005 (Table 4-1), while Western Europe's feed use (4 kg P ha^{-1} yr^{-1}) was around seven-times the global rate (0.6 kg P ha^{-1} yr^{-1}) in 2005 (Table 4-1 and Figure 4-3A). The P import to the grassland systems in Asia and Latin America remained minimal at 0.7 and 0.2 kg P ha^{-1} yr^{-1}, respectively, in 2005. North America's input rate as feed was 1.9 kg P ha^{-1} yr^{-1} in 2005 (Table 4-1 and Figure 4-3).

North America with 14.5 Tg, Asia 13.3 Tg, Eastern Europe, 11.7 Tg and Western Europe with 8.2 Tg were responsible for about 90% of cumulative global feed use (53.4 Tg) in 1970-2005. In contrast, the imported P from croplands to grassland (as livestock feed) in Latin America (3.4 Tg), Africa (2.1 Tg) and Oceania (0.4 Tg) accounted only for 10% of the global feed value.

In 1970, the application of manure to cropland soils varied from 0.1 kg P ha^{-1} yr^{-1} in Oceania to the 4.5 kg P ha^{-1} yr^{-1} in Western Europe (Table 4-1 and Figure 4-3 G, H). The maximum manure P export occurred in 1980 in Western Europe (4.9 kg of P per hectare of grassland to croplands) followed by Eastern Europe (3.7 kg P ha^{-1} yr^{-1}). The other world regions never reached rates greater than the 1.8 kg P ha^{-1} yr^{-1} observed in Asia and North America in 2005. Cumulative global use of manure as fertilizer in croplands for the 1970-2005 period was 113 Tg of P. Asia alone was responsible for 44% of the global number, with a total of 50 Tg P. For instance in China, grazing system was confronted with a severe P deficit problem due to the massive transfers of P in the form of manure, while grasslands were hardly fertilized (Chen et al., 2008; Sattari et al., 2014b).

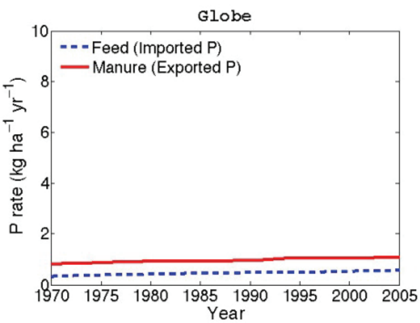
Africa (11.7 Tg P), Eastern (11.3) and Western Europe (11.5) had an almost equal share in the global P export from grassland to cropland (≈10% each). Cumulative transfer of P to cropland was only 1.7 Tg in Oceania, and 14.3 Tg in Latin America and 12.6 Tg in North America.

4.3.4. Net P transfers between grassland and cropland systems

It is clear that part of the P feed used in grassland-based systems is imported from other countries. Due to lack of data we ignore feed trade, but in regions such as Western Europe, P in feed produced in croplands in other world regions may be a significant part of total feed use. However, this does not affect the budget of the grassland-based systems, but may lead to underestimation of the net transfer from grassland to cropland within the feed importing regions. The net transfer from African grassland systems to cropland soils was 0.3 kg P ha^{-1} yr^{-1} in 1970 and peaked in 2005 at 0.4 kg P ha^{-1} yr^{-1} (calculated from Table 4-1). As a result, a net 10 Tg of P was transferred from grassland to cropland soils between 1970 and 2005 in Africa.

In Asia there was a constant net P transfer from grassland to croplands over the entire period (ca. 1.0 kg P ha^{-1} yr^{-1}) resulting in a net transfer of 36.7 Tg P from grasslands to croplands in the 1970-2005 period.

North America and Eastern Europe were the only regions that showed a cumulative net P transfer from cropland to grasslands systems, i.e. 1.9 and 0.4 Tg P, respectively. In Latin America, P was moved from grasslands to croplands throughout the entire period, at a rate of 0.3 kg P ha^{-1} yr^{-1} in 1970 and of 0.8 kg P ha^{-1} yr^{-1} in 2005, when it reached its peak. The cumulative number for Latin America was 10.9 Tg P. Oceania presented the lowest transfer rate, with an average of <0.1 kg P ha^{-1} yr^{-1}, which translated into the smallest cumulative amount of 1.3 Tg of P. In contrast, Western Europe showed the highest net transfer rate from grassland to cropland soils, i.e. 2.4 kg P ha^{-1} yr^{-1} in 1970, after which the yearly transfer to cropland gradually declined, to become negative in 2005 (-0.4 kg P ha^{-1} yr^{-1}). In total, Western Europe's grasslands provided 3.3 Tg of P to cropland soils between 1970 and 2005. As discussed above, a large part of this feed may stem from croplands in other regions.



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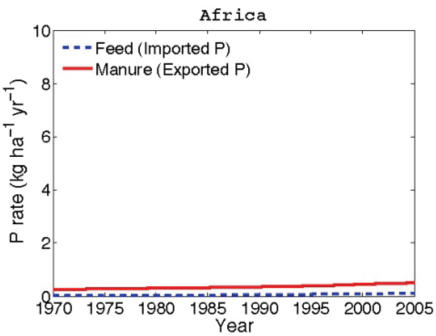

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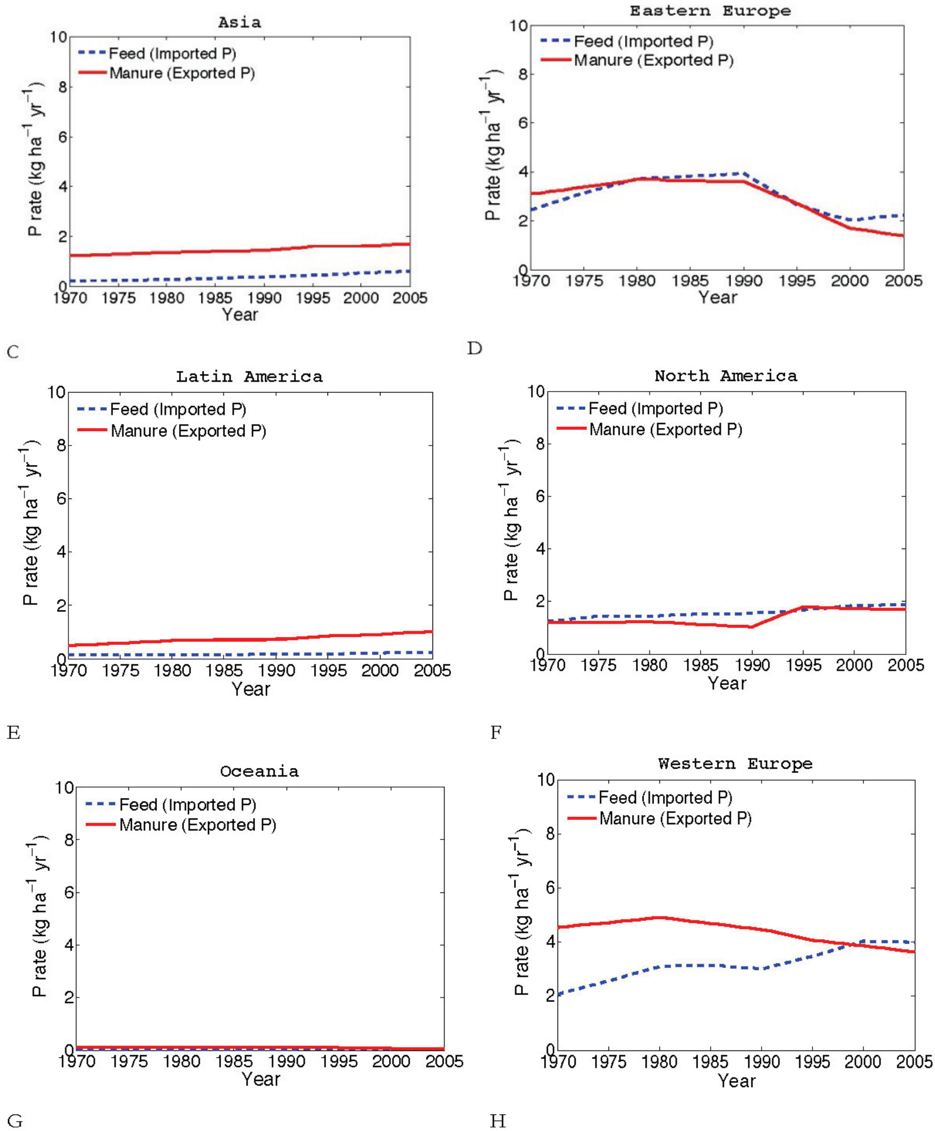


Figure 4-3 Historical trends of annual P imports to and exports from grasslands as livestock feed and manure spread in croplands, respectively, for the period 1970-2005 in (A) World, (B) Africa, (C) Asia, (D) Eastern Europe, (E) Latin America, (F) North America, (G) Oceania and (H) Western Europe. Dashed and solid lines represent imported P (feed) and exported P (manure), respectively.

4.3.5. Future demand of phosphorus in grassland

Historical data show that grass P uptake in the grassland systems (intensive and pastoral) exceeded P application through fertilizer and manure, which was confirmed in the present study. The rates of P uptake and P application in the intensive systems were systematically larger than in the pastoral systems (Figure S4-7A and Figure S4-8A).

We used the DPPS model to estimate P application rates (fertilizer and manure) based on target grass production from the Rio+20 scenario. The Rio+20 scenario projects a rapid increase in global grass production and P uptake from 4.6 to 8.3 kg P ha^{-1} due to the rapid increase in global meat and milk consumption and production (Figure 4-4).

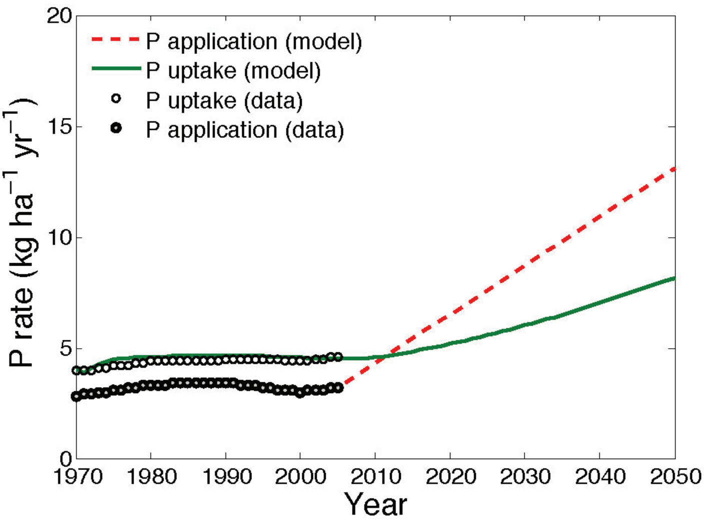


Figure 4-4 Trends of annual P application and grass P uptake in grassland for the period 1970 to 2050 in the entire globe according to Rio+20 scenarios. Long-term historical data and simulation results are illustrated by circles and lines, respectively. Shaded and open circles refer to P application and P uptake rates, respectively. Dashed and solid lines refer to P application and P uptake rates, respectively.

Using the DPPS model and assuming that no expansion of the grassland area will occur, we estimate that to achieve the 2050 target grass production the cumulative amount of P application that will be needed to be applied to grassland between 2005 and 2050 has to reach 372 kg P ha^{-1}, resulting in a global cumulative P input of ca.1215 Tg (Figure 4-4). According to the Rio+20 scenario, 654 Tg of the cumulative P input can be supplied by manure application (50% of the global cumulative). To achieve the target grass production and P uptake until 2050, the rest of the required cumulative P input (561 Tg) has to be supplied in the form of mineral P fertilizer to avoid further loss of grassland soil fertility under grasslands.

4.4. Discussion

4.4.1. Mineral P fertilizer use

Very little mineral fertilizer has been used in grasslands in most regions of world. Globally a total of 23 Tg of mineral P was applied to grassland soils over the period of 1970-2005, which is comparable to the annual use of P fertilizers in croplands of 18.5 Tg of P yr^{-1} in 2010 (FAO, 2011). In addition, rates of mineral P fertilizer use in grasslands differed strongly among regions of the world. Eastern and Western Europe alone accounted for almost 80% of the global total P fertilizer use in grassland. Given the positive P budgets in Europe there has been accumulation of residual soil P in grassland soils in a similar way that has been reported for croplands (Sattari et al., 2012).

4.4.2. Manure management

Manure management was the most important cause of negative soil P budgets in grasslands. For instance, in Asia only 34% of the manure P remained within the grassland boundaries between 1970 and 2005. The total livestock P excretion was about 196 Tg, out of which 66 Tg P spread in grasslands and 50 Tg P spread in croplands. Manure storage and subsequent active spreading in grasslands was significantly more important in North America, Eastern Europe and Western Europe than in the rest of the world. This reflects the higher historical development of intensive production systems in these regions.

Besides mineral P fertilizer, adequate manure recycling will be another key element to reduce the net P withdrawal from grasslands. As shown in our study, grass uptake and erosion accounted for 536 Tg of P over 1970-2005. Over the same period, P found in manure excretion was 530 Tg P of which 63% stayed within the grassland system. Such large transfers result in a loss of soil fertility in grassland systems. To avoid future unsustainable ruminant production, manure management systems are crucial determinants for the viability of grassland-based production, especially for poor farmers who lack access to mineral fertilizers (Rufino et al., 2007).

4.4.3. Grassland soil P depletion

Spreading of manure generated in grassland-based systems in croplands has two implications. On the one hand it helps to sustain and increase crop productivity (Sattari et al., 2012). On the other hand, reduced soil P fertility can deteriorate the status of those grasslands, which are already degraded (Suttie et al., 2005). In Africa, for instance, 70% of the population relies on dry and sub-humid lands for their livelihoods (FAO, 2009).

Our results show that P depletion is taking place in grassland soils. Globally, 4.4 kg of P per hectare were annually extracted from grassland soils via livestock grass intake. Meanwhile, the sum of natural weathering and atmospheric deposition replaced only 1.3 kg P $ha^{-1}yr^{-1}$ (Liu et al., 2008). For the entire period (1970-2005), global soil P replenishment through weathering and atmospheric deposition (15 Tg P) was clearly insufficient to compensate the cumulative grass uptake (510 Tg P) and soil erosion loss (26 Tg P). This clearly shows the dependency of world's grasslands to more P fertilizer to maintain soil fertility.

4.4.4. Global phosphorus demand until 2050

FAO expects a yearly increase of meat production between 0.5 and 4.0% between 2005 and 2050 for developed countries and South Asia, respectively (Alexandratos and Bruinsma, 2012). Bouwman et al. (2005) questioned how the large future increase in grass demand can be met, and in view of the loss of soil fertility as reported in the present paper, this is a question that has been answered by analyzing the soil P budgets and the impacts for the availability of soil P to sustain grassland production.

Whilst our results demonstrate a continuous process of soil P depletion in grasslands over the past decades, more grass will be needed in the future, which can either come from expansion of grasslands or from increased productivity. Raising productivity will only be possible if the P status of soils under grassland is improved, keeping an eye on nitrogen and potassium as well (Sattari et al., 2014a).

Penning De Vries et al. (1995) estimated that globally 7800 Mha are suitable for agriculture including 3800 Mha suitable for cropland and 4000 Mha for grazing. However, Young (1999) claimed that the area of usable land in the developing regions has been overestimated. Currently about 1520 (Sattari et al., 2012), 537 and 2730 Mha are used for cropland, intensive and pastoral grassland systems, respectively. Most of the additional lands are less fertile and easily degradable, and much remains covered by forest. Only some regions in South America and in Africa retain significant reserves of land suitable for crop production (Fischer and Schrattenholzer, 2001).

Some researchers believe that transferring of nutrients from grasslands to croplands through livestock manure could help in redistribution of nutrients in time and space (Delgado et al., 1999). Lands that are less suitable for cropping could be grazed to produce manure, which in turn provides organic matter and nutrients to make other land more suitable for crop production (Delgado et al., 1999). However, we expect that given the increasing future demand for grass (Sattari et al., 2012), grassland soils also need better management. To maintain the soil P status in grassland systems and meet the target grass production in 2050 as estimated by the Rio+20 scenario, and assuming no expansion of grasslands, our results show that increased use of mineral fertilizer will be essential.

Our estimate of the global P input required in grassland of 1215 Tg P for the 2005-2050 period includes P from both mineral fertilizer and animal manure. In 2050, globally 24 Tg yr^{-1} of mineral P fertilizer will be needed to avoid loss of soil fertility and declining grassland productivity, with a split of 11 and 13 Tg P fertilizers in the intensive and pastoral systems, respectively. A lower fertilizer use in grassland may cause grassland expansion that would lead to an increase in deforestation in the future.

There are only a few global studies that estimated the P inputs and outputs in croplands in 2050. Steen (1998) estimated that the global annual consumption on cropland would be 26-31 Tg P in 2050. Total P use in 2050 calculated by Bouwman et al. (Bouwman et al., 2009; Bouwman et al., 2013) ranges between 23 and 33 Tg yr^{-1}. These projections ignored the effect of residual soil P. Considering the effect of residual P in future projections of P demand, Sattari et al. (2012) estimated that average global P fertilizer use in croplands must change from the current 17.8 to 20.8 Tg yr^{-1} in 2050, which is substantially less the estimations by Steen and Bouwman et al.

The total amount of mineral fertilizer P needed in cropland and grassland systems is estimated to be 45 Tg in 2050, corresponding to 103 Tg P_2O_5 and around 350 Tg phosphate rock with an average grade of 29% P_2O_5. In the next four decades, a global cumulative amount of mineral P of 1380 Tg, accounting for 820 Tg in cropland (Sattari et al., 2012) (2008-2050) and 560 Tg in grassland (2005-2050), is required to achieve global food production spread in 2050. This implies that until 2050 the global production of phosphate rock will need to be 10,700 Tg to produce the P fertilizer for the agricultural and food production sectors.

While Sattari et al. (2012) estimated that future P requirements from fertilizer on cropland would increase less than the increase in crop production due to residual P effect, the P requirements from fertilizer in grasslands will increase more than proportional to the increase of grassland production, pressurizing the global P consumption.

4.5. Concluding remarks

We present the first global estimate of the required amount of P in 2050 for the total global food production including both cropland and grassland.

Our results point at a large P depletion in grassland soils in all regions in the world, except for Eastern and Western Europe, which are currently in equilibrium but have built up residual P in grassland soils in past decades. It is clear that the export of manure from grassland to cropland is primarily responsible for these globally unbalanced budgets. Mineral fertilizers in grasslands are of minor importance except in Europe, and P in ruminant meat and milk represents a small fraction compared with P in the manure reflecting the low P conversion efficiency in grassland-based systems.

Systematic loss of soil fertility through export in manure will worsen current rates of soil degradation across most of the global grasslands, affecting the livelihood of a vast number of world inhabitants. Future demand of meat and milk will increase the pressure on grasslands to provide grass and fodder for livestock. Increase of grass production for future ruminant production can be achieved, while avoiding grassland expansion and deforestation, through mineral P fertilization, adequate management of livestock and manure, and soil conservation practices to control runoff P losses. Importantly, more intensive ruminant production based on feed concentrates or less manure transfer from grasslands to croplands will both imply more P requirements in the world's croplands.

4.6. Supplementary Information

4.6.1. SI Material and Methods

Livestock production systems and animal groups

In their global classification, Seré and Steinfeld (1996) distinguished livestock production systems (LPS) and mixed farming production systems (MPS). LPS includes systems in which more than 90 per cent of dry matter fed to animals comes from rangelands, pastures, annual forages and purchased feed. Mixed farming production systems are systems in which more than 10 per cent of the dry matter fed to animals comes from crop by-products, stubble or more than 10 per cent of the total value of production comes from non-livestock farming activities (Seré and Steinfeld, 1996). They further distinguished four sub-groups: landless LPS, grassland LPS, rainfed MPS and irrigated MPS (Figure S4-5A). They produced data on buffaloes, cattle, goats, pigs, poultry and sheep population numbers and meat and milk production. Later, Bouwman et al. (2005) modified the classification from Seré and Steinfeld (1996) for their global analysis of ruminant production systems. Landless LPS and MPS were merged into mixed-landless systems (or intensive systems) while grassland LPS was renamed into pastoral systems (Figure S4-5B, C).

Following Bouwman et al. (2005) in the present study also two production systems were distinguished, i.e. grasslands in mixed and landless (referred to as intensive hereafter) and pastoral livestock production systems. Within each system, two groups of animals were considered: grassland based livestock (grassland-based livestock) including asses, buffaloes, camels, dairy cattle, horses, mules, non-dairy cattle, sheep and goats, and non-grassland based livestock (Non-grassland-based livestock) including pigs and poultry. Due to lack of data, it was not possible to include all animal categories for all the calculations (Table S4-2). Non-grassland-based livestock categories were not included in products P flow calculations since in our definition they are not located within grassland system boundaries (Figure 4-1 in the main text).

All abbreviations used in this part are shown in Table S4-3 and the list of countries in Table S4-4. Where IMAGE uses 5-year intervals, annual values were calculated by linear interpolation.

Phosphorus flows – Products

Livestock production data include milk, meat and livestock by-products. Livestock by-products include adipose tissue, skeleton, viscera, blood, skin, hair and digestive content. Country scale meat production data for non-dairy cattle and sheep & goats as well as milk production were obtained from FAOSTAT (Bouwman et al., 2005; FAO, 2014). Meat and/or milk production for buffaloes, horses, asses, mules and camels were not included due to lack of data. The total amount of P in livestock products is the sum over milk, meat and livestock by-products.

Meat and livestock by-products. The carcass weight (CW) is the most common way of expressing livestock meat production. CW is defined as the weight left after slaughter and removal of head, skin, genito-urinary organs and offals. The ratio between the carcass weight and the live weight (LW) is called the dressing percentage (DP). Live weight consists of the following fractions: muscles, adipose tissue, skeleton, viscera, blood, skin, hair and digestive content. Live weight partitioning allows for a more accurate P accounting due to large differences in P concentration in different fractions such as bones and blood.

In order to allocate the total national production to mixed-landless and pastoral systems, national, annual values of the fraction allocated to mixed-landless systems (*Fraction intensive*) were obtained from IMAGE (Bouwman et al., 2006). In addition, data on dressing percentages were obtained from Kempster et al. (1982), who reported 53 and 50% DP for non-dairy cattle and for sheep and goats, respectively. Live weight partitioning fraction data and P content of those fractions were obtained from different sources (Table S4-5).

Equation 4-1 is used to calculate meat and livestock by-products:

$$PLMe_{y,c,s} = \sum_a \left(\frac{LPD_{y,c,s,a}}{DP_{y,c,s,a}} \cdot \sum_i \left(LWF_{y,c,s,a,i} \cdot FPC_{y,c,s,a,i} \right) \right) \quad (4\text{-}1)$$

(*y* – year; *c* – country; *s* – production system; *a* – animal category; *i* – fraction of meat and by-products)

Phosphorus in meat and livestock by-products is denoted by *PLMe* (kg P). *LPD* and *DP* refer to the livestock production data (kg carcass weight) and dressing percentage (kg carcass per kg live weight), respectively. *LWF* (kg meat or livestock by-products per kg live weight) represents live weight partitioning and *FPC* refers to P content (kg P per kg meat or livestock by-products).

Milk. The total amount of P in milk was calculated by multiplying the total amount of milk production with the P content of milk, as shown in Equation 4-2. The P content was considered to be $8.4 \cdot 10^{-4}$ kg P per kg milk (USDA, 2012).

$$PLM_{y,c,s} = MPD_{y,c,s} \cdot MPC \quad (4\text{-}2)$$

(*y* – year; *c* – country; *s* – production systems)

Phosphorus in milk is expressed by *PLM* (kg P). *MPD* and *MPC* refer to the milk production data (kg milk) and milk phosphorus content (kg P per kg milk), respectively.

Phosphorus flows – Livestock manure and its allocation. To calculate the total amount of manure P and its allocation the following variables are needed for both mixed-landless and pastoral systems: annual nitrogen excretion, fraction other uses of manure, fraction grazing, fraction application of manure to grasslands and the ratio P to N in the manure. All these values are produced by the IMAGE model based on FAO data (FAO, 2014) on animal populations, expert estimations, literature review and model calculations (Bouwman et al., 2006). The methodology and equations described in this section are applied equally to all animal categories.

The manure allocation framework is shown in Figure S4-6. The total amount of manure P is calculated for all the animal categories, as shown in Equation 4-3. A certain amount of manure P that is allocated to the other uses (e.g. fuel, building material) is calculated as shown in Equation 4-4. A second amount, P excretion by grazing animals, is subtracted from the total manure according to Equation 4-5. The rest of the manure P is considered to be excreted in animal houses or stored, and available for spreading (Figure S4-6). Spreading occurs in grasslands (Equation 4-6) and croplands (Equation 4-7).

Livestock excretion (in Figure 4-1) equals the sum of the total P manure for the grassland-based livestock (non-dairy cattle, dairy cattle, buffaloes, sheep & goats, horses, asses, mules and camels). The *Other uses* flow corresponds directly to the other uses calculated in Equation 4-4 for the grassland-based livestock. *Spreading in croplands* in Figure 4-1 is the sum, for the grassland-based animal categories, of the outcome of Equation 4-7. Internal and external manure inputs correspond with the sum of P of manure excreted during grazing and application to grasslands from grassland-based and non-grassland-based livestock categories, respectively.

$$Manure_{y,c,s} = \sum_a \left(LPN_{y,c,s,a} \cdot Excretion\ rate_{y,c,s,a}\right) \qquad (4\text{-}3)$$

(*y* – year; *c* – country; *s* – production systems; *a* – animal category)

Manure refers to the total P in manure excreted by livestock (kg P). LPN is livestock population number (number of heads). *Excretion rate* indicates is the annual amount of P excreted for each animal category (kg P per head per year).

Amount of P (kg) in the manure allocated to the other uses is calculated by Equation 4-4.

$$Other\ uses_{y,c,s} = Manure_{y,c,s} \cdot FrOthers_{y,c,s} \quad (4\text{-}4)$$

FrOthers expresses the fraction of total manure allocated to the other uses.

Amount of P (kg) in the manure allocated to grazing is estimated by Equation 4-5.

$$Grazing_{y,c,s} = (Manure_{y,c,s} - Other\ uses_{y,c,s}) \cdot FrGrazing_{y,c,s} \quad (4\text{-}5)$$

FrGrazing denotes the fraction of total manure allocated to grazing.

Applied manure P (kg) in grasslands as fertilizer is calculated by Equation 4-6.

$$Grasslands_{y,c,s} = (Manure_{y,c,s} - Other\ uses_{y,c,s}) \cdot (1 - FrGrazing_{y,c,s}) \cdot FrGrass_{y,c,s} \quad (4\text{-}6)$$

FrGrass refers to the fraction of stored manure that is applied in grasslands as fertilizer.

Amount of P (kg) in the manure that is transferred from grazed livestock to cropland as fertilizer is estimated by Equation 4-7.

$$Croplands_{y,c,s} = (Manure_{y,c,s} - Other\ uses_{y,c,s}) \cdot (1 - FrGrazing_{y,c,s}) \cdot (1 - FrGrass_{y,c,s}) \quad (4\text{-}7)$$

Phosphorus flows – Livestock feed

The IMAGE model used FAOSTAT data on crops for feed and animal energy requirements to estimate the feed amount for each country, year and animal category (Bouwman et al., 2006). Only the animal categories of non-dairy cattle, dairy cattle and sheep & goats were considered for estimating feed crop use. Thus, a 100% grass ration was assumed for the rest of grassland-based animal categories (buffaloes, horses, asses, mules and camels).

IMAGE provides feed data for eleven feed items: oil crops, maize, pulses, rice, root and tuber crops, temperate cereals, tropical cereals, crop residues and other by-products, such as residues from breweries, and grass and scavenging (such as road-side grazing, food waste, etc.). However, those feed data were available only at regional scale. Therefore, disaggregation to country scale was needed. To disaggregate these IMAGE data on feed crops to country data scale, it was assumed for each animal category that the fraction of feed crops in the ration is proportional to livestock productivity. The total amount of feed crops available for a specific animal category within a world region is thus allocated on the basis of the productivity of the animal category considered in a specific country relative to the regional productivity. Country and regional productivity were calculated for each animal category as the country and regional amount of product divided by the country and regional animal numbers, respectively. Since this calculation was done for each animal category, productivity numbers were expressed in terms of kg of carcass weight per head^{-1} (for non-dairy and sheep and goats) or kg milk per head^{-1} (for dairy cattle). Then, for each country the ratio between its productivity over the regional productivity was computed, which yielded a weighting factor (first term of Equation 4-8). With the information on the regional feed amount and the animal numbers, the regional average feed intake was calculated for each feed item and animal category. The total amount of P for each feed item for a specific animal category within a certain country is thus the product of the weighting factor times the country's animal numbers times the regional average feed intake times the P content of the feed item. By adding up all the feed items and animal categories, the country total P amount in livestock feed crops was calculated.

$$PFE_{y,c,s} = \sum_a \left(\frac{LPR_{y,c,a}}{LPR_{y,R,a}} \cdot LPN_{y,c,s,a} \cdot \sum_f \left(\frac{FEED_{y,R,s,a,f}}{\sum_{C=1}^{C\ in\ R} LPN_{y,c,s,a}} \right) \cdot PFI_f \right) \tag{4-8}$$

(y – year; *c* – country; *s* – production systems; *a* – animal category; *f* – feed item; R-regional data)

PFE (kg P) is total amount of P in livestock feed. *LPR* refers to the livestock productivity (kg per head). It is calculated as the total amount of products associated with the animal category (carcass weight or milk) over the total number of animals. *LPN* refers to the animal numbers and the total amount of feed item used in a certain region as presented by *FEED* (kg). *PFI* (kg P per kg feed item) is the P content for a given feed item.

Phosphorus flows – Livestock grass intake and grass P uptake

The estimation of grass uptake uses a mass balance approach. At animal level, P inputs (feed plus grass intake) equal the P outputs (manure plus products). Thus, P in the livestock grass intake equals grass P uptake from the soil. This approach ignores any P losses during mowing, transporting or stall-feeding of grass. The grass P uptake is thus calculated according to Equation 4-9. The amount of feed *(PFE)* and products (*PLMe* and *PLMi*) were assumed to be zero for buffaloes, horses, asses, mules and camels. For these animal categories the amount of P in grass intake equals the excretion of P.

$$PGU_{y,c,s} = (PLMe + PLMi)_{y,c,s} + Manure_{y,c,s} - PFE_{y,c,s} \tag{4-9}$$

(*y* – year; *c* – country; *s* – production systems)

PGU (kg P) is the total grass phosphorus uptake and *PFE* (kg P) refers to the total amount of phosphorus (kg) in livestock feed.

Mineral phosphorus fertilization

The IMAGE data includes the total use of mineral phosphorus fertilizers in grasslands for 1950, 1970, 1980, 1990, 1995, 2000 and 2005 at country level. The values are based on FAOSTAT (FAO, 2014) and the International Fertilizer Industry Association (FAO/IFA/IFDC, 1999; IFA/IFDC/FAO, 2003) for the period 1995-2000, and extrapolations for earlier and later years. Those numbers are exclusively allocated to the mixed and landless production systems, as we assumed no use of mineral fertilizer in pastoral systems.

Soil phosphorus erosion/runoff

Two nutrient loss pathways have been distinguished to estimate P runoff and erosion. Losses from recent nutrient applications (P_{rec}) in the form of fertilizer, manure or organic matter (Hart et al., 2004), and a residual or "memory" (P_{mem}) effect related to long-term historical changes in soil nutrient stocks for the top 30 cm (McDowell and Sharpley, 2001; Tarkalson and Mikkelsen, 2004). The approach presented by Cerdan et al. (2010) were used as a basis for calculating P_{mem} based on slope, soil texture and land cover type.

P_{rec} is calculated from P inputs and depends on slope (using the approach of Bogena et al. (2005)) and is further modified by land use and soil texture, i.e. those factors that reduce surface runoff according to Velthof et al. (2009; 2007).

The initial P stock in the top 30 cm is taken from Yang et al. (2010) for the year 1900. All inputs and outputs of the soil balance are assumed to occur in the top 30 cm; the model replaces P enriched or depleted soil material lost at the surface by erosion with fresh soil material (with the initial soil P content) at the bottom.

Comparing the results with other studies

The estimates of global P budgets of grassland's soil and P transfers between grasslands and croplands are compared in Table S4-6 with those reported by other authors. Although it is not easy to compare the full results obtained in this study with other studies since the approach, system boundaries and the scope are different. However, a comparison of selected flows calculated here at country level with other studies is possible (Table S4-6). Frequently, in literature larger values than the ones reported here are found. Inclusion of pigs and poultry in the other studies can explain most of the differences found.

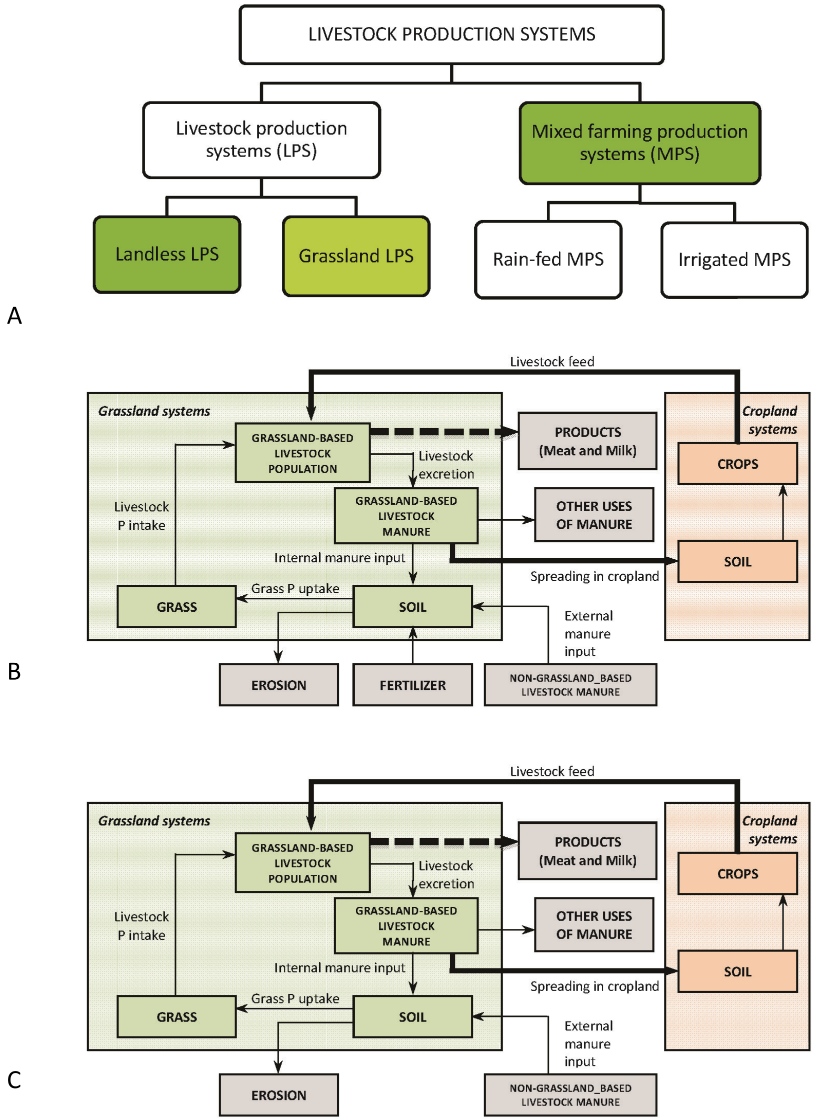


Figure S4-5 Livestock production systems and grassland systems classification. The livestock production systems classification was adopted from (Seré and Steinfeld, 1996) (A). Landless livestock and mixed farming production systems (dark green boxes in panel A) constitute the base of intensive systems adopted in this study (B). Pastoral systems (C) are based on entirely grassland-based (GB) livestock production systems (light green in A).

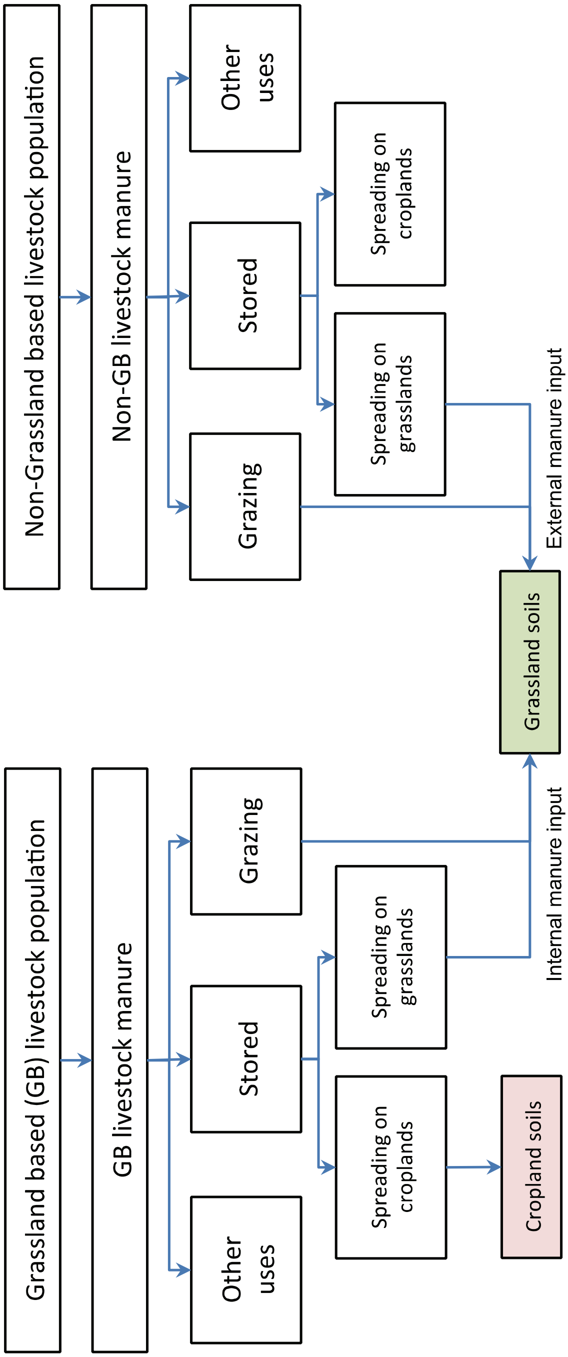
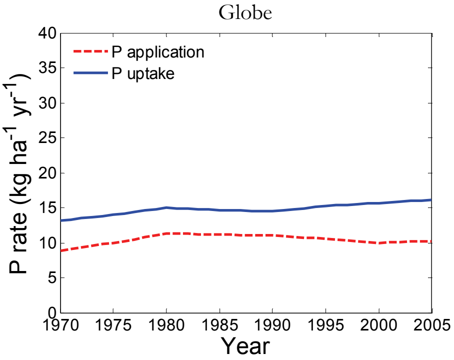


Figure S4-6 Manure allocation framework.



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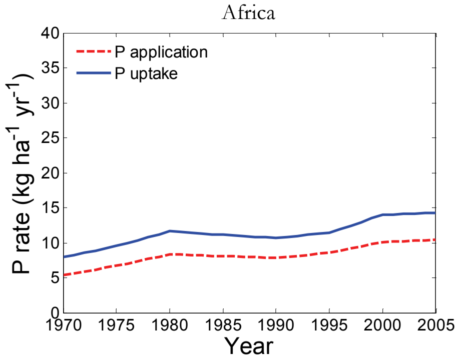

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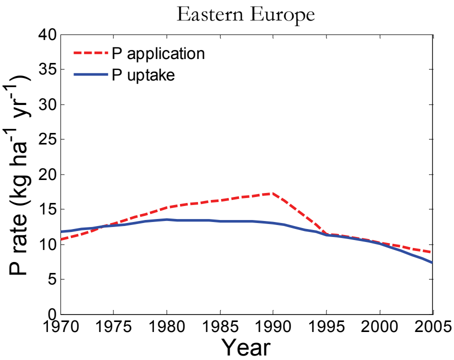

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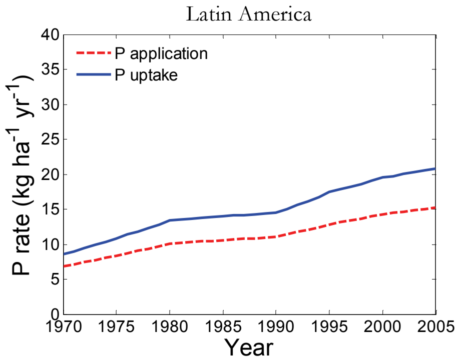

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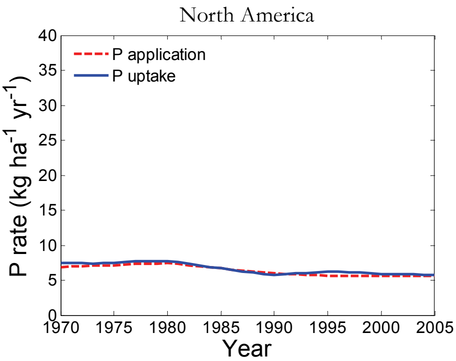

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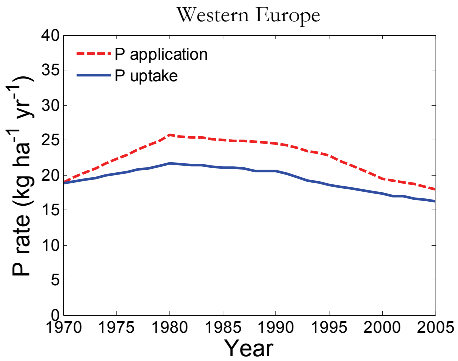

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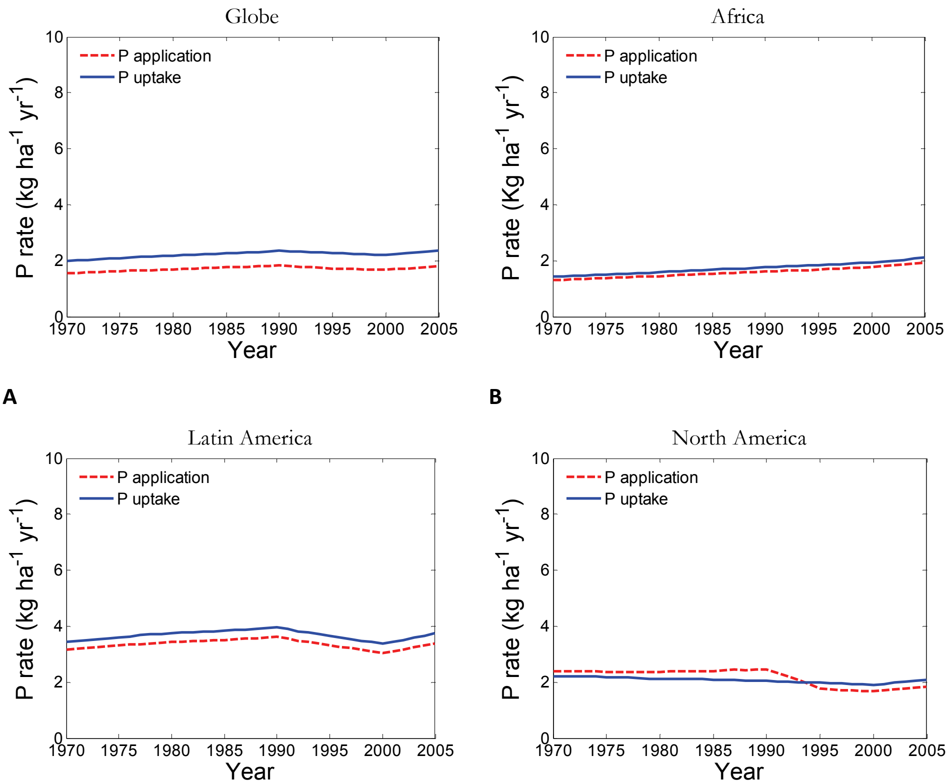
Figure S4-7 Intensive grassland system. Historical trends of annual P application and P uptake in intensive grassland systems for the period 1970-2005 in (A) World, (B) Africa, (C) Eastern Europe, (D) Latin America, (E) North America, and (F) Western Europe. Dashed and solid lines represent P application and P uptake, respectively.



A B C D

Figure S4-8 Pastoral grassland systems. Historical trends of annual P application and P uptake in pastoral grassland systems for the period 1970-2005 in (A) World, (B) Africa, (C) Latin America, and (D) North America. Dashed and solid lines represent P application and P uptake, respectively. There are no pastoral systems in Europe.

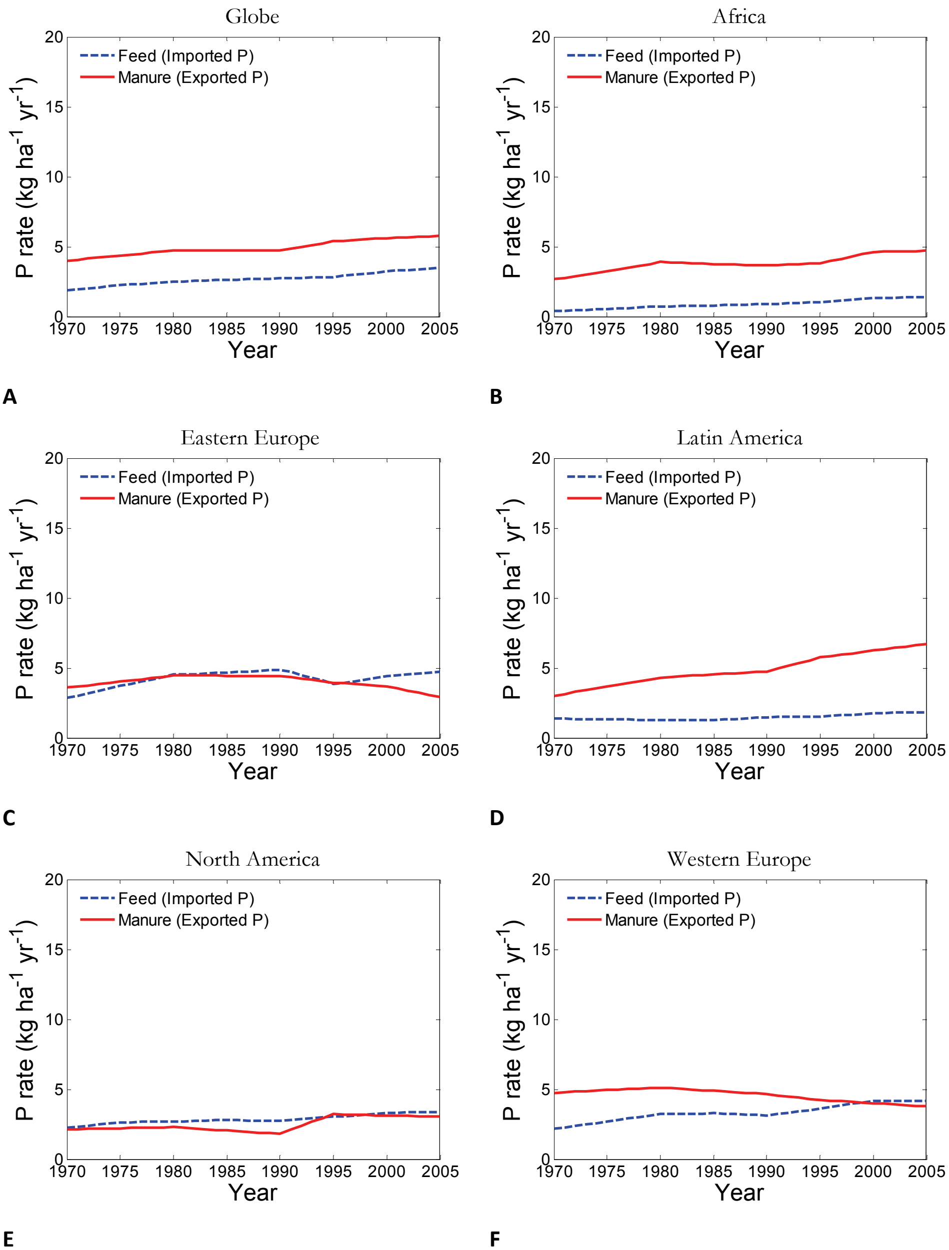


Figure S4-9 Intensive grassland system. Historical trends of annual P imports to and from grasslands as livestock feed and manure applied to croplands, respectively, for the period 1970-2005 in (A) World, (B) Africa, (C) Eastern Europe, (D) Latin America, (E) North America, and (F) Western Europe. Results in Asia and Oceania are not shown since the data for these two regions cannot disaggregate.

Table S4-2 Description of phosphorus flows and the animal categories involved

Flow	**Description**	**Animal category**
Products	Amount of P that leaves the grassland systems through animal products, e.g., meat and milk. By-products such as bones, viscera, blood, etc.	Non-dairy cattle, dairy cattle and sheep & goats.
Livestock excretion	Total amount of P in grassland-based livestock excretion.	Non-dairy cattle, dairy cattle, buffaloes, sheep & goats, horses, asses, mules and camels.
Internal manure input	Amount of P in grassland-based livestock excretion that is returned to grassland soils.	Non-dairy cattle, dairy cattle, buffaloes, sheep & goats, horses, asses, mules and camels.
Spreading in cropland	Amount of P in grassland-based livestock excretion that is used as organic fertilizer in croplands.	Non-dairy cattle, dairy cattle, buffaloes, sheep & goats, horses, asses, mules and camels.
Other uses	Amount of P in grassland-based livestock excretion allocated to non-agricultural uses (fuel, building purposes, etc.).	Non-dairy cattle, dairy cattle, buffaloes, sheep & goats, horses, asses, mules and camels.
External manure input	Amount of P in non-grassland-based livestock excretion that is used as organic fertilizer in grasslands.	Pigs and poultry.
Fertilization	Amount of P applied to grassland soils through mineral fertilization.	NA
Erosion	Amount of P that is lost from grassland soils due to erosion and runoff.	NA
Grass uptake*	Amount of P that grass takes from grassland soils.	Non-dairy cattle, dairy cattle and sheep & goats.
Grass intake*	Amount of P in the grass used as feed for grassland-based livestock animals.	
Livestock feed	Amount of P in food crops used as feed for grassland-based livestock animals.	Non-dairy cattle, dairy cattle and sheep & goats.

* Grass uptake and grass intake are assumed to be equal in the model, i.e. losses of mown grass are neglected.

Table S4-3 Abbreviations used in this study, description and units.

Symbol	Description	Units
y	Year	Year
c	Country	-
s	Production systems	-
a	Animal category	-
i	Fraction of meat and by-products	-
R	Regional data	
PLMe	Phosphorus in meat and livestock by-products	kg P
LPD	Livestock production data	kg carcass
DP	Dressing percentage	kg carcass per kg live weight
LWF	Live weight partitioning fraction	kg fraction (muscle, adipose tissue, etc.) per kg live weight
FPC	Phosphorus content	kg P per kg products or by-products
PLMi	Phosphorus in milk	kg P
MPD	Milk production data	kg milk
MPC	Milk phosphorus content	kg P per kg milk
Manure	Total P in manure excreted by livestock	kg P
LPN	Livestock population numbers	heads
Excretion rate	Annual P excretion for each animal category	kg N per head per year
Other uses	Amount of P in the manure allocated to other uses	kg P
FrOthers	Fraction of total manure allocated to other uses	-
Grazing	Amount of P in the manure allocated to grazing	kg P
FrGrazing	Fraction of total manure allocated to grazing	-
Grasslands	Amount of P in the manure that is applied in grasslands as fertilizer	kg P
FrGrass	Fraction of stored manure that is applied in grasslands as fertilizer	-
Croplands	Amount of P in the manure that is used in croplands as fertilizer	kg P
PFE	Total amount of phosphorus in livestock feed	kg P
LPR	Livestock productivity. It is calculated as the total amount of products associated with the animal category (carcass weight or milk) over the total number of animals	kg carcass per head (non-dairy cattle and sheep & goats) kg milk per head (dairy cattle)
LPN	Livestock population numbers	# heads
FEED	Total amount of feed item used in a certain region	kg feed item
PFI	Phosphorus content for a given feed item	kg P per kg feed item
FE	Total amount of phosphorus in livestock feed	kg P
PGU	Total grass phosphorus uptake	kg P

Table S4-4 List of countries and regions as used in the present study based on IMAGE (Bouwman et al., 2006)

AFRICA	Algeria, Angola, Benin, Botswana, Burkina Faso, Burundi, Cameroon, Cape Verde, Central African Republic, Chad, Comoros, Congo, C“te d'Ivoire, Djibouti, Egypt, Equatorial Guinea, Eritrea, Ethiopia, Gabon, Gambia, Ghana, Guinea, Guinea Bissau, Kenya, Lesotho, Liberia, Libyan Arab Jamahiriya, Madagascar, Malawi, Mali, Mauritania, Mauritius, Morocco, Mozambique, Namibia, Niger, Nigeria, R,union, Rwanda, Sao Tome and Prince, Senegal, Seychelles, Sierra Leone, Somalia, South Africa, Sudan, Swaziland, Tanzania, The Democratic Republic of the Congo, Togo, Tunisia, Uganda, Western Sahara, Zambia, Zimbabwe.
ASIA	Afghanistan, Armenia, Azerbaijan, Bahrain, Bangladesh, Bhutan, Brunei Darussalam, Cambodia, China, Cyprus, Democratic People's Republic of Korea, Georgia, India, Indonesia, Iran, Iraq, Israel, Japan, Jordan, Kazakhstan, Kuwait, Kyrgyzstan, Lao People's Democratic Republic, Lebanon, Malaysia, Maldives, Mongolia, Myanmar, Nepal, Oman, Pakistan, Philippines, Qatar, Republic of Korea, Saudi Arabia, Singapore, Sri Lanka, Syrian Arab Republic, Taiwan, Tajikistan, Thailand, Timor-Leste, Turkey, Turkmenistan, United Arab Emirates, Uzbekistan, Vietnam, Yemen.
LATIN AMERICA	Antigua and Barbuda, Argentina, Bahamas, Barbados, Belize, Bermuda, Bolivia, Brazil, British Virgin Islands, Cayman Islands, Chile, Colombia, Costa Rica, Cuba, Dominica, Dominican Republic, Ecuador, El Salvador, Falklands Islands, French Guyana, Grenada, Guadeloupe, Guatemala, Guyana, Haiti, Honduras, Jamaica, Martinique, Mexico, Montserrat, Netherlands Antilles, Nicaragua, Panama, Paraguay, Peru, Puerto Rico, Saint Kitts and Nevis, Saint Lucia, Saint Vincent end the Grenadines, Suriname, Trinidad and Tobago, United States Virgin Islands, Uruguay, Venezuela.

Table S4-4 Continued.

WESTERN EUROPE	Albania, Austria, Belgium, Bosnia and Herzegovina, Croatia, Denmark, Estonia, Faroe Islands, Finland, France, Germany, Greece, Iceland, Ireland, Italy, Latvia, Lithuania, Luxemburg, Macedonia, Malta, Netherlands, Norway, Portugal, Slovenia, Spain, Sweden, Switzerland, United Kingdom.
OCEANIA	American Samoa, Australia, Cocos Islands, Cook Islands, Fiji, French Polynesia, Guam, Kiribati, Marshall Islands, Micronesia, Nauru, New Caledonia, New Zealand, Niue, Papua New Guinea, Samoa, Solomon Islands, Tokelau, Tonga, Tuvalu, Vanuatu, Wallis and Futuna
EASTERN EUROPE	Belarus, Bulgaria, Czech Republic, Hungary, Poland, Republic of Moldova, Romania, Russian Federation, Slovakia, Ukraine
NORTH AMERICA	Canada, Saint Pierre and Miquelon, United States of America

Table S4-5 Fractions and P content of live weight for meat and livestock by-products

Animal category	Muscles		Adipose tissue		Skeleton		Viscera		Blood		Skin & hair		Gut content	
	%	P cont.	%	P cont.	%	P cont.	%	P cont.	%	P cont.	%	P cont.	%	P cont.
Non-dairy cattle	31.3	212[a]	19.8	730[*]	13.5	950[b†]	8.5	730[d]	3.0	38[d]	7.0	42[c]	17.0	730[*]
Sheep & goats	28.5	385[a]	15.5	730[*]	13.5	950[b†]	11.0	730[*]	4.0	38[d††]	13.5	42[c‡]	12.0	730[*]

P content is expressed in mg P · 100g fraction^{-1}. All values of live weight fractions were obtained from (Kempster et al., 1982). Data on P content were obtained from (a) USDA (2012), (b) Schalkwyk (2004), (c) Silva ASd, et al (2011) and (d) Williams et al. (1991). * Indicates that the numbers for adipose tissue, gut content and sheep & goats viscera were assumed to be equal as the P content in non-dairy cattle viscera. † Indicates that bone P content was assumed to be the same as the value for buffaloes found in Schalkwyk (2004). The blood content for sheep & goats was assumed (††) to be the same as the non-dairy cattle value. Similarly, P content for skin & hair in sheep & goats (‡) was taken from non-dairy cattle values.

Table S4-6 P flows comparison with other studies.

Flow	Country	Year	Amount	This study (2005)
Products	France*(1)	2002-2006	1.0 – 12.3 kg P ha^{-1}	2.5 kg P ha^{-1}
	UK*(2)	2009	32 Gg P yr^{-1}	21 Gg P yr^{-1}
	Finland(3)	1995-1999	4 Gg P yr^{-1}	3 Gg P yr^{-1}
	US*(4)	2007	390 Gg P yr^{-1}	157 Gg P yr^{-1}
Animal excretion	France*(1)	2002-2006	4.2 – 29.0 kg P ha^{-1}	17 kg P ha^{-1}
	Finland(3)	1995-1999	11 Gg P yr^{-1}	14 Gg P yr^{-1}
Application to grassland	US*(4)	2007	750 Gg P yr^{-1}	518 Gg P yr^{-1}
Application to croplands	US*(4)	2007	523 Gg P yr^{-1}	401 Gg P yr^{-1}
Grass uptake/ intake	France*(1)	2002-2006	2.8 – 11.3 kg P ha^{-1}	14.5 kg P ha^{-1}
	UK*(2)	2009	110 Gg P yr^{-1}	159 Gg P yr^{-1}
Feed	France*(1)	2002-2006	2.3 – 29 kg P ha^{-1}	3.2 kg P ha^{-1}
	UK*(2)	2009	95 Gg P yr^{-1}	26 Gg P yr^{-1}

(1) (Senthilkumar et al., 2012); (2) (Cooper and Carliell-Marquet, 2013); (3) (Antikainen et al., 2005); (4) (MacDonald et al., 2012). *Ref. includes pigs and poultry

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5

Key role of China and its agriculture in global sustainable phosphorus management

This chapter has been published as:
Sattari, S. Z., van Ittersum, M. K., Giller, K. E., Zhang, F., and Bouwman, A. F. (2014). Key role of China and its agriculture in global sustainable phosphorus management. Environmental Research Letters 9, 054003.

Growing global demand for food leads to increased pressure on phosphorus (P), a finite and dwindling resource. China is the largest producer and consumer of P fertilizer in the world. A mass balance analysis of historical P use on China's arable land shows that P input substantially exceeds crop P uptake leading to the accumulation of residual soil P. A Dynamic P Pool Simulator (DPPS) model is applied to estimate future P demand in China's arable land. Our simulations show that more sustainable use of P accounting for the residual P can save ca. 20% of the P fertilizer needed until 2050 in China relative to the Rio+20 Trend scenario. This saving would be equivalent to half of the P required in Africa or sufficient for Western Europe, to achieve target crop P uptake in 2050.

5.1. Introduction

Phosphorus is a major limiting nutrient in agriculture (Sattari et al., 2014; Syers et al., 2008) that is increasingly considered to be a new global sustainability challenge because of finite P resources (Cordell and Neset, 2014). In the short-term, increasing P demand may lead to rising prices of P-fertilizer and food, and in the long-term to depletion of global P reserves (Smil, 2000) that will seriously impact food security particularly in developing countries because agriculture is the prime P consumer (Koning et al., 2008). P scarcity has five dimensions, including physical, geopolitical, institutional, economic and managerial scarcity (Cordell and Neset, 2014). Some studies depict a world with rapidly depleting P reserves in the near future (Cordell et al., 2009). While the time scale of P depletion is debatable (Cordell et al., 2009; Dery and Anderson, 2007; Van Kauwenbergh, 2010; Van Vuuren et al., 2010), a critical question beyond the physical availability of P is whether P resource depletion can be managed by sustainable P consumption. This question is especially relevant as the global issue of P depletion is largely studied from the perspective of P sourcing (i.e. current production and use of phosphate rock) rather than future P requirements (i.e. P required to feed the world in future) (Sattari et al., 2012). The critical role of phosphorus management was recently illustrated by including residual soil P from past surplus fertilizer and manure applications in the estimation of required future P input (Sattari et al., 2012). Our results showed that by considering residual soil P, the global P crisis could be postponed. Since different countries experience P scarcity in different ways, assessing the P vulnerability and adaptive strategies to increase the resilience of the food system to P scarcity at national scale would be most appropriate (Cordell and Neset, 2014).

Among all countries, China has a strategic position in phosphorus production and consumption. China as a country with a long history of agriculture and currently feeding 20% of the world's population (1.3 billion) on only 7% of the arable land (130 million ha) (Li et al., 2011) has been selected for this assessment. At the start of the 21st century almost one third of the total P consumed in China was imported, and the remainder mined from its own phosphate rock deposits. Gradually, China started to increase P production, consumption and even export (Figure 5-1), with substantial government support (Zhang et al., 2008). Consequently, this country is now the largest global consumer (30%) (Zhang et al., 2008) and the largest producer (37.5% of world total in 2010) of P fertilizer (USGS, 2012) and is responsible for 50% of total Asian fertilizer P use. From the livestock side China also ranks first in terms of monogastric animal stocks (pigs and poultry) in the world (Chen et al., 2008). It is relevant to assess future P requirements to support food production, since Zhang et al. (2008) reported that in the next two decades, 30-50% more food will be needed to meet China's projected demand.

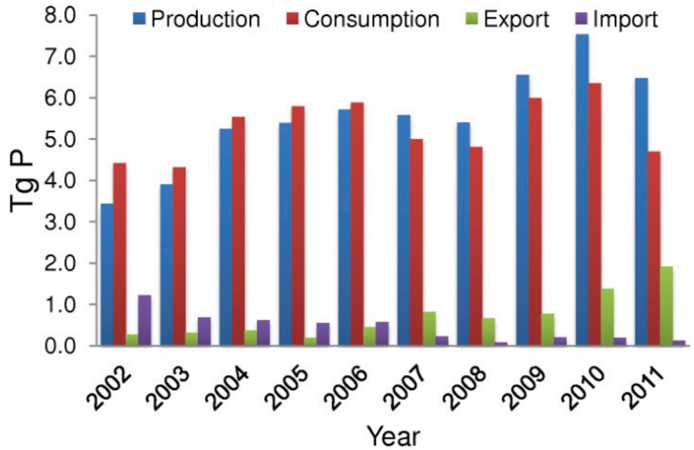

Figure 5-1 Historically produced, consumed, imported and exported phosphorus during 2002-2011 in China (FAO, 2011).

Hence, among all countries, China has an undeniable influence on global P production and consumption trends, which is key information for politicians and policy makers (Sheldrick et al., 2003b).

Environmental impacts associated with large P surpluses are multiple (Vitousek et al., 2009). Excessive P fertilizer application leads to accumulation of P in the soil and a significant amount of P can be lost by surface runoff to surface waters (Carpenter, 2005), a major cause of eutrophication of freshwater ecosystems and P leaching.

In this study we analyse the historical and future perspective of agricultural P use in China. We investigate whether China can play a significant role in global supply of P, while fulfilling its own need to feed a growing population in future decades.

We present an analysis of historical P budgets on China's arable land at national, regional and provincial levels. Using these budgets, we estimated the time needed to reach soil P saturation at the provincial scale and consequent P leaching from soils. A Dynamic Phosphorus Pool Simulator (DPPS) (Sattari et al., 2012; Wolf et al., 1987) was applied to simulate the history of four decades of P uptake. Subsequently, DPPS was used to estimate future P demand in China's arable land by accounting for the residual P stored in the soil due to the surplus P application between 1970 and 2010. Our estimates can be considered as sustainable P rates, as they are based on the soil available P stocks combined with production conditions and crop yields. The text refers to online supplementary information (SI) that provides details not included in the main text.

5.2. Materials and Methods

5.2.1. Overview of China's agriculture

Agriculture is a vital industry in China, employing over 300 million farmers. Through intensification, the agricultural revolution in China has influenced the usage and accumulation of nutrients, but at the same time increased the risk of water pollution (Chen et al., 2008). Crop production has greatly increased from the 1980s, due to improved crop varieties in combination with increasing use of fertilizers, pesticides and irrigation water (Ma et al., 2012). China ranks first in worldwide farm output, primarily producing rice, wheat, potatoes, sorghum, peanuts, tea, millet, barley, cotton, oilseed, pork, and fish.

Before 1978 most of the nutrients were recycled in mixed farming systems; the livestock was fed with feed crops, fodder and crop residues, and crops in turn were fertilized with animal and human manure and fertilizers (FAO, 1977).

Since 1978 a modern market-oriented farming started to gradually replace traditional farming. Mineral fertilizer became cheaper in comparison with labour costs to collect and recycle the wastes in arable land (Chen et al., 2008). Moreover, livestock production increased rapidly due to increased imports of animal feed (Ma et al., 2012) and became more centralized with an increase in the average number of animals per farm. Spatially, livestock production was increasingly uncoupled from crop production (Ma et al., 2012). Manure from confined animal feeding operations is increasingly discharged directly to surface water. In areas with highly intensive livestock production, the supply of nutrients in livestock manure often exceeds the crop needs, and surpluses accumulate in soils or are lost by surface runoff or leaching and can lead to eutrophication of freshwater and marine aquatic ecosystems (Chen et al., 2008).

China has 31 provinces, which were grouped in this study in five regions (North, South, Northeast, Northwest, and Yangtze Plain, See Figure S5-6). The farming and cropping systems in these regions differ significantly. Slightly more than 10% of the total land area is used for intensive crop production, mostly in the North, Northeast and Yangtze plain regions. Furthermore, animal production has partly moved from the rural areas to urban areas, close to the food processing industry and main food consumption centres (Ma et al., 2012).

The decoupling of livestock and crop production is most apparent around big cities. High animal densities are nowadays found around Beijing, Shanghai and Guangzhou, and less so in many rural areas in the North, Northeast, Yangtze plain and Southeast with traditional smallholders. Low-density grazing systems occur mostly in the Northwest region of China in five main pastoral districts (Inner Mongolia, Gansu, Qinghai, Tibet, and Xing Jiang). Fertilizer use is generally much lower in the western than in the eastern part of the country. In some areas, crop residues are used for animal feed or fuel, while animal manure is also used as fuel (Ma et al., 2012).

5.2.2. Model description

The Dynamic Phosphorus Pool Simulator (DPPS) -a simple two-pool P-model (Wolf et al., 1987) including labile and stable pools and long-term P input and output data (Sattari et al., 2012) – was used. The DPPS model reproduces historical crop uptake as a function of P inputs (fertilizer and manure). This model considers the essential P fluxes between crops and soil. It includes both labile and stable pools of phosphorus with a yearly time step. The model can calculate P transfer between different pools, the P uptake by crops, and the pool sizes. The model can also be applied for calculating the fertilizer requirement for a future target yield. The DPPS model successfully simulated the historical patterns of crop P uptake as a response to the application rates in all continents and the entire globe as shown in Sattari et al. (2012).

The model was used to calculate future P fertilizer and manure application rates in China based on target crop productions in 2050 derived the Rio+20 Trend or baseline scenarios. Details of the model, and its application to estimate future P requirements are given in Sattari et al. (2012) and Wolf et al. (1987).

5.2.3. Phosphorus inputs and outputs

Annual P input – output budgets were calculated from 1970 to 2010 at three different scales, i.e. provincial, regional and the country. Annual crop yields, annual mineral P fertilizer consumption and areas of arable crops are available at country scale from FAO (FAO, 2011) for the period 1970-2010 and from Chinese statistics (NBSC, 2010) from 1978 onwards at both national and provincial scales. Chinese statistics at provincial scales were used to disaggregate the long-term FAO national data to provincial level (Bouwman et al., 2006).

Manure. No data on manure P production and use is available from the agricultural statistics. We used data on animal stocks for dairy and non-dairy cattle, buffaloes, sheep and goats, pigs, poultry, horses, asses, mules and camels for the years 1970, 1980, 1990, 2000, 2005 and 2010 (Bouwman et al., 2009) and used linear interpolation between these years. Total manure production of P was calculated from total animal stocks and P excretion rates (see SI) (Bouwman et al., 2009; Bouwman et al., 2013). Animal manure available for application to crops and grassland includes all stored or collected manure and excludes excretion in grazing land and animal manure used for other purposes (fuel, building material), or manure not used at all (such as manure from confined animal feeding operations collected in lagoons or discharged to surface water). More explanations about the manure calculations, manure use and excretion rates per animal category as well as manure not recycled in agriculture have been given in the SI.

Weathering and atmospheric deposition. We used global values from Liu et al. (2008) for P supply from weathering (1.6 Tg yr^{-1}) and atmospheric deposition (0.4 Tg yr^{-1}) and to calculate the average P inputs per hectare (1 and 0.25 kg ha^{-1} yr^{-1} from weathering and deposition, respectively).

Runoff and erosion. Runoff is the overall dominant loss pathway for P from agricultural land, and P leaching is important in areas with P-saturated soils in some industrialized countries. To estimate P runoff loss, we used the increase of global P river export as presented by Seitzinger et al. (2010) for the period

1970-2000 (excluding the contribution of sewage), assuming that this increase can be completely attributed to agricultural activities. This increase includes sediment, particulate and dissolved P forms, and is corrected for P retention in river systems. This simple calculation yields a P loss rate to aquatic systems through surface runoff (Bouwman et al., 2009) of about 10% of the soil P inputs from fertilizer and manure.

In DPPS, fertilizer and manure inputs are directly allocated to the labile and stable pools. Wolf et al. (1987) proposed indicative values for the fractions of labile and stable pools in superphosphate fertilizer of 0.8 and 0.2, respectively. The calculated P loss by runoff is therefore actually taken from the labile and stable pools in a ratio of 4:1 (Sattari et al., 2012) before allocation to the pools.

Crop uptake. Yields of 116 crops grown in China are available from 1970 to 2010 (FAO, 2011). P contents for each crop were obtained from different sources (Bouwman et al., 2005a; USDA, 2010) and the amount of P harvested was calculated as production times the P content.

5.2.4. Scenario for the period 2010-2050

We used the Rio+20 Trend scenario for simulations of future P use in China. Similar to the baseline scenario of the Environmental Outlook of the Organization for Economic Cooperation and Development (OECD) (OECD, 2012), the Global Orchestration scenario of the Millennium Ecosystem Assessment (Alcamo et al., 2006) and the A1 scenario of IPCC-SRES (Nakicenovic et al., 2000), the Rio+20 Trend scenario is a baseline or business-as-usual scenario with similar assumptions on population growth and economic development pathways. Baseline scenarios represent a continuation of current trends, with no dramatic changes or shifts in production and management systems, attitude towards environmental problems, etc. The Rio+20 study (van Vuuren et al., 2012) describes four scenarios, the Trend scenario and three challenge pathways. The Trend scenario describes possible trends in the absence of the climate and sustainability policies. The three challenge pathways were designed to assess the potential to achieve sustainability goals.

In the Rio+20 Trend scenario, the world population is projected to grow from 7 to 9 billion people and the population of China from 1.360 billion to 1.415 billion inhabitants during the period 2010-2050. Towards 2040, the highest economic growth rates are projected for Asia compared with other continents, which will lead to a strong increase in demand for agricultural products and consequent changes in land use. Historically, almost 20% of the global increase in agricultural production was achieved by expanding the total agricultural area. This trend is expected to continue in the Rio+20 Trend scenario, leading area. This trend is expected to continue in the Rio+20 Trend scenario, leading to some further expansion of agricultural areas mainly for crops. However, as a reflection of the slow-down in global population growth around 2050, global land-use expansion will become stable towards the end of the scenario period.

According to FAO (Alexandratos and Bruinsma, 2012) the total agricultural area in China will decrease towards 2030 and 2050, mostly because of urbanisation. Due to the uncertainties about the future expansion or contraction of agricultural area in China, we used a constant arable land area from 2010 onwards for future estimations.

5.3. Results

We present the P budgets for the past and future four decades (1970-2010-2050) for the provincial (section 5.3.1), regional (section 5.3.2) and national scale (section 5.3.3).

5.3.1. Provincial P budget

The agronomic P budget of the arable lands in 1970 showed a P deficit or a small P surplus in provinces such as Tianjin in the North, Jilin in the Northeast, Jiangsu in the Yangtze Plain, Shanxi in the Northwest and Zhejiang in South China (Figure 5-2A). Since 1990 increasing P application as mineral fertilizer and manure led to positive soil P budgets in all provinces (Figure 5-2B). In 2010 the P surpluses reached more than 70 kg P ha^{-1} yr^{-1} in Sichuan, Guangdong, Shandong, Guangxi, and Hainan provinces. Hubei province in the Yangtze Plain showed the highest P surplus with 90 kg P ha^{-1} yr^{-1} in 2010 (Figure 5-2D).

5.3.2. Regional P budget

The divergence in incremental use of P fertilizer between the different regions of China is evident (Figure 5-3A). In 1970, P application was around 10 kg ha^{-1} in North, South and Yangtze Plain, 6.0 kg ha^{-1} in the Northeast and only 3.0 kg ha^{-1} in Northwest. Since then, it increased almost five-fold in South and Northeast and more than ten-fold in the Northwest, the Yangtze Plain and the North (up to 100 kg ha^{-1}) (Figure 5-3A). During the 1970-2010 period, P uptake in Northeast and Northwest regions remained less than 10 kg ha^{-1}. In the South, North and Yangtze plain, P uptake reached 17, 21 and 24 kg ha^{-1}, respectively, revealing that crop uptake does not respond proportionally to the higher rate of P inputs (Figure 5-3B).

Phosphorus surpluses increased dramatically in North, South and Yangtze plain between 1970 and 2010. Between 2000 and 2010, the surplus of P in the North and Yangtze Plain regions increased from ca. 50 kg P ha^{-1} to ca. 76 and 64 kg P ha^{-1}, respectively (Figure 5-3 C).

Using the Rio+20 Trend scenario for target P uptake for the five Chinese regions, the P application simulated by the DPPS model can go down as illustrated in Figure 5-3A. Figure 5-3C shows the agronomic P budget in the different regions from 1970 to 2050. The levels of application and uptake do not result in an equilibrium fertilization scheme, as the level of residual P is not as large as in industrialized regions such as Western Europe. Hence, the model predicts slightly positive P budgets in 2050 (Figure 5-3C).

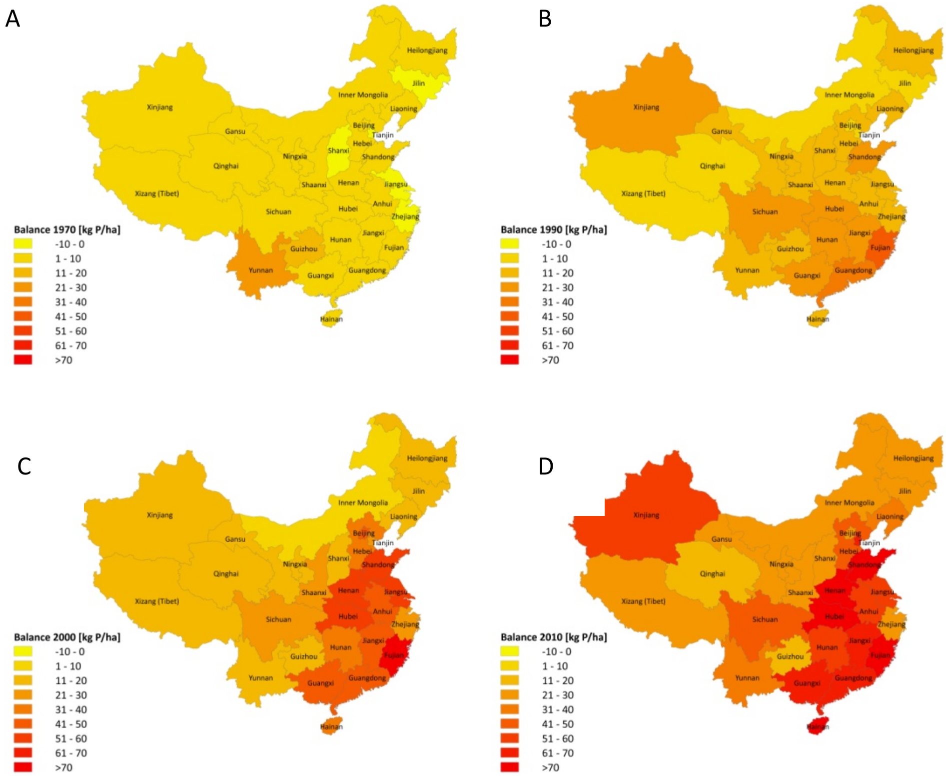

Figure 5-2 The annual agronomic phosphorus budget in different provinces of China in A)1970; B)1990; C) 2000 and D) 2010 (Province of Sichuan includes the municipality of Chongqing). The agronomic P budget is the difference between P inputs from mineral fertilizer and manure and P withdrawal through crop P uptake (kg P ha^{-1}). The light yellow color refers to P deficit and the dark red color refers to P surpluses.

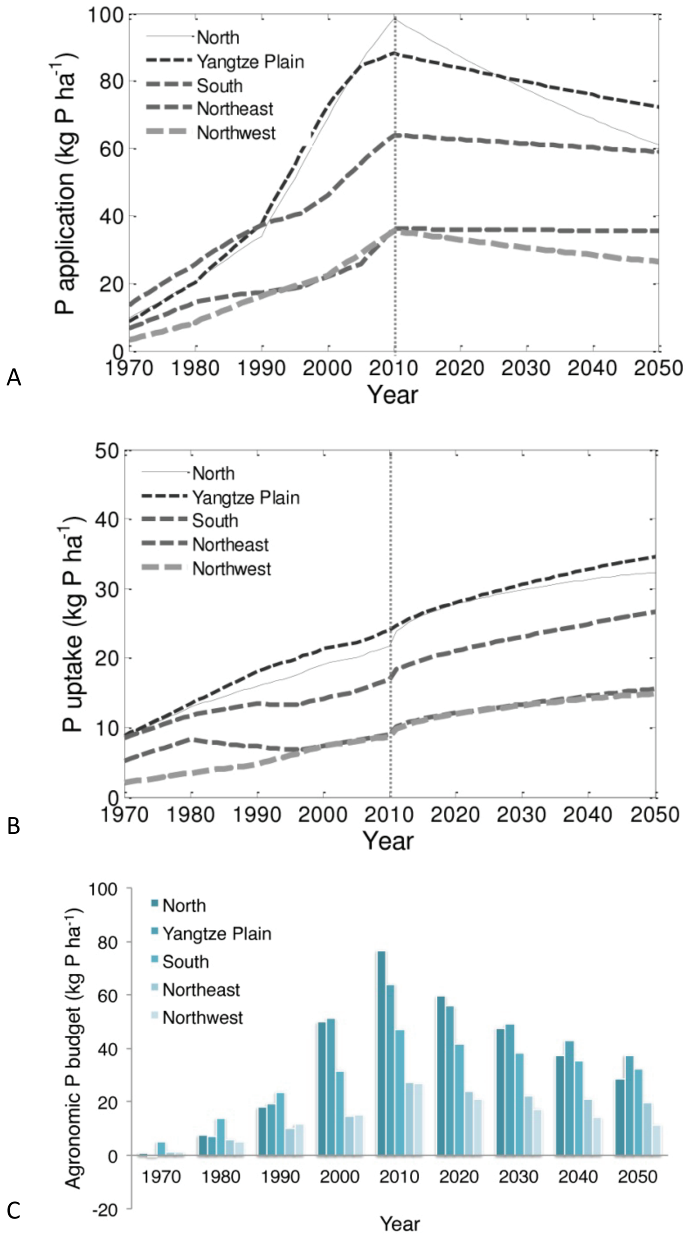

Figure 5-3 Trends of P application and P uptake. (A) Trends of annual P application (mineral P and manure), (B) Crop P uptake and (C) Agronomic phosphorus budget in different regions of China's croplands for the period 1970-2010 (historical data) and 2011-2050 (model).

The regional differences are mainly related to differences in production systems (traditional mixed smallholder versus the decoupled crop and landless, animal feeding operations), fertilizer management practices and differences in soil fertility. Furthermore, the type of livestock and its distribution has a large effect on P use efficiency and P surpluses. For example pig and poultry production have higher P use efficiency than beef and sheep production (Ma et al., 2012).

5.3.3. National P budget

Comparison of P application and P uptake between 1970 and 2010 in China's croplands showed that P input is much larger than crop uptake. Crop yield in China increased only about 2.5 times between 1970 and 2010, while the use of mineral P increased 7.5 times. P use efficiency in crop production (defined as the P uptake as a percentage of P application) decreased from about 88% in 1970 to only 25% in 2010 (Figure 5-4). The value for 2010 is lower than the 36% for 2005 from a recent study (Ma et al., 2010; Ma et al., 2012) because fertilizer P use still increased between 2005 and 2010.

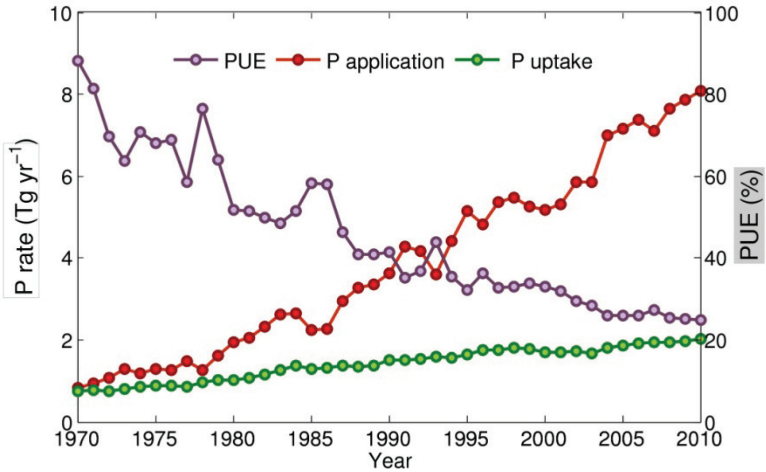

Figure 5-4 Historical P application, P uptake and P use efficiency (PUE) during 1970-2010 in China.

In 2010, China's mineral P fertilizer consumption exceeded 6.3 Tg (NBSC, 2010), with an additional 1.8 Tg P from animal manure spreading in croplands. At the same time P in grain yield was only 2.0 Tg, out of the 8.1 Tg supplied by fertilizer and manure. Consequently, the estimated total P surplus is about 6.1 Tg, which is equivalent to the mineral P fertilizer input in 2010.

5.4. Discussion

China's production of phosphate rock was 68 Tg in 2010 (corresponding with 8.7[1] Tg P) ,which was more than twice the amount mined in 2005 (30.4 Tg) (USGS, 2012). Total global production of phosphate rock was 181 Tg in 2010 (corresponding with 52.4 Tg P_2O_5 and 23 Tg P). Hence, the P surplus from synthetic fertilizers in China's cropland (4.3 Tg) (See section 5.3.2) was about 50% (USGS, 2012) -60% (FAO, 2011) of the China's total P production and around 20% of global P production in 2010. The total P surplus through mineral P fertilizer application in China's cropland for the period 1970-2010 was 56 Tg – more than 3.5 times the total mineral P application in Africa's croplands (15.3 Tg) during the same time period (Sattari et al., 2012) and more than twice the total mineral P production on the globe in 2010 (USGS, 2012).

The total accumulated P in the labile pool in China's cropland since 1970 is estimated to be 44 Tg P in the year 2010 in China's cropland corresponding to 345 Tg phosphate rock. This amount is almost five times the amount of Chinese phosphate rock production in 2010 and around one tenth of Chinese estimated phosphate rock reserves, which is equal to 3700 Tg in 2010 (USGS, 2012).

Estimates of future P fertilizer demand accounting for residual soil P (Sattari et al., 2012) were made with the DPPS model; DPPS calculates P application rates (fertilizer and manure) based on target crop production in the Rio+20 Trend scenario with the projected population of 1.415 billion people in China by 2050 (van Vuuren et al., 2012). This scenario turns out to be closer to the actual trends than any of the other Rio+20 scenarios assuming a deviation from current trends. Population and income growth are projected to lead to the strong demands for food and energy by 2050 (van Vuuren et al., 2012). According to the Rio+20 scenario, P uptake in China needs to increase by 60% between 2010 (2.0 Tg) and 2050 (3.2 Tg) to satisfy the 2050 food demand.

In this paper we show that residual soil P plays a crucial role to sustain the indicated P uptake by 2050 in China. Crop production can benefit from the P surplus from past P fertilizer and manure use. Theoretically, increased P uptake can be achieved even with a reduction in P application (Figure 5-5). The reduction trend in P application has been confirmed with the conceptual model of Li et al. (2011). When soil available P (Olsen-P) is sufficient, only low rates of P fertilizer need to be applied to obtain good crop yields. Readily

[1] Units of phosphorus (elemental P) are calculated assuming 29% P_2O_5 in phosphate rock and 44% P in P_2O_5.

available P in the soil solution, which in DPPS is a part of labile pool, provides most of the plant-available P. The two main factors that control the availability of P to plant roots are the concentration of phosphate ions in the soil solution and the P-buffer capacity, i.e. the ability of the soil to replenish these ions when plant roots remove them (Syers et al., 2008). A critical concentration of readily available P must be maintained to obtain good crop yields.

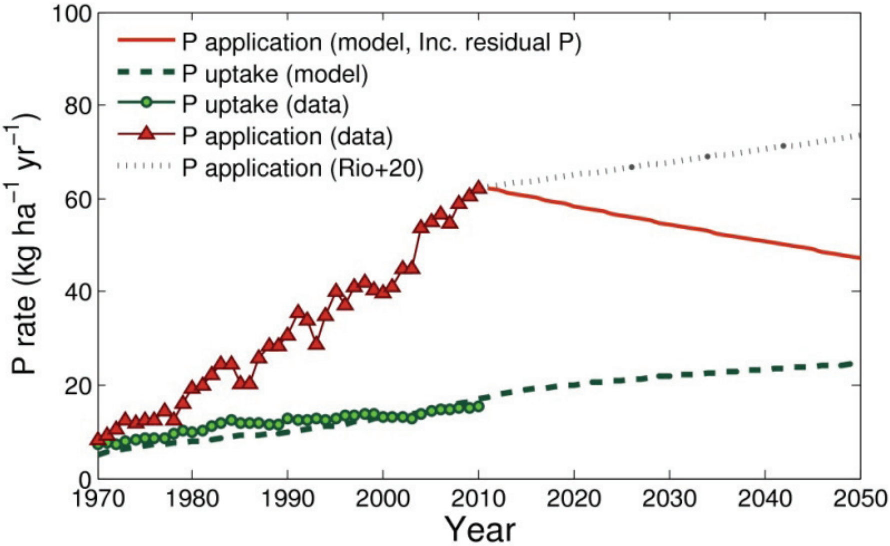

Figure 5-5 Trends of annual P application and P uptake in China's cropland for the period 1970–2050. Red triangles and green circles refer to P application and P uptake rates based on historical data, red solid and green dashed lines refer to P application and P uptake rates, simulated with the DPPS model accounting for residual P. The dotted grey line shows P application taken from the Rio+20 Trend scenario and does not account for residual P.

The importance of residual P has already been demonstrated through different experiments. For example a farm-scale study showed that equilibrium P fertilization between 1989 and 2006 did not lead to reduced crop yields in Dutch sandy soils (Verloop et al., 2010). Another national-level study on P fertilizer in Japan indicated that crop yields remained constant or even increased, despite a decline in use of P fertilizer and manure between 1985 and 2005 (Mishima et al., 2010). Even in high P-fixing soils a large initial application of P (around 600 kg ha^{-1}) can be adequate for cultivating maize for 7 to 9 years due to the effect of residual P (Kamprath, 1967).

To quantify the crucial impact of residual P for China at the country scale, our modelling results are compared with estimated P applications from the Rio+20 Trend scenario (Figure 5-5). The cumulative P application in the original Rio+20 study between 2010 and 2050 is 353 Tg P (van Vuuren et al., 2012) and the estimated P required based on the DPPS model simulations is only 283 Tg. The total amount of P that can be saved by accounting for residual soil P over the 2010-2050 period is thus about 20% (70 Tg) of the Rio+20 scenario estimate. The difference between Rio+20 and our calculations would be equivalent to more than twice the cumulative P required between 2010 and 2050 in Oceania (31.7 Tg), half of the P required in Africa (143 Tg) or sufficient for Western Europe (56.4 Tg) or Eastern-Europe (44.7 Tg) to reach their target P uptake in 2050 (Sattari et al., 2012).

The simulated trend in Figure 5-5, i.e. increasing P uptake and declining P application rates cannot continue over prolonged periods, since soil P stocks would deplete leading to soil degradation. Ideally, P withdrawal and runoff losses need to be balanced by P inputs (Sattari et al., 2012). In contrast, in soils that have been depleted due to minimal P application rates in the past, P input has to exceed crop requirements by 30-50% during a period of 30-50 years to build up a good soil P status (Steen, 1998).

Excessive P application through mineral fertilizer in the arable farming system has led to the accumulation of soil P and constitutes an eutrophication risk for surface water. In contrast, Chinese grazing systems may be confronted with a severe P deficit problem due to the massive transfers of P in the form of manure, while grasslands are hardly fertilized. An efficient way to solve these two problems (P excess in cropland, P deficit in grassland) simultaneously may be through regulating and balancing the P flows between the two systems (Chen et al., 2008).

Leaching of P may be a significant component of P losses to waterbodies from agricultural soils with low P sorption capacity and that have experienced large cumulative surpluses (Sims et al., 1998). Based on the P budgets for the past 40 years, soils may become P saturated (at the critical value of 25% saturation) in less than 10 years in poorly drained soils with high surplus of P in North, South and Yangtze plain, while the time needed to reach saturation is longer in well drained soils and where less P has been applied (See Section SI and Table S5-2).

Many industrialized countries have had periods with excessive P fertilizer use, for example in the 1970s parts of Europe were heavily over fertilized (FAO, 2011). However, since the 1980s in much of Europe, P application rates have been decreasing, and uptake continues to increase due to the use of plant-available P from the residual soil P pool (Sattari et al., 2012) as well as optimized agricultural systems. Efficient use of P in Western European countries could be considered as an example for other countries, such as China, that face the agronomic P surpluses challenge (Hvistendahl, 2010). Such a shift in nutrient management in crop production systems to stimulate the efficient use of P, will only be feasible when “smart cooperation” (Ulrich et al., 2013) occurs between all sectors along the P value chain. We argue that further reforms are needed in governmental policy, ultimately leading to an integrated nutrient management policy based on three pillars: food security and farmers’ income, environmental sustainability, and resource use efficiency. Implementing integrated nutrient management in practice requires a mix of policy instruments including education, demonstration, regulations, and targeted economic incentives (Li et al., 2013).

5.5. Conclusions

As a result of residual soil P built up in the past four decades in China, crop production targets can be achieved in combination with a reduction of soil P surpluses through reducing P application. This brings the concept “Less input, more output” (Hvistendahl, 2010) into practice. Such shifts have been observed in other regions such as in Europe. Reduced nutrient inputs will be agronomically, economically, and environmentally beneficial. Using our quantitative model approach we illustrate the undeniable key role of China in managing the global P crisis if policy makers adopt a proper and sustainable P application strategy. Such improvements will only be achieved when “smart cooperation” occurs between all sectors along the food production-consumption chain and also different countries.

5.6. Supplementary Information

5.6.1. Manure use

Total P in manure production within pastoral, mixed and industrial systems is calculated from the animal stocks and P excretion rates. We used P excretion rates per head for dairy and non-dairy cattle, buffaloes, sheep and goats, pigs, poultry, horses, asses, mules and camels based on various sources (Midwest_Plan_Service, 1985; Sheldrick et al., 2003a; Smith, 1991; Van Horn et al., 1996; Wilkerson et al., 1997). We used constant excretion rates per animal categories, so that the P excretion per unit of product decreases with increasing milk and meat production per animal.

For each country, animal stocks and P in the manure for each animal category are spatially allocated between mixed and pastoral systems. The distribution of livestock production over these systems is based on the Integrated Model to Assess the Global Environment (IMAGE) (Bouwman et al., 2005b) for 1970-2005, and for the period 2005-2050 the distribution over these systems is provided by the Rio+20 study.

Table S5-1 Excretion rates per animal categories in kg P per year.

Animal category	kg P year^{-1}
Beef cattle	7.0
Dairy cattle	10.5
Buffaloes	7.0
Pigs	1.8
Poultry	0.1
Sheep and goats	1.4
Horses	6.5
Asses	4.3
Mules	4.3
Camels	7.9

Within each system, the manure is distributed over different management systems, i.e. (i) grazing or excretion in the meadow or field (P_{gra}), (ii) storage in animal housing and storage systems (P_{sto}) and (iii) manure not recycled in agriculture (P_{out}). P_{out} includes manure excreted outside the agricultural system, for example in urban areas, forests and along roadsides or manure collected in lagoons or discharged to surface water (Bouwman et al., 2005b), and manure used as fuel or for other purposes (Mosier et al., 1998).

Total P excretion P_{exc} is thus:

$$P_{exc} = P_{gra} + P_{sto} + P_{out} \qquad (5\text{-}1)$$

Animal manure available for application to crops and grassland (P_{man}) includes all stored or collected manure. The input of manure for the soil budget therefore excludes P_{out}

$$P_{man} = P_{exc} - P_{out} \qquad (5\text{-}2)$$

We assumed that in most industrialized countries, 50% of the available animal manure from storage systems is applied to arable land and the remainder to grassland (Lee et al., 1997). In China, 99% of the available manure is assumed to be applied to cropland and 1% to grassland, thus accounting for some stubble grazing and manure excretion in croplands, and the lower economic importance of grass compared to crops in developing countries (Seré and Steinfeld, 1996).

5.6.2. Manure not recycled in agriculture

A significant amount of animal manure is not recycled in agriculture. In 2005 at global level, this amounted to 12% of all manure excreted in animal houses or stored manure; this amount was 30% of the manure from pig production. The fate of this manure is not well known. In 2050, the fraction of all manure (from all animals) not recycled increases to 16% of all stored manure in the Rio+20 scenario, which is a result of a rapid increase of industrial pork production in large confined animal feeding facilities relative to beef, milk and poultry production. This implies, theoretically that considerable amounts of additional P are available to be integrated in Chinese crop production systems.

In the Decentralized Solutions Pathway of the Rio+20 study the impact of such changes in production systems and manure management has been analysed. This scenario considers better integration of manure in crop production and recycling of human excreta in agriculture, and leads to a reduction of global P fertilizer use by 20-30% compared to the baseline scenario used in this study. Major reductions are projected in industrialized and transition countries like China, and increasing fertilizer use in many developing countries to restore soil fertility.

5.6.3. P saturation time in soils

The ability of soil to store P is limited in soils with high P surpluses due to the very high P inputs. The degree of phosphate saturation (DPS) of soils has been identified as a potential phosphorus loss risk indicator (Breeuwsma et al., 1995). A critical value of DPS is defined as the saturation percentage of soil binding capacity that should not be exceeded to prevent the leaching of P and negative impacts on water quality. In the Netherlands, a critical DPS of 25% or more has been established, above which the potential for P losses through runoff and leaching are greater (Breeuwsma et al., 1995). A simple approach (Behrendt and Boekhold, 1993) was used to calculate the required time (years) to reach P saturation (t_E) in different regions in China with different historical P input and output.

$$t_E = R/(P_I - P_O) \quad (5\text{-}3)$$

Where P_I denotes the P input and P_O the P output. R stands for the surplus P at which the soil is considered to be saturated critically (reaching to 25% DPS). Saturation surplus depends on different parameters such as soil type and ground water table depth. In poorly drained soils with a shallow water table (20-40 cm below surface), the soil can become saturated with P at a surplus of 350-800 kg P ha^{-1}, while in moderately drained and well-drained soils with ground water table at 50-100 cm depth, soil saturation capacity is reached at a surplus of 800-1600 kg P ha^{-1} (Breeuwsma et al., 1995). We assumed three degrees of soil drainage: poorly drained, moderately drained and well drained with ground water depth of 20, 50 and 100 cm, respectively. Corresponding P surpluses that can be applied to these soils before they reach the critical P saturation level are 350, 800 and 1600 kg P ha^{-1}. The forty year (1970-2010) P budget was used in Equation 5-3 to calculate the time needed to reach the P saturated soil for each province (Table S5-2). We did not consider the possible impacts of adoption of new P fertilizer recommendations. Table S5-2 shows that particularly poorly and moderately drained soils in some regions are likely to be saturated, resulting in P leaching.

Table S5-2 Phosphorus saturation time in different regions and provinces of China’s cropland.

Region	Province	Saturation time [year]			Total surplus 1970-2010[Tg P]
		Poorly drained	Moderately drained	Well drained	
North	Beijing	22	50	99	0.2
	Tianjin	20	46	92	0.3
	Hebei	15	34	68	6.3
	Shandong	12	26	53	9.2
	Henan	10	22	44	11.5
South	Zhejiang	24	55	110	1.3
	Fujian	7	16	31	2.4
	Jiangxi	11	25	50	4.0
	Hunan	14	32	64	4.3
	Guangdong	11	25	51	3.9
	Guangxi	11	24	49	5.3
	Hainan	13	30	60	0.7
	Sichuan*	14	32	64	8.1
	Guizhou	18	40	80	2.2
	Yunnan	18	40	81	3.7
Yangtze Plain	Shanghai	23	53	107	0.2
	Jiangsu	14	32	63	5.2
	Anhui	13	30	61	6.1
	Hubei	9	22	43	7.2
Northeast	Liaoning	28	64	129	1.9
	Jilin	74	168	336	1.1
	Heilongjiang	21	48	96	5.4
Northwest	Shanxi	29	67	134	1.9
	Inner	56	127	255	2.2
	Xizang (Tibet)	45	102	204	0.1
	Shaanxi	22	51	103	2.9
	Gansu	24	55	109	2.5
	Qinghai	36	82	164	0.3
	Ningxia	32	74	148	0.4
	Xinjiang	16	35	71	3.0

* Including Chonqing municipality after 1995

Figure S5-6 Distribution of five regions in China (courtesy of Guohua Li).

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6

General Discussion

The main goals of this research were to: a) assess the long-term P application and historical P withdrawal by harvested crop and grazed grass using long-term data in different world regions, b) calculate future global P fertilizer demand for food production from cropland and grasslands via animals in different world regions, and c) estimate the amount of P that can be saved through a more sustainable use of P. These goals were achieved by analysing long-term data of P application through mineral P and manure P and P withdrawals, history matching, and then estimation of the future P demand by choosing and applying an appropriate model-based approach.

In assessing the sustainability of the P cycle, focus should be on the adequacy and maintenance of soil P rather than on the conservation of P rocks (Dumas et al., 2011). To my knowledge most of the studies that have already raised concerns about the P depletion and future of global P issues, emphasized the amount and availability of global reserves of P rather than the amount of P that humanity needs in agriculture and food production to meet the world's population P requirements in the next decades. Thus, this thesis provides initial insights into global P needs for food production from crop and livestock productions. The major findings of this research are addressed in the following sections. First, in Section 6.1 I discuss what models are most appropriate for our purposes at global scale. Next, Section 6.2 addresses the integrated phosphorus demand in cropland and grassland. Outcomes of this thesis for policy makers are formulated in Section 6.3 and the contributions of this thesis to the global P debate are presented in Section 6.4. Finally in Section 6.5 those areas where this research can be refined or extended are discussed.

6.1. Modelling soil P-crop at different scales

Soil-crop models have been developed for different scales ranging from the field scale to the regional or even global scales. This thesis covers the field and farm scale in Chapter 2, regional and country scale in Chapter 5, and continental and global scale in Chapters 3 and 4. In Chapter 2 the QUEFTS model was applied to assess NPK interactions in soil-crop systems. In Chapters 3, 4 and 5, the DPPS model was used, a model that assumes that P is the only limiting factor in crop growth and other nutrients and limiting factors are assumed absent. Analyses were performed at yearly basis for prolonged time periods of 50 to 100 years. To provide the required data for studies in the different chapters substantial data analysis and acquisition were done. Data aggregation, disaggregation and modification of model parameters (Ewert et al., 2011) are the scaling methods used in this research.

Modelling crop production in response to soil fertility and nutrient inputs at the global scale is challenging. Heterogeneity of climatic conditions, soils, crops and crop management makes it almost impossible to develop simple, yet adequate models for this broad range of conditions. Thus, an important task was to find an appropriate model for assessing global P requirements for future food production.

I reviewed most of the available soil-crop models, designed for analysing the P status in soil and its interaction with plants (Chapter 1). Crop production models can be grouped into empirical, semi-empirical, and mechanistic ones. Empirical models directly employ a relationship between variables and outputs without description of fundamental (physical) processes. These models usually are site-specific (Smaling and Janssen, 1993). For a given scale, mechanistic models are often more complex as they describe known physical interaction in crops-soil systems, and usually require detailed data on soil and crop characteristics, which make them less appropriate for application at large scales. Semi-mechanistic (or semi-empirical) models are relatively simple and are usually employed to predict yield responses as a function of soil fertility and fertilizer applications (Reid, 2002). Such simpler models with lower input data requirements are more appropriate at larger scales.

Chapter 2 tests and discusses the QUEFTS model that calculates yield responses to NPK fertilizer applications in relation to soil fertility (Janssen et al., 1990). QUEFTS considers three major nutrients (N, P, and K) and calculates final yields based on their availability. I found that using QUEFTS it is impossible, however, to deal with heterogeneity in soil properties in the field, and with variation in weather conditions from season to season. Major problems encountered in the use of QUEFTS were related to availability of data on soil properties and fertilizer field trials, irrigation and extreme soil pH values.

For a proper testing of QUEFTS substantial information such as nutrient accumulation and dilution, harvest index and soil chemical properties are required. For example, Olsen P is a critical data that should be known. But only in 1147 soil profiles distributed with a limited spatial coverage P-Olsen data have been measured (Batjes, 2010). For the implementation of QUEFTS, and for (re-) calibration of its soil and crop related parameters, use of complete treatments of control, N, P, K, NK, NK, PK, and NPK are needed. However, such experimental data are very scarce.

Some physical processes are not included in the model, which make this model less applicable for different world regions across the globe. For example, flooded systems cannot be simulated directly by applying the original parameters of the model. Because under oxygen-deficient conditions several soil chemical processes change, and thus the solubility of especially P increases. Because QUEFTS was originally developed in the humid tropics where most soils have a rather low pH, the pH correction factors had a weak validity at higher pH values, and were not at all applicable at pH values above 7. The empirical parts of the model do not allow QUEFTS to become a generic model with global applicability, capturing all combinations of soils, climates and management conditions.

As the QUEFTS model is a static model, it cannot capture the dynamics of soil nutrients; specifically the dynamics of soil P in the different pools and its accumulation as the residual P cannot be estimated. In contrast, the model of Wolf et al. (1987) is a dynamic model that calculates P transfer between the labile and stable P pools and fertilizer requirement for a target yield.

Phosphorus in soils occurs in many different forms and it is possible to extend the Wolf et al. (1987) model with more pools (available, labile and stable). However, increasing the complexity of the model may not necessarily improve the model predictions, and will make calibration more difficult. Bhogal et al. (1995) used a modelling approach to investigate the effects of residual P fertilizer in a field trial from 1978 to 1990. They concluded that a simple model with two pools was adequate to describe the effect of residual P on yield production. Thus, for a global-level study, a relatively simple model with fewer calibrating parameters seems feasible and adequate.

The Wolf et al. (1987) model (named DPPS in Chapter 3) was designed to simulate the long-term response of crop production to P inputs assuming that crop yields are limited only by P supply.

In the original model (Wolf et al., 1987) it was assumed that crop P withdrawal can be partly replaced by a net input of P, which was defined as P additions by weathering and rainfall, volcanic dust, and flood water, minus losses, mainly through soil erosion and leaching. However, in the modified version of the DPPS model (Sattari et al., 2012) different sources of P input to the system were distinguished separately. It means that P inputs through weathering and atmospheric depositions were allocated to the labile and stable pools, respectively. The total amount of mineral fertilizer and manure was allocated to the labile (80%) and stable (20%) pools and P uptake and P in runoff were distinguished as the P outflows from the system.

In the modified model (DPPS) we accounted for the expansion of cultivated land. Each year, the initial conditions (with no fertilizer history) are assigned to the new, additional area and the crop P uptake in each segment is calculated based on the history of that segment and finally weighted by its corresponding share of the total area. This allows us to differentiate productivity and soil contents of residual P for different parts of the total area considered. With these modifications, the DPPS model was used to simulate the long-term historical P uptake by crops for time series of P inputs, and to estimate the future P inputs for a specific future target P uptake. Model simulations closely fitted historical P uptake for all continents and the entire globe as presented in Chapter 3. Subsequently, DPPS was implemented to assess soil P budgets and residual soil P in global grasslands and to predict the future demand of P as discussed in Chapter 4. In Chapter 5, the DPPS model was applied to assess soil residual P and future P demand for national and provincial scales in China's croplands.

In comparison with QUEFTS, the DPPS model can also be formulated in a target-oriented approach (Van Ittersum and Rabbinge, 1997), in which the (future) P uptake is a model input and the P application a result. Since the DPPS model is a dynamic model, it calculates the amount of P that is accumulated in the soil - due to the low recovery of P fertilizer – and considers this amount of residual P in the calculations of the required P for a target yield.

6.2. Integrated cropland and grassland phosphorus demand

Chapters 3 and 4 show that a large part of the P in animal manure spread in croplands as fertilizer originates from grasslands. In opposite direction, there is a P transfer from croplands to grasslands through livestock feed. However, these two transfers are not balanced at the global scale, which leads to soil P depletion in grasslands. While Chapter 3 shows that future P requirements from fertilizer on cropland would increase less than the increase in crop production, on grassland the opposite is true (Chapter 4).

6.2.1. Global P budget in croplands

Historical data analysis presented in Chapter 3 showed that in Europe and some other regions of the world, soil P status in croplands has been improved over the past decades by applying P fertilizer and manure. Build-up of soil P fertility as a result of substantial inputs of organic and mineral P fertilizer in the past in different croplands across the world (Europe, Asia and Latin America) has reduced the P inputs required, although P uptake by crops has stabilized or continued to increase. Thus, P application rates have declined while P uptake continued to increase. The results indicate a hysteretic behaviour in crop P uptake versus fertilizer application, which is well pronounced in some world regions with large historical P applications. For the same P application rate, two different P uptake rates were observed at different moments in time. The higher uptake rate is the result of the residual P that has been accumulated in the soil (legacy P). The P budget analysis in croplands shows that accounting for legacy P in estimating the required fertilizer leads to a lower fertilizer requirement and thus decreases the future P demand compared with other studies that ignored the effect of residual soil P. A commentary paper (Townsend and Porder, 2012) argues that such optimism needs to be tempered. They wrote: “Sattari *et al.*’s (2012) analysis creates optimism that present and future crops can recoup some benefits from the mistakes of our past. Yet, while information on accumulated soil P ought to inform decisions about fertilizer needs, there are several reasons to temper our optimism. History also teaches us that, in addition to crop requirements, a web of biophysical and socioeconomic factors drives agricultural decisions and outcomes.” Townsend and Porder (2012) argue that for example farmers often apply too much fertilizer to maximize the production, and consequently high rates of erosion and runoff cause eutrophication. In addition, extreme climate events can make the situation even worse. They also mentioned that since biogeochemical cycles are coupled, P does not act in isolation. Thus, greater understanding of the interactions between C, N and P cycles is needed, through the development of models that could help in understanding feedbacks that play out at multiple time scales.

Chapter 2 considers the interactions between N, P and K. However, since this NPK model is not suitable for global application primarily due to data limitations, soil P budgets and soil P status were addressed with the P model DPPS; by reproducing long-term historical data all aspects mentioned by Townsend and Porter (2012) are addressed, including over use of fertilizer by farmers, runoff and erosion and also climate variability. Considering the results shown in Chapter 3, we propose that both farmers and policy makers could learn from the past to plan for future. For example, the history of P application in the Netherlands demonstrates awareness of farmers about the importance of residual P, as they used much lower amounts of fertilizer and manure recently (Oenema, 2013; Verloop et al., 2010).

What has been happening in the Western Europe as to P application and P production can be a lesson for other countries or regions that are facing a soil P saturation problem. For example in Chapter 5 I addressed this issue at a national case study in China's croplands. The results show that P application exceeds P uptake in China's croplands as a result of the high P input from the past and current applications. This country can employ the same approach as used in Western Europe by reducing fertilizer and manure P inputs. This brings the concept "Less input, more output" (Hvistendahl, 2010) into practice, which has several socio-economy and environmental benefits.

Chapter 3 highlights that global P application rates of croplands will need to increase, but less than proportional to the required increases in crop production. But the situation differs among different regions. In Europe, Asia, Latin America and Oceania crop production could benefit from the residual P and these continents could reduce their P application rates between 2008 and 2050. In contrast, in Africa as the continent with the largest P reserves, soils are very poor and have been depleted steadily over the years due to the low rate of P inputs. Thus, more than a five-fold increase in P application is needed to achieve the target P uptake in this continent in 2050. This result is consistent with conclusions of Steen (Steen, 1998) who suggested that 30-50% more P fertilizer than crop requirements must be applied for almost 30-50 years to restore soil P in depleted soils. Finally, we estimated that 1200 Tg P application is required in global croplands for the 2008-2050 period. This amount includes P from both mineral fertilizer and animal manure. However, a large part of the P in animal manure that is spread in cropland originates from grasslands.

6.2.2. Global P budget in grasslands

Increasing grass demand as a consequence of the yearly increase of meat production (0.5-4.0%) (Alexandratos and Bruinsma, 2012), led us to formulate Chapter 4 to analyse the P demand for projected future grass production, and associated sustainability aspects regarding phosphorus. To address these issues quantitatively and to show the importance of P transfer from grasslands to croplands, and to estimate the total amount of required P in 2050 for both cropland and grassland we analysed the global P budget under grassland as presented in Chapter 4. We distinguished three sources of soil P inputs:

manure generated inside the grasslands and spreading in the grasslands, manure produced by pig and poultry outside the grassland boundaries and mineral P fertilizer. Soil P erosion and grass P uptake were considered as grassland's soil P outflows. Manure generated inside the grasslands was divided in three parts: spreading in grasslands soil, spreading in croplands and other uses of manure (Figure 6-1).

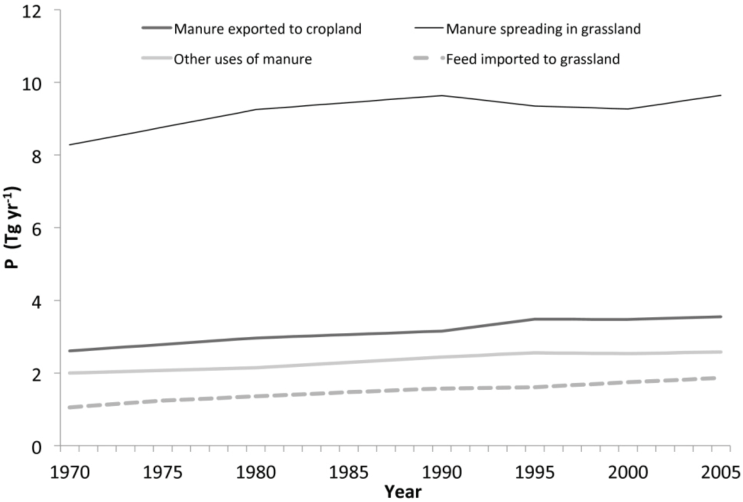

Figure 6-1 A global partitioning of manure P originated in grasslands and imported feed P from cropland to grassland. Manure partitioning includes manure spreading in grassland, manure exported to cropland, and other uses of manure (more information in Chapter 4).

Our analysis indicated that manure was the most important source of P for grassland soils worldwide. The world´s grassland soils received 3.0 kg P ha^{-1} yr^{-1} from manure on average from 1970 to 2005, more than ten times the P input to grasslands through fertilizer (Figure 6-2). Inorganic P fertilizer use in grasslands has been minimal in many regions in the past. A total of 23 Tg of mineral P was globally applied to grassland soils over the period of 1970-2005, which is in the same order of magnitude as the annual use of P fertilizers in croplands, i.e. 18.5 Tg of P yr^{-1} in the single year 2010 (FAO, 2014). In addition, P fertilizer use in grasslands was highly unbalanced across the regions. Eastern and Western Europe alone accounted for almost 80% of the global total P fertilizer use (Table 6-1). With the positive P budgets in Europe there has been accumulation of residual soil P in grasslands in a similar way as reported for croplands in Chapter 3.

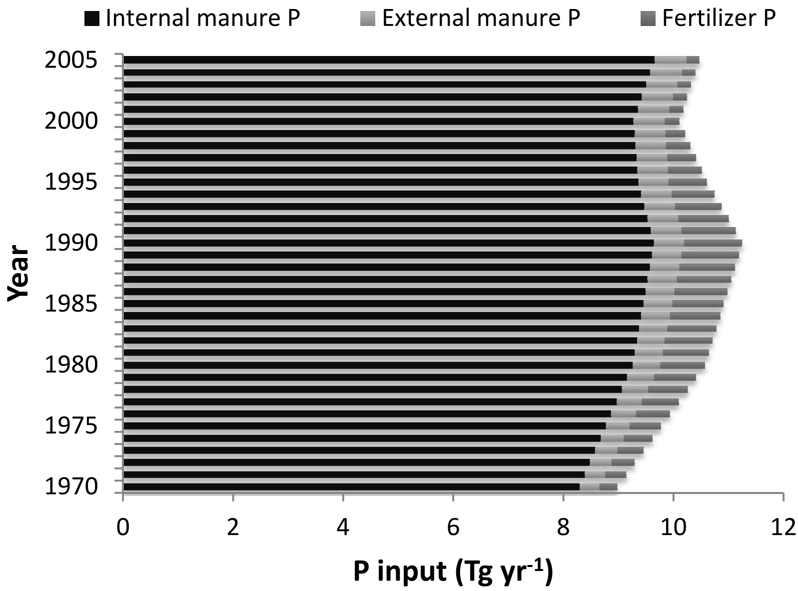

Figure 6-2 Historical soil P inputs in global grasslands through fertilizer and manure (Internal manure is the amount of P in the grassland-based livestock excretion that is actually returned to grassland soils. External manure input is the amount of P from non-grassland-based livestock excretion that is spread in grasslands as organic fertilizer).

Chapter 4 shows that the cumulative global manure originating from grassland and spread in croplands was 113 Tg of P for the 1970-2005 period. Asia alone was responsible for 44% of the global number, with a total of 50 Tg P between 1970 and 2005 (Table 6-1). In the 2005-2050 period, a net 10 Tg of P was transferred to cropland soils in Africa, 36.7 Tg P in Asia, 10.9 Tg in Latin America, 3.3 Tg in Western Europe and less than 2 Tg in Oceania. North America and Eastern Europe were the only regions that showed a cumulative net P transfer from cropland to grasslands systems, i.e. 1.9 and 0.4 Tg P, respectively. Note that a part of the feed P used in grassland-based systems is imported from other countries. Due to lack of data we ignored feed trade, but in regions such as Western Europe, P in feed produced in croplands of other world regions may be a significant part of total feed use. However, this did not affect the budget of the grassland-based systems, but may have led to underestimation of the net transfer from grassland to cropland within the importing region considered.

Spreading a large part of manure generated in grasslands on croplands has two consequences. On the one hand it helps sustaining and increasing crop productivity. On the other hand, reduced soil P fertility can deteriorate the status of those grassland soils, which are already degraded (Suttie et al., 2005). The transfer of P from grasslands to cropland is particularly important in developing countries (Bouwman et al., 2009), and contributes to the build-up of residual soil P in cropland.

The results indicated a continuous process of soil P depletion in grasslands during the past decades in most of the world's regions. At the same time, more grass is needed in the future, which can come from either expansion of grasslands or from increased productivity. Higher productivity is only possible if we improve the P status of soils under grassland. Given the increasing future demand for grass (Bouwman et al., 2005), additional fertilizer P will be required to maintain soil fertility in the world's soils under grassland. If the N supply will come (partly) from the biological N_2-fixation by grass-clover mixtures (or other grass-legume swards), even more P would be needed. Besides the mineral P fertilizer, adequate manure management practices will be another key element to reduce the net P withdrawal from grasslands. As shown in our study, grass uptake and erosion accounted for 536 Tg of P over 1970-2005 at global scale. Over the same period, P in manure excretion was 530 Tg P of which 63% stayed within the grassland system and hence 37% was exported from grassland (Table 6-1). Such large transfers imply severe loss of soil fertility in grassland systems. To avoid future unsustainable ruminant production, manure management systems are crucial determinants for the viability of grassland-based production, especially for poor farmers who do not have access to mineral fertilizers (Rufino et al., 2007).

Table 6-1 Cumulative manure and mineral fertilizer between 1970 and 2005, and their allocations to the different partitions

P Flow	Africa	Asia	East Europe	West Europe	North America	Latin America	Oceania	World
Grassland-based manure (Tg P)	77	196	41	50	51	95	20	530
Other uses of manure (Tg P)	1	81	1	0	1	0	0	84
Spread in croplands (Tg P)	12	50	11	12	13	14	2	114
Spread in grasslands (Tg P)	64	66	28	38	37	81	18	332
% Manure spread in grassland from grassland-based manure	84	33	69	77	73	85	90	63
% Other uses from grassland-based manure	<1	41	3	<1	2	0	0	16
% Manure spread in cropland from grassland-based manure	15	25	28	23	25	15	10	21
Mineral Fertilizer (Tg)	1	2	6	12	0	1	2	24
Grassland- and non grassland-based manure (Tg)	64	70	31	44	42	81	19	351
Total P inputs to grassland's soil (manure +mineral P) (Tg)	65	72	37	56	42	82	21	375
% Manure of the total inputs	99	97	84	79	100	99	90	94
% Fertilizer of the total inputs	1	3	16	21	0	1	10	6

6.2.3. Full picture of food production in 2050

Transferring nutrients from grasslands to croplands through livestock manure could help to redistribute nutrients in time and space. Lands that are less suitable for arable cropping could be grazed to produce manure. Spreading the manure on croplands, makes these lands more suitable for crop production (Delgado et al., 1999). However, given the increasing future demand for grass, grassland soils also need better management. To maintain the soil P status in grassland systems and meet the target grass production in 2050 and assuming no expansion of grasslands, we concluded that mineral fertilizer P for grasslands would be essential. Lack of fertilizer use in grassland may stimulate grassland expansion in future, accompanied by deforestation.

Our estimate of the global P input required in grassland of 1215 Tg P for the 2005-2050 period includes P from both mineral fertilizer and animal manure. Accounting for the contribution of P from animal manure (655 Tg) in global grasslands under the Rio+20 scenario, 560 Tg P in the form of mineral P fertilizer will be needed between 2005 and 2050. In 2050, globally 24 Tg yr^{-1} of mineral P fertilizer will be needed to avoid loss of soil fertility and declining grassland productivity. In cropland, excluding the share of manure P (32%) from the total required P, the global mineral P was estimated to be 20.8 Tg in 2050. Thus, the amount of mineral fertilizer P needed in cropland and grassland systems in total is estimated to be 45 Tg in 2050, corresponding to 350 Tg phosphate rock. In the next four decades, a global cumulative amount of mineral P of 1380 Tg, including 820 Tg in cropland (Chapter 3) and 560 Tg in grassland (Chapter 4), is required to achieve global food production in 2050. This implies that, up to 2050, 10700 Tg of phosphate rock will need to be mined across the globe to produce the P fertilizer for the agricultural and food production sectors. This amount of phosphate rock is about 16% of the total phosphate rock reserves estimated in 2014 (USGS, 2014). Thus, the world will not face as rapid a P depletion as earlier thought. For the world population of 9 billion in 2050, the global average footprint of inorganic P is estimated to be ca. 5 kg per person in 2050.

6.3. Outcomes for policy makers

Phosphorus as a major limiting nutrient in agriculture is increasingly considered to be a new global sustainability challenge (Cordell and Neset, 2014). While the time scale of P depletion is debatable (Cordell et al., 2009; Dery and Anderson, 2007; Van Kauwenbergh, 2010; Van Vuuren et al., 2010), the “peak phosphorus” debates certainly raised public, political and scientific awareness of a formerly barely notified topic (Heckenmüller et al., 2014). As a result, different research based networks and initiatives such as the Global

Transdisciplinary Processes for Sustainable Phosphorus management (Global TraPs), the European Phosphate Platform and the Global Phosphorus Research Initiative (GPRI) have been founded, and the European Commission aims to publish a Green Paper on the topic. Nowadays, because agriculture is the prime P consumer and also having experienced significant price volatility, different sectors such as science, policy, agro-industry and other public stakeholders are increasingly concerned about the sustainable use of P (Heckenmüller et al., 2014; Scholz et al., 2013).

Phosphorus scarcity has different perspectives. Scholz and Wellmer (2013) propose that scarcity is relative and may depend on specific demand and knowledge. Physical scarcity refers to the physical availability of the world's phosphate rock reserves and is fundamentally different from economic scarcity. Economic scarcity refers to the lack of access to P because of financial limitations (Cordell and Neset, 2014). Past price fluctuations of P fertilizers seem not to reflect physical P scarcity but rather other demand and supply factors.

Unequal distribution of P production and reserves in the world can pose a threat to food security in soil P-deficient regions through factors such as the instability of fertilizer price. Consequently, a price shock has more severe effects on agricultural production in tropical countries than in the industrialized countries of the North with P saturated soils (Heckenmüller et al., 2014). Access to P fertilizer may be economically difficult for smallholder farmers, despite the urgent and increasing demands for improved soil P fertility to support increased agricultural production by Africa's farmers (as discussed in Chapter 3). Scholz and Wellmer (2013) concluded that in the next centuries P scarcity is more an issue of economic than physical scarcity. We will not probably run out of P, but prices will increase, which will trigger farmers to use less P.

Many industrialized countries have had periods with excessive P fertilizer use (which could be referred to as managerial scarcity), for example in the 1970s parts of Europe were heavily overfertilized (FAO, 2011). Since the 1980s, due to new legislations for the fertilizer application in much of Europe, P application rates have been decreasing. Controlling nutrient leaching as a cause of ground and surface water pollution was the prime aim of the legislations to reduce nutrient inputs into the agricultural soils. This act had a good side effect on managing P application. As historical data showed, even with decreased P application, crop production continued to increase due to the use of plant-available P from the residual soil P as well as optimized agricultural systems. Therefore, in those regions that face the agronomic P surplus challenge efficient use of P in Western European countries could be a good example.

Since different countries experience P scarcity in different ways, assessing the P vulnerability and adaptive strategies to increase the resilience of the food system to P scarcity at national scale would be very appropriate (Cordell and Neset, 2014). Therefore, in Chapter 5 we selected China as a case study for our further assessment. Among all countries, China has a strategic position in P debates since this country is the largest producer and consumer of P in the world. China as a country with a long history of agriculture, feeds 20% of the world's population on only 7% of the arable land (Li et al., 2011). Our estimations presented in Chapter 5 show that since 1970 the total amount of accumulated P in the labile pool in China's cropland was 44 Tg P in the year 2010, corresponding to 345 Tg phosphate rock. This amount is almost five times the amount of Chinese phosphate rock production in 2010 and around one tenth of the estimated Chinese phosphate rock reserves, which were equal to 3700 Tg in 2010 (USGS, 2012). The total P surplus through mineral P fertilizer application in China's cropland for the period 1970-2010 was 56 Tg – more than 3.5 times the total mineral P application in Africa's croplands (15.3 Tg) during the same period (Chapter 3) and more than twice the total mineral P production on the globe in 2010 (USGS, 2012). In Chapter 3, we showed crop production can benefit from the P surplus from past P fertilizer and manure use. Theoretically, increased P uptake can be achieved even with a reduction in the rates of P application. As stated in Chapter 5, the total amount of P that can be saved in China by accounting for residual soil P over the 2010-2050 period is about 20% (70 Tg) of the Rio+20 scenario estimate. We concluded that the difference between Rio+20 and our calculations for China would be equivalent to half of the P required in Africa (143 Tg) or sufficient for Western Europe (56.4 Tg) or Eastern Europe (44.7 Tg) to reach their target P uptake in 2050 (as shown in Chapter 3). The feasibililty of reducing the P application as suggested in Chapter 5 of this thesis has been also confirmed with the conceptual model of Li et al. (2011). They suggest when soil available P (Olsen-P) is sufficient, only low rates of P fertilizer need to be applied to obtain good crop yields. A farm-scale study also showed that long-term (1989–2006) equilibrium P fertilization did not lead to reduced crop yields in Dutch sandy soils (Verloop et al., 2010) .

Although excessive P application, besides other wastes (human sewage, detergents, etc) has caused eutrophication in surface water in China, grazing systems in China are confronted with a severe P deficit problem due to the massive transfers of P in the form of manure and absence of fertilization (Figure 6-3). An efficient way to solve the P excess in cropland and P deficit in grassland simultaneously may be through regulating and balancing the P flows between these two systems (Chen et al., 2008).

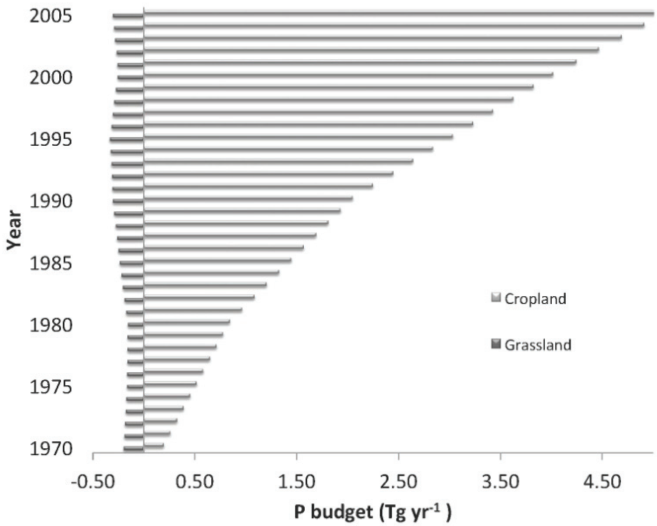

Figure 6-3 Long-term P budget in China's croplands and grasslands.

I suggest that such a shift in nutrient management in crop production systems to stimulate the efficient use of P can be feasible when cooperation occurs between all sectors along the food production–consumption chain (facing the institutional scarcity). Many simultaneous shifts in the agri-food systems such as dietary changes, reduction of losses across the food supply chain and efficient recycling can make a substantial contribution (Kahiluoto et al., 2013).

Using our quantitative modelling approach we illustrated the key role of China in managing the global P crisis if policy makers adopt a proper and sustainable P application strategy. It can be argued that further reforms are needed in governmental policy, ultimately leading to an integrated nutrient management policy based on three pillars: food security and farmers' income, environmental sustainability, and resource use efficiency. Implementing integrated nutrient management in practice requires a mix of policy instruments including education, demonstration, regulations, and targeted economic incentives (Li et al., 2013).

6.4. Contributions of this research

This research was designed to estimate the global P demand for food production in 2050. The thesis highlights the vital role of residual or legacy phosphorus in maintaining productivity in cropping and grassland systems with reduced P inputs and minimal P transfer from land to water. Main outcomes of this thesis are:

1. Recently, the scientific debate has focused on current production and use of phosphate rock rather than on the amounts of P required in future to feed the world. This thesis concentrates on adequacy and maintenance of reserves of P in soils and demonstrates that building up the soil P fertility as the result of past P application can reduce the need of P inputs without decreasing crop P uptake.
2. Residual soil P can contribute to crop production with a considerable lag time. The results provide important information on where and how much P is needed to achieve food security in 2050 while maintaining soil fertility and limiting agricultural expansion and deforestation.
3. Including the residual P in the estimations leads to a reduced P requirement compared with other studies that did not account for residual P.
4. A large P depletion has been shown in grassland soils in all regions in the world, except for Eastern and Western Europe, which are currently in equilibrium but have built up residual P in grassland soils in past decades.
5. Phosphorus transfer from grassland to cropland is the principal reason for the globally unbalanced budgets, through manure transfers from grasslands to croplands. Mineral fertilizers in grasslands are of minor importance except in Europe, and P in ruminant meat and milk represents a small fraction compared with P in the manure.
6. Future demand of meat and milk will increase the pressure on grasslands to provide grass and fodder for the animals. Given the past and current P status of grassland soils, increase of grass production for future ruminant stocks can only be achieved, while avoiding grassland expansion and deforestation, through mineral P fertilization, adequate management of livestock and manure, and soil conservation practices to control runoff P losses.
7. As a result of residual soil P built up in the past four decades in China, crop production targets can be achieved in combination with a reduction of soil P surpluses through reducing P application. Using our quantitative modelling approach we illustrate the key role of China in managing the global P crisis if policy makers and farmers adopt a proper and sustainable P application strategy.
8. Up to 2050, 10,700 Tg of phosphate rock will need to be mined across the globe to produce the P fertilizer for the food production on croplands and

grasslands. This amount of phosphate rock is about 16% of the total phosphate rock reserves (67,000 Tg) estimated in 2014 (USGS, 2014). Thus, the world will not face as rapid a P depletion as earlier thought, though increasing mineral fertilizer prices may pose a serious risk to regions with low P soil stocks.

6.5. Outlook

Most of the recent research and debate focusing on P use and sustainability is highly fragmented. All stakeholders including scientific experts, industry and decision-makers should be involved in a participatory process to find sustainable pathways for P use in the food production-consumption chain (Neset et al., 2013). This research provides new insights into the scientific aspects of P use at global scale that can be integrated into sustainable P use pathways.

To estimate the future need of P for a target food production in this research, some of the available scenarios that include future food demand, production, and the nutrient uptake were used (Chapters 3, 4 and 5). In this study we used two scenarios. A recommendation could be to look at more scenarios that can cover a wider range of uncertainties. More transparent scenarios with a clear focus on future demand and supply of P are required to evaluate options for future food and energy production (Neset and Cordell, 2012).

For a global scale study as presented in this thesis, there is a substantial lack of data. There is no data available on the amount of livestock manure and grass production in the existing databases or in the literature. There are also relatively few data available about livestock feed and those limited data are only available at regional scale. Thus, I highlight the need for further data collection and research on these issues in future.

Furthermore, in the DPPS model, it was assumed a constant transfer time from the labile to the stable pool and two different transfer times from the stable pool to the labile pool, respectively in tropical and non-tropical regions. These time constants of transfers between the labile and stable pools can be estimated from field experiments. Although we showed with a sensitivity analysis in Chapter 3 that the model is relatively insensitive to changes in these parameters (Sattari et al., 2012; Wolf et al., 1987), to have more precise information on P accumulation in a specific agricultural soil and to estimate the magnitude of P saving by efficient P application as discussed in Chapter 5, more field experiments in different soil types are required.

Finally, I conclude that given the increasing amount of P reserves in recent explorations and the important role of residual P in long-term production, projections of peak phosphorus should be significantly shifted. A rough estimate shows that the identified economically exploitable P reserves (67,000 Tg) could last at least for 250 years, given the boundary and future P trajectory conditions of this study considered for 2050.

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Summary

The demand for food is increasing due to the growing world population and changing diets. To meet the food demand it is widely acknowledged that land, water and energy are the key limiting resources of global crop production. However, less attention has been paid to nutrients such as phosphorus (P). Phosphate rock is the primary source of P for mineral fertilizer production and has been used excessively in agricultural systems for decades in many industrialized countries, whereas in other countries it has barely been used. Studies show that depletion of P reserves across the world makes P a limiting nutrient for future global food production. While the time scale of P depletion is debatable, a critical question beyond the physical availability of P is whether P resource depletion can be managed by sustainable P consumption. This question is also relevant because over supply of P in intensive agricultural systems lead to environmental risks.

This research aimed to investigate the relationship between P application and P in harvested products at different scales by analysing historical data on P use and P yields. These relationships were also simulated using a modeling approach. Objectives of this research were:

1. To assess the long-term P application through mineral P fertilizer and manure, historical P withdrawal by harvested crop and grazed grass and P transfer between cropland and grassland using long-term data from different world regions.

2. To calculate future global P fertilizer requirements for food production in different world regions until 2050.

3. To estimate the amount of P that can be saved through a more sustainable use of P accounting for the contribution of residual P in soil.

This research is the first study that quantifies the global-scale P requirement until 2050 for croplands as well as grasslands. Analyses reported in this thesis

were supported by a) long-term data on P application and crop production data and b) a modeling approach that considered major P flows in a soil-crop system.

Analysis and assessment of long-term P application and P uptake, future global P requirement and interactions between nutrients and plants required a quantitative modelling approach. After an extensive survey, we tested the QUEFTS model (Quantitative Evaluation of Fertility of Tropical Soils) for assessing crop yields in response to the three key nutrients, N, P and K, in different environments (Chapter 2). Since the QUEFTS model was initially developed to simulate interactions between N, P and K for tropical soils under a maize crop, the model needed to be adapted to different environments and crops (maize, wheat and rice). Recalibration and modification resulted in a good agreement between model predicted and observed yields for six different data sets from tropical and temperate regions. Although the recalibrations and modifications increased the applicability of the model for application in global studies, QUEFTS is still data demanding. Moreover, further testing (and probably improvement) is probably needed since this model includes semi-empirical relations.

In the next phase of the research a simple model of the crop-phosphorus interaction was used referred to as the “Dynamic Phosphorus Pool Simulator” (DPPS) model. The major feature of the model is inclusion of two pools for soil P, namely labile and stable P and the assumption that except for P no other limiting factor influences crop growth. Using the model, the effect of residual soil P on long-term crop and grass production in croplands and grasslands and their contributions to global food production was studied. Accounting for legacy P in estimating the required fertilizer led to a reduced fertilizer requirement compared with other studies that did not account for residual P. Due to contribution of soil residual P, the future P requirements from fertilizer in croplands was estimated to increase less than the increase in crop production. In grasslands the opposite was found due to the low fertilizer use on grasslands and the export of manure to croplands.

The analysis showed that in Europe, Asia, Latin America and Oceania crop production will benefit from the residual soil P built up over the past decades, even with a reduction in P application rates between 2008 and 2050 (Chapter 3). In contrast, in Africa (although being the richest continent in terms of P reserves), soils are very poor and have been depleted steadily over the years due to the low rate of P inputs. Thus, more than a five-fold increase in P application is needed to achieve the target P uptake in this continent in 2050. To meet the target crop production according to the GO scenario of MEA scenarios in 2050, globally about 1200 Tg P (from mineral fertilizer and animal manure) was estimated to be required between 2008 and 2050.

A large part of the P in animal manure that is applied in cropland actually originates from grassland. The transfer of nutrients from grassland to cropland is particularly important in developing countries, where it helps to satisfy crop requirements and to build-up residual soil P in cropland areas. Phosphorus transfers between grasslands and croplands (in both directions) were quantified and this was used to calculate the P required for grasslands and croplands until 2050 (Chapter 4).

Two categories of grassland, i.e. grasslands in mixed and landless livestock production systems (intensive systems) and in pastoral systems were distinguished. A general conceptual framework was elaborated that focuses on agricultural grassland systems including the key P inflows, outflows and distinguishing four compartments within the grassland system boundary: grassland-based livestock population, grassland-based livestock manure, soil and grass. For calculation of historical soil P budgets for soils under grassland, internal and external manure spreading in grassland and mineral fertilizer P were considered as inflows and grass uptake and erosion were regarded as outflows. We then calculated the transfer of P between croplands and grasslands over the period 1970-2005. Phosphorus is imported through livestock feed from cropland to grassland, while manure is exported from grassland to cropland. Since P transfer from grasslands to croplands is not compensated by P transfer from croplands to grasslands, grassland soils are depleted from P.

Using the DPPS model the amount of P needed for meeting the global grass requirement in intensive and pastoral systems was calculated according to the Rio+20 scenario for the period until 2050. The global P input required in grassland between 2005 and 2050 was estimated at ca. 1215 Tg P. Accounting for the contribution of P from animal manure (655 Tg) in global grasslands, 560 Tg P in the form of mineral P fertilizer would be needed between 2005 and 2050. In 2050, globally 24 Tg yr^{-1} of mineral P fertilizer would be needed to avoid loss of soil fertility and declining grassland productivity. In cropland, excluding the share of manure P (32%) from the total required P, the global mineral P was estimated as 20.8 Tg P in 2050. Thus, the amount of mineral fertilizer P needed in cropland and grassland systems in total was estimated to be 45 Tg in 2050, corresponding to 350 Tg phosphate rock. To achieve global food production in 2050, 10700 Tg of phosphate rock needs to be mined across the globe to produce the P fertilizer for the agricultural and food production sectors. This amount of phosphate rock is only about 16% of the total global phosphate rock reserves (67000 Tg) reported to exist in 2014.

Thus, we will not probably run out of P, but mineral fertilizer prices will increase, which has more severe effects on agricultural production in regions with low soil P stocks than in the industrialized countries of the North with P saturated soils.

In Chapter 5, I emphasized that the transfer to a much more efficient use of P in Western European countries might be regarded as a relevant experience for other countries that face the challenge of agronomic P surpluses. To provide a tangible example, China, as a key player in P consumption and P fertilizer production at global scale, was studied. Among all countries, China has a strategic position in phosphorus production and consumption. China has a long history of agriculture and currently feeds 20% of the world's population (1.3 billion) on only 7% of the arable land (130 million ha). This country is now the largest global consumer (30%) and the largest producer (37.5% of world total in 2010) of P fertilizer and is responsible for 50% of total fertilizer P use in Asia. From the livestock side China also ranks first in terms of monogastric animal stocks (pigs and poultry) in the world. It is relevant to assess future P requirements to support food production as 30-50% more food will be needed to meet China's projected demand. Thus, historical data on P application and uptake in different regions of China was studied from 1970 to 2010. Using the modeling approach the required P application to meet the P uptake in agricultural production by 2050 was estimated. Based on the quantitative analysis, it is concluded that if China applies sustainable P use policies that take residual P into account, almost 20% of the P fertilizer (mineral and manure) would be saved until 2050. This amount is equivalent to half of the P required in Africa or sufficient for the requirements of Western Europe until 2050. This highlights the potential role of China in global P consumption to improve the sustainability of current P footprint at global scale. Furthermore, the effective use of P should follow from cooperation among all sectors along the food chain. Sustainable P application is achievable at global scale if all stakeholders of the agricultural production-consumption chain and policy makers collaborate to develop and use appropriate technologies and enabling legislations.

Finally, I argued that further reforms are needed in governmental policy, ultimately leading to an integrated nutrient management policy based on three pillars: food security and farmers' income, environmental sustainability, and resource use efficiency.

Samenvatting

De vraag naar voedsel stijgt vanwege een groeiende wereldbevolking en veranderende diëten. Het is breed geaccepteerd dat land, water en energie belangrijke beperkende hulpbronnen zijn om aan de toenemende vraag naar voedsel te voldoen. Echter, minder aandacht wordt besteed aan de mogelijke schaarste aan nutriënten, zoals fosfor (P). Fosfaatrots, de primaire bron van P voor het produceren van fosfaatkunstmest, is gedurende de laatste decennia overvloedig gebruikt in landbouwsystemen in geïndustrialiseerde landen, terwijl het in de meeste ontwikkelingslanden amper is toegepast. Studies laten zien dat de uitputting van P reserves in de wereld, P een limiterend nutriënt maakt voor de toekomstige voedselvoorziening. Een belangrijke vraag, naast die omtrent de tijdschaal van fysieke beschikbaarheid en uitputting van P, is of we P gebruik en uitputting kunnen beperken door een meer duurzame consumptie. Deze vraag is ook relevant omdat overbemesting van P in intensieve landbouwsystemen tot milieurisico's leidt.

Dit onderzoek had als doel de relatie te onderzoeken tussen P toediening en P in geoogste producten op verschillende schalen, door het analyseren van historische data omtrent P gebruik en P opbrengsten in de landbouw. Deze relaties werden ook gesimuleerd met behulp van computermodellen. De drie belangrijkste doelen van het onderzoek waren:

1. De schatting van lange-termijn P toediening door P kunstmest en dierlijke mest, historische P onttrekking door het geoogste gewas en gegraasd gras, en P transfer tussen akkerbouw en grasland, door gebruik te maken van lange-termijn data van verschillende wereldregio's.

2. Berekening van de toekomstige mondiale behoefte (tot 2050) aan P kunstmest voor voedselproductie in verschillende regio's van de wereld.

3. Schatting van de hoeveelheid P die kan worden bespaard door een meer duurzaam gebruik van P, door rekening te houden met de bijdrage van residuele P in de bodem.

Dit onderzoek is het eerste wat een schatting maakt van de mondiale behoefte aan P in kunstmest en dierlijke mest tot 2050 voor akkerbouw en graslanden. Analyses die in dit proefschrift worden gerapporteerd werden onderbouwd door a) lange-termijn data omtrent P toediening en gewasproductie en b) een modelleermethode die de belangrijkste P-stromen in een bodem-gewassysteem beschouwt.

Voor de analyse en schatting van de lange-termijn P toediening en opname, toekomstige mondiale P behoefte en interacties tussen nutriënten en gewassen is een kwantitatieve modelleringsbenadering toegepast. Na een uitgebreide verkenning, hebben we QUEFTS (Quantitative Evaluation of Fertility of Tropical Soils) getest voor het schatten van gewasopbrengsten in afhankelijkheid van drie belangrijke nutriënten (N, P en K) in verschillende omgevingen (Hoofdstuk 2). Omdat het QUEFTS model oorspronkelijk ontwikkeld was om interacties tussen N, P en K in tropische bodems onder mais te simuleren, moest het model aangepast worden voor andere omstandigheden en gewassen (mais, tarwe en rijst). Kalibratie en aanpassing resulteerde in een goede overeenkomst tussen modelvoorspelling en waargenomen opbrengsten voor zes verschillende datasets uit tropische en gematigde regio's. Hoewel door de kalibraties en aanpassingen de toepasbaarheid van het model werd vergroot, is de toepasbaarheid voor mondiale studies nog steeds beperkt door de grote databehoefte van QUEFTS. Bovendien, is het waarschijnlijk nodig om QUEFTS verder te testen (en vermoedelijk te verbeteren), omdat het model semi-empirische relaties bevat.

In de volgende fase van het onderzoek werd een simpeler model van gewas-fosfor interacties gebruikt, namelijk de Dynamische Fosforpool Simulator (DPPS). Het belangrijkste kenmerk van het model is dat het twee pools voor bodem P bevat, namelijk labiele en stabiele P en de veronderstelling dat behalve P er geen andere factoren de gewasgroei limiteren. Met dit model is het effect bestudeerd van residuele bodem P op de lange-termijn gewas- en grasproductie op bouwland en grasland, alsmede de bijdrage van bodem P aan de mondiale voedselproductie. Door rekening te houden met de residuele P in de bodem bij het schatten van de benodigde hoeveelheid P uit kunstmest en dierlijke mest, komt de schatting van de behoefte aan P uit kunstmest lager uit dan andere studies die geen rekening hielden met de residuele P. Vanwege de bijdrage van de residuele bodem P, neemt de geschatte behoefte aan P uit kunst- of dierlijke mest tot 2050 minder toe dan de gewasproductie moet stijgen in dezelfde periode. Voor graslanden werd het omgekeerde gevonden, vanwege het lage kunstmestgebruik op graslanden en de netto export van mest naar bouwland.

De analyse liet zien dat in Europa, Azië, Latijns Amerika en Oceanië de gewasproductie zal profiteren van de residuele bodem P die is opgebouwd over de laatste tientallen jaren, zelfs met een reductie van de P toediening tussen 2008 en 2050 (Hoofdstuk 3). Hoewel Afrika de grootste mondiale voorraad rotsfosfaat heeft, zijn Afrikaanse bodems erg arm en uitgeput in de loop der tijd, vanwege de lage P inputs. Voor Afrika is daarom een meer dan vijfvoudige toename in P toediening noodzakelijk om de nodige P gewasopname in 2050 te realiseren. Om de beoogde gewasproductie in 2050 te realiseren volgens het zogenaamde Global Orchestration scenario van de Millennium Ecosystem Assessment scenario's, is mondiaal naar schatting ongeveer 1200 Tg P (uit kunstmest en dierlijke mest) nodig tussen 2008 en 2050.

Een groot deel van de P in dierlijke mest die is toegediend op bouwland is eigenlijk afkomstig van grasland. De transfer van nutriënten van grasland naar bouwland is vooral belangrijk in ontwikkelingslanden, waar het helpt bij het voldoen aan de gewasbehoefte en de opbouw aan residuele P in de bodems van bouwlanden. Fosfor transfer tussen grasland en bouwland (in beide richtingen) werd gekwantificeerd en dit werd gebruikt om de nodige hoeveelheid P voor grasland en bouwland tot 2050 te berekenen (Hoofdstuk 4).

Twee categorieën graslanden werden onderscheiden, namelijk graslanden in gemengde en niet-grondgebonden dierlijke productiesystemen (intensieve systemen) en in meer-extensieve, pastorale systemen. Een algemeen conceptueel kader werd uitgewerkt wat zich richt op landbouwkundige graslandsystemen, met de belangrijkste P instromen en uitstromen tussen vier compartimenten binnen het graslandsysteem: grasland-gebaseerde dierlijke productie; grasland gebaseerde dierlijke mest; bodem en gras. Voor de berekening van historische budgeten van P in graslandbodems, werden verspreiding van dierlijke mest op grasland en minerale P bemesting beschouwd als instromen en grasopname en erosie als uitstromen. Daarna berekenden we de transfer van P tussen bouwland en grasland over de periode 1970-2005. Fosfor wordt geïmporteerd naar grasland via veevoer van bouwland, terwijl dierlijke mest wordt geëxporteerd van grasland naar bouwland. Omdat P transfer van grasland naar bouwland niet gecompenseerd wordt door de P transfer van bouwland naar grasland, worden graslanden netto uitgeput in termen van bodem P.

Gebruikmakend van het DPPS model is de hoeveelheid P geschat die nodig is om aan de mondiale grasbehoefte voor intensieve en pastorale systemen te voldoen, volgens het Rio+20 baseline scenario en voor de periode tot 2050. De mondiale P input die nodig is tussen 2005 en 2050 in grasland werd geschat op ca. 1215 Tg P. Rekening houdend met de bijdrage van P uit dierlijke mest op mondiale graslanden (655 Tg), is 560 Tg P in de vorm van kunstmest P nodig tussen 2005 en 2050. In 2050 is mondiaal 24 Tg per jaar aan minerale kunstmest P nodig om verlies aan bodemvruchtbaarheid en afname van de graslandproductie te voorkomen. Op bouwland werd (rekening houdend met het aandeel van 32% van dierlijke mest P in de totale behoefte) de totale kunstmest P behoefte geschat op 20.8 Tg P in 2050. Dus, de hoeveelheid kunstmest P nodig voor bouwland- en graslandsystemen in totaal werd geschat op 45 Tg in 2050, hetgeen overeenkomt met 350 Tg fosfaatrots. Om de noodzakelijke mondiale voedselproductie in 2050 te realiseren, is tot dat jaar 10700 Tg fosfaatrots nodig, die wereldwijd moet worden gewonnen om de P kunstmest te kunnen produceren voor de gehele landbouwsector. Deze hoeveelheid fosfaatrots is slechts ca. 16% van de totale reserves aan fosfaatrots (67000 Tg), zoals die gerapporteerd werden in 2014. Dus lijkt het alsof we voorlopig nog geen tekort aan P zullen krijgen, maar indien door verschillende oorzaken de prijzen van minerale fosfaatkunstmest toenemen, zullen de gevolgen voor de landbouwproductie groter zijn in regio's met lage bodem P voorraden (veelal ontwikkelingslanden), dan in geïndustrialiseerde landen met P verzadigde gronden op het noordelijk halfrond.

In hoofdstuk 5 wordt benadrukt dat de overgang naar een meer efficiënt gebruik van P in West-Europese landen gezien kan worden gezien als een relevante ervaring voor andere landen met landbouwkundige P overschotten. China wordt als een concreet voorbeeld nader bestudeerd omdat dit land een kernspeler is in de mondiale P kunstmestproductie en -consumptie. China heeft een lange historie van landbouw en voedt momenteel 20% van de wereldbevolking (1,3 miljard inwoners) op slechts 7% van het bouwlandareaal (130 miljoen ha). Dit land is nu de grootste mondiale consument (30%) en de grootste producent (37,5% van het wereldtotaal in 2010) van P kunstmest en is verantwoordelijk voor 50% van het totale kunstmest P gebruik in Azië. China heeft de grootste aantallen varkens en kippen in de wereld. Het is belangrijk om de toekomstige P behoefte te schatten voor het voldoen aan de voedselbehoefte, omdat in 2050 30-50% meer voedsel nodig zal zijn om aan de Chinese behoefte te voldoen. Daarom zijn historische data omtrent P gebruik en -opname in verschillende regio's van China (1970 tot 2010) bestudeerd. Gebruikmakend van het DPPS model werd de benodigde hoeveelheid P om aan de P opname in landbouwproducten tot 2050 te voldoen, geschat. Gebaseerd op de kwantitatieve analyse werd geconcludeerd dat als China beleid toepast dat gericht is op duurzaam gebruik van P en residuele bodem P, tot 2050 bijna 20% op de hoeveelheid P uit dierlijke mest en kunstmest kan worden bezuinigd. Deze hoeveelheid is equivalent aan de helft van de P behoefte van Afrika tot 2050 of gelijk aan de P behoefte van West-Europa voor dezelfde periode. Dit illustreert duidelijk de potentiele rol van China in de mondiale P consumptie en bij het vergroten van de duurzaamheid van de P footprint op mondiale schaal. Het effectieve gebruik van P vraagt om samenwerking tussen alle sectoren in de voedselketen. Duurzame P toediening is haalbaar op mondiale schaal als alle belanghebbenden van de landbouwkundige productie-consumptieketen en de beleidsmakers samenwerken om geschikte technieken en beleidsmaatregelen te ontwikkelen en toe te passen.

Tenslotte wordt betoogd dat verdere hervormingen van het overheidsbeleid nodig zijn, om te komen tot een geïntegreerd beleid voor nutriëntenmanagement, gebaseerd op drie pilaren: voedselzekerheid en boereninkomen, ecologische duurzaamheid en efficiënt gebruik van hulpbronnen.

Acknowledgements

In March 2009, just after having an interview in Plant Production Systems Group for the PhD project entitled" Imagine a world with no phosphorus" – what an exciting title! - I left the Netherland to Iran for celebrating the Persian New Year with my family. On the first day of the New Year, I got a phone call from the PPS that informed me I was accepted as a PhD candidate. It was a great and unforgettable New Year's present I received.

This thesis is the fruit of uncountable interactions and exchanges with number of people who stimulated and inspired my work. Their supports facilitated fulfillment of my research in the past five years that I wish to acknowledge them.

First and Foremost I wish to express my special appreciation and thanks to my supervisors for the continuous support of my PhD research, for their patience, motivation, enthusiasm, and great knowledge. Martin, Lex, Ken and Bert from the start and then during the last five years your help, positive comments, and feedbacks gave me the courage to continue my job.

Martin I would like to thank you for your precious support and guiding me to grow as a research scientist. Martin, I never forget that evening when you surprised me by your unplanned visit to inform me about the Third Publication Award I received from Graduate School for Production Ecology and Resource Conservation (PE&RC). Also many thanks for preparing the Dutch summary of this thesis, which was difficult for me to write it myself.

Lex, your advices on both my research and my career have been priceless. Special thanks for your availability in both PBL and PPS. It was always a pleasure to come and work with you in PBL, and it helped me to make a significant progress. Lex, we had many lively discussions during these years and

I have learned a lot from you. Thank you for your invaluable support and encouraging my research!

I am also grateful to Ken, your encouragement and guidance helped me in all the time of research and writing of this thesis. Thanks for believing and pushing me to stand as a scientist. Your contribution to this study is priceless.

I would like to show my gratitude to Bert Smit. Your support, insightful comments, and some difficult questions led to a higher quality research.

My sincere thanks also goes to Bert Janssen, for very fruitful discussions, which led to the production of Chapter 2 of this thesis. Thank you for your patience with answering my questions.

I am grateful to Joost Wolf who in early 90s built a modeling approach for studying the dynamics of phosphorus in soil. This approach was developed and used in Chapters 3, 4 and 5 of this research as the DPPS model. Joost, thank you for all the stimulating discussions and many helpful suggestions for improvement of the first draft of Chapter 3.

Hanna and Ruben, supervising your BSc and MSc thesis was a great experience for me. I really enjoyed working with you. Your valuable contributions are reflected in Chapters 4 and 5 of this thesis. Thank you both!

Thanks to Achim Dobermann, Stephan Haefele, Jianbo Shen, Pablo Tittonell and Rothamsted Research (Paul Poulton, Andy Macdonald and Margaret Glendining) for kindly providing data used in Chapter 2. Jantine van Middelkoop, Jan Vos and Jaap Schroder, I would like to thank you for helping me to find more data, when I was challenged to find more datasets for testing the QUEFTS model. Thanks to Fusuo Zhang, Haigang and Guohua Li for kindly sharing the data.

Plant Production Systems Group provided a pleasant and interdisciplinary environment, which led to inspiring discussions with my colleagues during weekly PPS-Lunch meeting, the PV events, conferences, coffee breaks and lunches. Ria, Charlotte and Alex-Jan I should thank you for your support in administrative tasks, and nice chats we had during the coffee breaks. My special thanks to my former officemates Myriam, Sander and later Argyris, Lenny, Maryia and Esther for all the fun we have had in the last years.

I would also like to thank all of my friends who visited me after giving birth to our twins and supported me to strive towards my goal. I would especially like to thank Madeliene, Marieke, Renske, Esther, Dora, Myriam and Rik, Maryia and Bram, Diony and Nikos, Akvile and Pasha, and Mayumi for all the good moments shared.

Thanks to my other friends and colleagues Peter, Pytrik, Sjanie, Mink, Katrien, Bert Rijk, Marcel, Maja, Gerrie, Murilo, Naomie, Andre, Merel, Aisha, katrin, Stefano, Chrispen, Sohail Khan, Masood, Guillaume, Benjamine, Jairos, Lotte, Juliana, Alba, and Charles.

Special thanks to my lovely family. Words cannot express how grateful I am to my parents who support me spiritually throughout my life and give me happiness and love, even remotely.

I would like to express my appreciation to my beloved husband who spent sleepless nights with me supporting me in any moment when there was no one to answer my queries. Vahid, you supported me during all stages of my research, and at each and every moment. Without your encouragement and understanding, fulfillment of this work would have been too difficult. Thank you for your unconditional love!

Finally, I am blessed with my lovely twins, Artin and Radvin, who went with me through the every stage of my research in the last two years. You gave me the confidence and trust that “nothing is impossible”.

Curriculum Vitae

I was born in Karaj, a big city at the beautiful foothills of the Alborz Mountains, Iran. After finishing high school, I started my BSc to study environmental engineering with specialization in Natural Resources at the Azad University of Tehran. Following my BSc, I completed my first MSc on “Environmental Management” in 2001 at the Science and Research University of Tehran. Besides doing my MSc, in 1998 I started my part time career as a teacher in high school with teaching biology and geology in different grades. When I completed my MSc study, I started my professional career as an environmental expert in water demand and meteorology studies and project manager in Environmental Impact Assessment (EIA) projects in sewerage systems, irrigation and storage dam projects.

In 2006 I received an Excellence Scholarship from Utrecht University to start my second MSc on “Sustainable Development (Land use, Environment and Biodiversity)”. Graduated in July 2008, I fulfilled a research project on “Cellular automata modeling of surface patterns in peatlands” in 2007 as well as an MSc thesis ”Effects of global warming on vegetation dynamics and carbon balance in peatlands” in 2008. Subsequently, I worked for the Department of Environmental Sciences at Utrecht University as a researcher until April 2009. I started my PhD in Plant Production Systems group (PPS), Wageningen UR in May 2009 to study the effect of phosphorus scarcity on global food security. In my PhD I aimed to assess global P fertilizer requirements for future food production, with a regional focus. During my PhD, I collaborated with the Netherlands Environmental Assessment Agency (PBL), Utrecht University, Department of Earth Sciences–Geochemistry and the Plant Research Institute (PRI) and China Agricultural University (Centre for Resource, Environment and Food Security). Results of this research have been published in several peer-reviewed journal papers and one publication (Chapter 3 of this thesis) received the PE&RC publication award (2012), and the Global Center for Food Systems Innovation grant (2013) and significant coverage in the media.

List of Publications

Sattari, S.Z., Martinez Rodríguez, R., Van Ittersum, M.K., Beusen, A.H.W., and Bouwman, A.F. (2014). Massive transfer of phosphorus to croplands threatens sustainability of grasslands. To be submitted.

Li, G., van Ittersum, M. K., Leffelaar, P., Sattari, S.Z., Li, H., Huang, G., and Zhang, F. (2014). Quantifying phosphorus flows at different levels in China to identify potential policy measures to improve phosphorus management in agriculture. Agricultural Systems (under reveiw).

Sattari S.Z., van Ittersum M.K., Giller K.E., Zhang F., Bouwman A.F. (2014) *Key role of China and its agriculture in global sustainable phosphorus management,* Environmental Research Letters *9, 054003.*

Sattari S.Z., van Ittersum M.K., Bouwman A.F., Smit A.L. and, Janssen B.H. (2014) *Crop yield response to soil fertility and N, P, K inputs in different environments: testing and improving the QUEFTS model.* Field Crop Research Journal 157:35-46.

Sattari S.Z., van Ittersum M.K., Bouwman A.F., Li G., Giller K.E. (2013) Efficient use of phosphorus*: the role of China in saving a vital resource for global food security.* Proceeding of the First International Conference on Global Food Security. Noordwijkerhout, The Netherlands 29th Sep-2nd October.

Sattari S.Z., Bouwman A.F., Giller K.E., Van Ittersum M.K. (2012) *Residual soil phosphorus as the missing piece in the global phosphorus crisis puzzle*. Proceedings of the National Academy of Sciences of the United States of America 109:6348-6353.

Sattari S.Z., Bouwman A.F., Giller K.E., & van Ittersum M.K. (2012) *Residual soil phosphorus substantially decreases global P fertilizer requirements*. 12th congress of the European Society for Agronomy Helsinki, Finland, 20-24 August 2012.

Sattari S.Z., van Ittersum M.K., Bouwman A.F., Smit A.L. and, and Giller, K.E. (2010). *Generalization of QUEFTS for global-scale assessment of crop-phosphorus response.* Proceedings of the IVth International Symposium on Phosphorus Dynamics in the Soil-Plant Continuum (ISPDSPC) Beijing, China, 19-23 September 2010 (Reference No.: S4-28) Beijing, China.

PE&RC Training and Education Statement

With the training and education activities listed below the PhD candidate has complied with the requirements set by the C.T. de Wit Graduate School for Production Ecology and Resource Conservation (PE&RC) which comprises of a minimum total of 32 ECTS (= 22 weeks of activities)

Review of literature (5 ECTS)

- Global Phosphorus budget in agriculture and future food security

Writing of project proposal (1.5 ECTS)

- Imagine a world with no phosphorus

Post-graduate courses (4.2 ECTS)

- iGIS; PE&RC (2009)
- Bayesian linear model; PE&RC (2010)
- Introduction to R for statistical analysis; PE&RC (2011)

Invited review of (unpublished) journal manuscript (2 ECTS)

- Nature (2012)
- AMBIO, A journal of the Human Environment (2012)
- Biogeosciences discussions (2013)
- Environmental Science & Technology (2013)
- AMBIO, A journal of the Human Environment (2013)

Deficiency, refresh, brush-up courses (3 ECTS)

- Analysis and design of organic farming systems (2009)
- Nutrient management (2010)
- Basic statistics; PE&RC (2011)

Competence strengthening / skills courses (3 ECTS)

- English for academic purposes; James Boswell Institute (2008)
- Information literacy and endnote introduction; Wageningen UR Library (2009)
- How to write a world-class paper; Wageningen UR (2010)
- Project and time management; Valley Consult (2011)

PE&RC Annual meetings, seminars and the PE&RC weekend (1.5 ECTS)

- One-day PE&RC symposium-selling science (2010)
- PE&RC Weekend (2010)
- PE&RC Symposium: roots: a comprehensive approach to the hidden half (2011)

Discussion groups / local seminars / other scientific meetings (7.5 ECTS)

- PPS Weekly lunch seminars (2009-2013)
- Maths and stats; PE&RC discussion group (2009-2010)
- Soil-plant interaction; PE&RC discussion group (2010-2011)

- Global Soil Fertility; Wageningen UR (2011)
- WaCASA Monthly lunch meetings (2012-2013)
- Plant Sciences Seminars; Global nutrient cycles and food security (2013)
- Sustainable intensification of agricultural systems (2013)
- Contested agronomy: the politics of agricultural research (2013)

International symposia, workshops and conferences (9 ECTS)

- Mini-seminar: Phosphorus shortage; the Hague, the Netherlands (2009)
- Mini symposium, Phosphorus availability; Wageningen UR (2009)
- IMAGE workshop; Wageningen UR (2009)
- IVth International symposium on phosphorus dynamics in the soil-plant continuum (ISPDSPC); poster presentation; Beijing, China (2010)
- Mini symposium: the role of organic resources in soil fertility management for sustainable agricultural intensification, rural poverty alleviation and risk mitigation across social-ecological contexts in Sub Saharan Africa; Wageningen UR (2010)
- IMAGE workshop; oral presentation; PBL-NL (2010 and 2011)
- Mini-seminar phosphorus shortage-nutrient platform; IHE Delft, the Netherlands (2011)
- Nutrient cycling and management workshop; Wageningen (2011)
- 12th Congress of the European Society for Agronomy; oral presentation; Helsinki, Finland (2012)
- Sustainable phosphorus use in the EU workshop; Hof van Wageningen, the Netherlands (2013)
- European Sustainable Phosphorus Conference; Brussels (2013)
- Dies Natalies: phosphorus is essential for global food security; oral presentation (2013)
- Food security and the productivity, health and nutrition workshop; Wageningen UR (2013)
- First International Conference on Global Food security; oral presentation; Noordwijkerhout, the Netherlands (2013)

Lecturing / supervision of practical's / tutorials (3 ECTS)

- Models for ecological systems (2012)

Supervision of MSc student (3 ECTS)

- First exploration of global phosphorus budgets in grassland

The research presented in this thesis was funded by the Wageningen University and Research Centre Program Kennisbasis4 and the C.T. de Wit graduate school for Production Ecology and Resource Conservation (PE&RC).

Cover Illustration	David Schotting
Cover design	Proefschriftmaken.nl || Uitgeverij BOXPress
Printed by	Proefschriftmaken.nl || Uitgeverij BOXPress