TOWARDS EFFICIENT USE OF RESOURCES IN FOOD SYSTEMS

Exploring circular principles and strategies

Heleen van Kernebeek

Propositions

1. The environmental impact of food must be expressed per capita per year and for the total population.
(this thesis)

2. Resource-efficient food systems feed animals as if they were waste bins.
(this thesis)

3. The worse the state of our world, the more precise we assess improvement options.

4. Missions to Mars make us more aware of our planetary boundaries.

5. Studies yielding short-term solutions should be published on rapidly degradable paper.

6. Rather than a minimum number of women, Executive Boards should have a minimum number of inquisitive people.

Propositions belonging to the thesis, entitled

Towards efficient use of resources in food systems. Exploring circular principles and strategies

Heleen van Kernebeek

Wageningen, 12 May 2020

Towards efficient use of resources in food systems

Exploring circular principles and strategies

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Towards efficient use of resources in food systems

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Thesis

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“Ze zeggen: waar de mens is geboren, daar is de aarde zijn wieg. En de dood legt je er als het ware alleen in terug. En ze wiegt je, ze wiegt je, tot je weer ongeboren bent en niet verwekt”

Steen op steen - Wiesław Myśliwski

Abstract

Natural resources for food production, such as land, phosphate rock and fossil energy, are scarce. Despite their scarcity, these resources are currently inefficiently used in the food system. The objective of this thesis was to understand the combined effects of technical and consumption strategies, to reduce the use of natural resources in a food system. To this end, we first reviewed 12 life cycle assessments studies that explored various diets. We concluded that the 'daily' or 'yearly' diets instead of meals should be used to compare environmental impacts of diets. We also found that accounting for nutritional quality of the diets hardly affected our comparison of environmental impacts of diets varying in % of animal-source food. We then explored the minimum requirements of land, phosphorus (P) and energy independently to feed a human population with diets varying from 0% protein from animals (PA) (i.e. a vegan diet) to diets containing 80% PA, using an integrated food systems approach. Our material and nutrient flow model was a conceptual representation of a food system that was parameterised with crop and animal production data from the Netherlands. We assumed that there was no import and export of food and feed. While the Dutch food system is not representative of all food systems in the world, sensitivity analyses demonstrated that the principles deduced from our model results also hold for other food systems. Results indicated that land is used most efficiently if people consume ca. 12% of PA, especially from milk. The role of animals in such a land-based diet is to convert co-products from crop production and the human food industry, otherwise not used within the food system, into milk and meat. The optimal %PA in the human diet depends on population size and the relative share of land unsuitable for crop production. Recycling of human excreta showed most potential in reducing P waste, followed by prevention and finally recycling of agricultural waste. Fully recycling P reduces mineral P input by 90%. The optimal amount of animal protein in the diet depended on whether or not P waste from animal products was fully prevented or recycled: if it was fully prevented or recycled, then a small amount of animal protein in the human diet resulted in the most sustainable use of P, but if it was not fully prevented or recycled, then the most sustainable use of P results from a complete absence of animal protein in the human diet. In situations with anaerobic digestion and/or waste prevention, energy input continuously increased with increasing %PA, whereas if none of these strategies were applied, energy input was minimised at about 15% PA. Account must be taken of combined effects of technical and consumption strategies to reduce the use of natural resources in food systems. It requires efforts from all actors to develop a food system that is able to supply the global population with safe and healthy food within environmental limits.

Contents

Chapter 1

General introduction

1. Background

The growing and increasingly prosperous human population is facing the challenge of producing sufficient food for all without running out of natural resources or destroying the Earth's ecosystem (Foley et al., 2011). Natural resources, such as land, phosphate rock and fossil energy, are essential for the production of our food, but their availability is limited.

The availability of natural resources for food production is under pressure, and has led to concerns about how our future demand for food will be met (IPCC, 2019; Reitzel et al., 2019). Land for food production, for example, is under pressure due to competing claims for food, feed and fuel production, urbanisation, and biodiversity and climate targets (Bren d'Amour et al., 2017; Muscat et al., 2019). Although the net area of global agricultural land has remained relatively constant since the mid-20th century, it has incurred a substantial reduction in temperate regions (e.g. Europe, Russia and North America), and a substantial expansion in biodiversity–rich tropical areas. The latter has led to increased carbon emissions and biodiversity loss (Dudley and Alexander, 2017; Willett et al., 2019). The challenge for future agricultural land use, therefore, is to avoid further changes (e.g. deforestation), and to safeguard soil fertility (Willet et al., 2019). Moreover, global reserves of phosphate rock are depleting (Cordell and White, 2014). Rock phosphate is the fossil resource for mineral phosphorus (P), which is an essential macronutrient for crop growth. Globally, about 90% of the extracted rock phosphate is used for food production (Cordell and White, 2014). Due to huge losses and inefficient use of P in the food system, however, only a fifth of that is actually consumed by humans (Cordell, 2015). Estimates of global phosphate rock reserves and their longevity are highly uncertain, and range between a hundred and a thousand years (Edixhoven, 2014; Scholz and Wellmer, 2016). However, as higher quality and more easily accessible phosphate rock is mined first (Smil, 2000), remaining reserves of phosphate rock will contain more impurities, and will be more difficult to mine (Neset et al., 2016). It is therefore widely acknowledged that P should be used more efficiently. Furthermore, fossil energy is the main source of energy input to our food system (Sims and Dubois, 2011; Monforti-Ferrario et al., 2015). It is used to fuel processes such as on-farm field activities, processing and transport, and to produce inputs such as mineral fertiliser. Fossil energy reserves are, however, expected to deplete within the next 50 to 100 year (Shafiee and Topal, 2009). Until a full transition towards renewable energy is reached, the use of fossil fuel in food production is a fact, and fossil energy should be used more efficiently. Our ability to secure food for the future global population thus depends on how much, and how efficiently we use these scarce resources.

In our current, linear food systems, however, land, phosphate rock and fossil energy are wasted and inefficiently used (Cordell et al., 2009; Pelletier et al., 2011; Willett et al., 2019). This is partly due to high average per capita food consumption, which, especially in wealthier countries, exceeds actual requirements (FAOSTAT, 2013; Herrero et al., 2015), high consumption of animal-source food (ASF), again especially in wealthier countries (FAOSTAT, 2013; Herrero et al., 2015), and limited rates of waste prevention and waste recycling in the food system (Cordell et al., 2009; Cuéllar and Webber, 2010; Soethoudt and Timmermans, 2013). Several strategies have been proposed to increase resource use efficiency in food production, among which precision farming, biotechnology, closing yield gaps, a transition to a circular food system, and a reduction in, or even exclusion of the consumption of ASF (Vemireddy, 2014; Jurgilevich et al., 2016; Aune et al., 2017; Poore and Nemecek, 2018; van der Linden et al., 2018). This thesis addresses the potential of the last two strategies to increase the resource use efficiency in the food system: a transition to a circular food system, i.e. a food system which minimises the input of natural resources and the occurrence of waste, and a shift in human diets towards consumption of less ASF. The impact of these strategies on resource use efficiency are assessed for three scarce but essential resources for food production: land, phosphorus (as an illustration for minerals), and fossil energy.

2. Strategies to increase the resource use efficiency in the food system

The two strategies included in this dissertation relate to both the *production* of food (technical strategies to increase circularity in the food system), and the *consumption* of food (consumption strategy to reduce consumption of animal protein).

Technical strategies to increase circularity in the food system include, among others, prevention and recycling of food and feed waste, recycling of co-products not edible to humans (hereafter inedible) such as wheat straw, recycling of human excreta, and recovery of bio-energy from waste.

It is estimated that globally roughly a third of the food produced for human consumption is wasted (Gustavsson et al., 2011). These wastes occur when food products do not meet safety or quality standards or simply get lost during production, post-harvest processing and consumer stages of the food system. As considerable amounts of food products are wasted, prevention of these wastes is a first priority to reduce the use of scarce resources (Papargyropoulou et al., 2014). If waste is not prevented, however, then according to the waste hierarchy it should be recycled as animal feed or fertiliser (Papargyropoulou et al., 2014). Recycled food waste can substitute, for example, feed that is actively produced for animals, and mineral fertiliser, and therefore can contribute to lower requirements for resources, such as land and phosphate rock to cultivate feed crops. If recycling food waste as feed or fertiliser is not feasible, then it can be used to produce bio-energy via, for example, anaerobic digestion (Papargyropoulou et al., 2014). Energy produced from waste can substitute fossil energy, and can contribute, therefore, to lower requirements of fossil fuel. In analogy with food waste, prevention, recycling and recovery of a) feed waste, arising during feeding of animals as a result of spillage and degradation of feed (Remmelink et al., 2012), b) inedible co-products, here defined as co-products that humans cannot or do not want to consume, such as wheat straw, and animal bones and skins, and c) human excreta, also contributes to lower use of scarce resources (Guzha et al., 2005; Heinonen-Tanski and van Wijk-Sijbesma, 2005).

Along with these technical strategies, the resource use efficiency in the food system can also be improved by reducing the human consumption of ASF. Consumption of ASF is generally considered resource inefficient compared to consumption of crop products because animals convert only part of the consumed feed into ASF, i.e. meat and milk. The remainder is converted into inedible products such as, for example, bones and manure, and into heat (i.e. energy loss). This inefficiency implies that a diet without ASF, i.e. a vegan diet, is more sustainable than a diet with ASF (Hallström et al., 2015; Poore and Nemecek, 2018; Shepon et al., 2018). Following the same logic, and if ASF is to be included in the human diet, pork and chicken meat would be more sustainable than beef, given the higher feed conversion efficiency of pigs and broilers compared to beef cattle (Stehfest et al., 2009; Wirsenius et al., 2010; Shepon et al., 2018). However, ruminants have a better ability to convert grass into ASF than monogastric animals. Thus, if grasslands unsuitable for crop production, i.e. marginal grasslands, are available, then ruminants are better able to convert the available feed into food than pigs and poultry. Moreover, if inedible co-products from crops, and/or food waste, are available, animals can convert these resources into ASF without competing with humans. Animals fed on marginal grasslands, inedible co-products and food waste, therefore contribute to resource use efficiency and food security (Fairlie, 2010).

3. Knowledge gaps

So far, a generally accepted method to assess environmental impacts of production and consumption strategies is the life cycle assessment (LCA) (Poore and Nemecek, 2018). LCA assesses the environmental impact or resource use of a product during the entire production chain (Guinée et al., 2002). Besides this main product, a production process, however, might also yield another valuable output. Cultivation of wheat, for example, yields wheat grain and straw. The total environmental impact or resource use has to be allocated to the main product (e.g. wheat grain) and co-products (e.g. wheat straw), and is done so usually on the basis of the relative economic value of the multiple outputs. This results in a per-kg impact for each of the main and co-products. The per-kg impact of main products is used as a building block for either assessing the impact of a production strategy (i.e. increasing crop yield) or the impact of a consumption strategy (i.e. a vegetarian or vegan diet) (Mackenzie et al., 2016; Poore and Nemecek, 2018).

LCA has often been used to compare the environmental impact of human diets that varied in their percentage of protein from animals (Carlsson-Kanyama and Gonzalez, 2009; Davis et al., 2010; Hallström et al., 2015). These studies generally compared meals or daily diets that were similar in terms of energy, protein and fat content, but differed in the amount of plant and animal-source food, and hence in the amount of micro-nutrients, such as vitamins and minerals. It is unclear, furthermore, which unit, i.e. meals or daily diets, we need to compare the impacts of human diets, due to their differences in nutritional value.

Moreover, LCA does not account for differences in the suitability of land to cultivate food crops. When accounting for these differences, as Van Zanten et al. (2016) do in their land use ratio, using marginal land for grazing appears more land use efficient than using this land for crop production, and production of ruminants on marginal land appear more land use efficient than production of monogastric animals fed from cropland. The land use ratio, however, does not account for the total availability of marginal land, and cannot be used, therefore, to assess the potential contribution of marginal land to food security.

Furthermore, LCA studies demonstrated that inclusion of inedible co-products and food waste in animal feed, as replacement of human edible crop products, can reduce the impact of animal production (Mackenzie et al., 2016; Röös et al., 2016). However, an attributional LCA does not account for competition for these resources between various animal production systems, and between animal production and, for example, energy production (van Zanten et al., 2014). The same limitation holds for static system analyses that quantify food waste and nutrient losses at the global or national level (Cordell et al., 2009; Gustavsson et al., 2011; Suh and Yee, 2011; Soethoudt and Timmermans, 2013; Jedelhauser and Binder, 2015). From these static overviews of where in the system losses occur, actions to prevent or reduce these losses can be formulated. However, single interventions to prevent or reduce losses do not consider interaction effects between actors in the food system. An example of an interaction effect resulting from the implementation of a single strategy to improve resource use efficiency in the food system, is the reduced availability of recycled food waste as input to, for example, animal production, as a result of the prevention of food waste. Prevention of food waste is a strategy embedded in the Sustainable Development Goal of the UN to ensure sustainable consumption and production of food. If food waste is prevented throughout the food system, less food needs to be produced to feed a human population. This contributes to lower requirements of scarce resources in the food system. At the same time, however, the food that was previously wasted can no longer be recycled for other purposes, such as, for example, to feed animals. As a consequence, either fewer animals can be produced, and, hence, less ASF can be consumed by humans, or more crops will have to be cultivated for the specific purpose to feed animals. Moreover, if waste by individual actors or processes in the food system is *not* prevented, then recycling this waste within the food system may still reduce the use of scarce resources. When assessing the potential to use recycled waste as feed, fertiliser or bio energy-source, one has to consider the competition for these resources between these various purposes. This competition depends, among others, on whether or not recycling is combined with other strategies, such as, for example, the strategy to reduce the human consumption of ASF. Understanding these combined effects within and between strategies is important in defining effective (combinations of) strategies to reduce the use of natural resources in future food systems.

4. Aim of the thesis

The objective of this thesis is to understand the combined effects of technical and consumption strategies to reduce the use of natural resources in food systems.

To achieve this objective, the following sub-objectives are defined:

- To explore whether accounting for nutritional quality affects the comparison of the environmental impacts of human diets varying in their percentage of ASF;
- To explore whether the units meals and daily diets are equally suitable to compare environmental impacts of diets;
- To identify which factors influence the relation between the human diet, population size, land availability and quality, and minimised land use;
- To assess the potential of preventing and recycling P waste in a food system, to reduce the dependency on phosphate rock;
- To assess the potential of preventing, recycling and recovering waste, to reduce energy input to the food system;

5. Research approach

Firstly, to explore if accounting for nutritional quality affects the comparison of the environmental impacts of human diets varying in their percentage of ASF, a literature review was conducted on environmental impact assessments of human diets. The nutritional quality of these diets was computed, and the environmental impacts of these published diets were then expressed relative to their nutritional quality. The newly developed index (i.e. impact per unit of nutritional quality) was compared with the index as published in the reviewed literature (i.e. impact per meal or diet).

Secondly, from the same literature review, the relevant unit for resource optimisation in food production and consumption is explored by comparing the nutrient scores of meals and daily diets based on their adjusted nutrient score relative to a recommended daily energy intake of 2000 kcal (EFSA, 2009).

Thirdly, to identify which factors influence the relation between the human diet, population size, land availability and quality, and minimised land use, a linear optimisation model is developed. A linear optimisation model has the objective to assess the best outcome, in this case to determine the minimum requirement of land to feed a human population. Land use was optimised for diets that varied in their percentage of protein from animal (PA), from 0% PA, i.e. a vegan diet, to 80% PA. The linear optimisation model is a conceptual representation of a food system consisting of interrelated crop production, animal production, post-harvest processing, and human consumption (Figure 1). The food system is parameterised with crop and animal production data from the Netherlands where the availability and accessibility of data is high. The model is also used to demonstrate land use efficiency in other food systems that differ in population size, and land availability and quality.

Fourth, the optimisation model is extended with the process of waste water treatment to assess the potential of preventing and recycling P waste in a food system, to reduce the dependency on phosphate rock. The model is also used to demonstrate P use efficiency in other food systems that differ in P losses through leaching and run-off.

Finally, the optimisation model is extended with the processes of anaerobic digestion and transport to assess the potential of preventing, recycling and recovering waste, to reduce energy input to the food system.

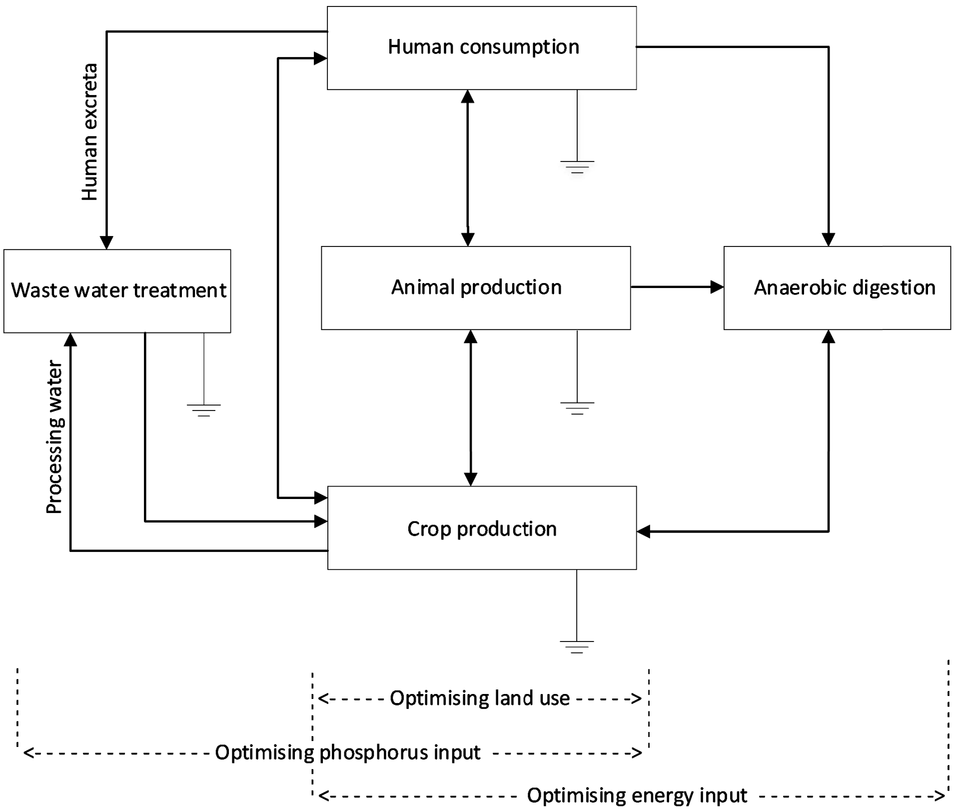

Figure 1. Modelled processes and flows of the food system to assess land use, P input and energy input. Post-harvest processing not shown. The sink-shape indicates wastes and losses.

6. Outline of the thesis

The chapter outline of the thesis is presented in Figure 2. Chapter 2 explores whether accounting for nutritional quality affects the comparison of the environmental impacts of human diets varying in their percentage of ASF. This chapter also explores whether the units meals and daily diets are equally suitable to compare environmental impacts of diets . The selected unit is used as the basis for resource optimisation in the following chapters. Chapter 3 studies the relation between land use, the share of animal protein in the human diet, population size, and land availability and quality using an integrated optimisation approach. Chapter 4 assesses the potential of preventing and recycling P waste in the food system in order to reduce the dependency on phosphate rock. In chapter 5, the potential of preventing, recycling and recovering waste, to reduce energy input to the food system is assessed. Moreover, in Chapter 6 the methodology to assess resource use efficiency is discussed, and implications of increasing resource use efficiency, and conclusions are given.

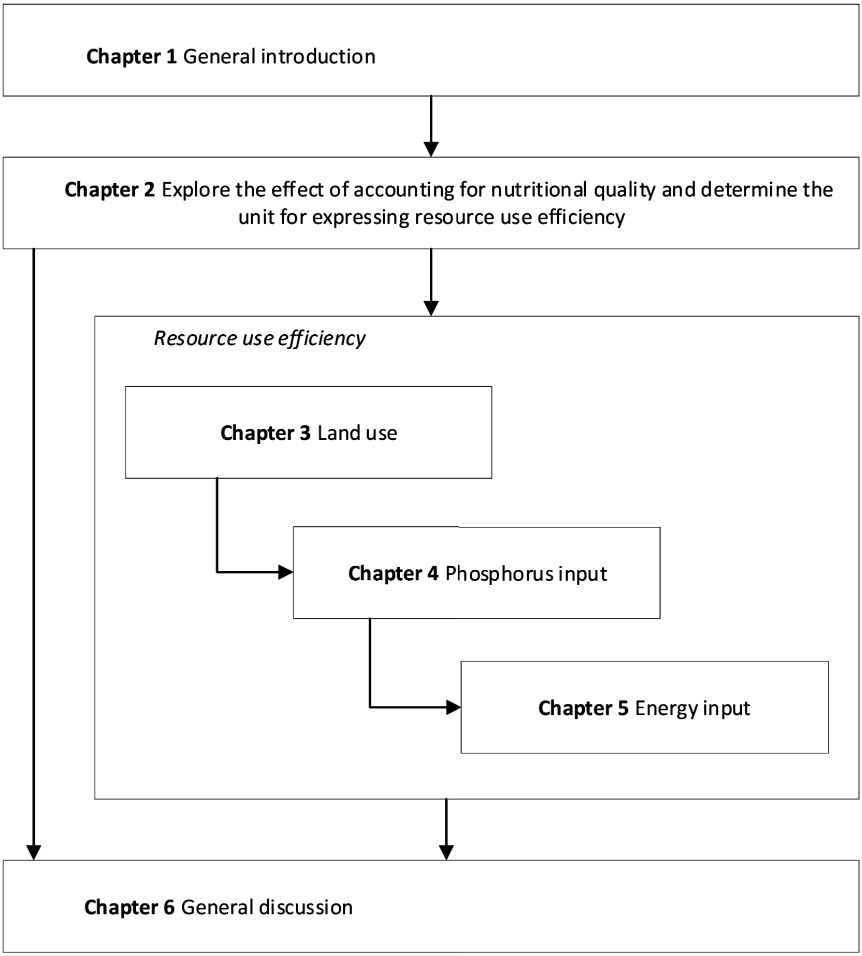

Figure 2. Outline and chapters of the thesis.

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Chapter 2

The effect of nutritional quality on comparing environmental impacts of human diets

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Abstract

Several studies support the general conclusion that plant-based diets have a lower environmental impact than animal-based diets. These studies, however, do not account for the nutritional quality of diets. The main objective of our study, therefore, was to explore if accounting for nutritional quality affects the comparison of the environmental impacts of human diets varying in their percentage of animal-source food products (ASFP). We also explored whether meals or daily diets are equally suitable to compare environmental impacts of diets. Fifty peer-reviewed studies were found that examined the environmental impact of diets, generally using life cycle assessment (LCA). Only 12 of these studies were reviewed, based on five criteria: study contains more than one scenario; diet scenarios vary in their percentage of ASFP; the weight of each food product was provided; the study assessed global warming potential and/or land use; diet scenarios are not designed for specific (health) groups. For each diet described in the reviewed studies, we quantified the daily intake of nine qualifying and three disqualifying nutrients. Global warming potential and land use, as provided by the reviewed studies, were expressed in four ways: per day, per daily protein intake capped to the recommended intake level of 57 g; per daily protein intake uncapped; and per NRD9.3 (i.e. a composite nutrient score of a diet).

We concluded that the nutrient intake resulting from a meal cannot be used to assess the nutritional quality of a daily diet and, hence, the environmental impact of meals cannot be compared to that of daily diets. Studies on meals were therefore excluded from further analysis. Our results further show that daily diets that had higher percentages of ASFP were associated with higher (excess) intakes of total protein and lower values of NRD9.3. Diets that had higher percentages of ASFP were associated with higher GWPs and LU's per gram protein capped and per unit NRD9.3. Without capping protein to the recommended intake level, GWP and LU per gram of protein were generally lower for diets that had higher percentages of ASFP. Without capping, diets with higher percentages of ASFP are credited for overconsumption of protein. Since overconsumption of protein does not benefit health, we recommend capping to the recommended intake level. The effect of using NRD9.3 rather than day as functional unit was small for GWP. For LU we found no effect. When using NRD9.3 as functional unit, it must be considered that this functional unit requires more data than day or protein. Our analysis is based on a limited number of studies. Although initially a substantial number of studies were found, many of these were excluded because insufficient data were provided about diet composition, only one diet scenario was assessed, or because the studies assessed the environmental impact of meals rather than of diets. We found mainly Western-oriented diets, often designed by the researchers and not representative for actual consumption. For further research on the environmental impact of diets, we therefore recommend analysis on representative daily diets.

1.Introduction

Compared to plant-source food products, production and consumption of animal-source food products (ASFP) is generally associated with a high environmental impact (Cordell et al., 2009; Steinfeld, 2006). ASFP can provide, however, high-quality protein and are rich sources of micronutrients (FAO, 2009). A moderate intake of ASFP, therefore, can improve the nutritional adequacy of the poor (FAO, 2009).

Several studies have assessed the environmental impact of human diets that varied in percentage of ASFP (Carlsson-Kanyama and Gonzalez, 2009; Davis et al., 2010; Saxe et al., 2012). Most of these studies used life cycle assessment (LCA) to compare the impacts of two or more diet scenarios. LCA is a holistic method to assess the environmental impact (i.e. emission of pollutants and use of resources) of a product during the entire production chain (Guinée et al., 2002). These studies support the general conclusion that plant-based diets have a lower environmental impact than animal-based diets.

To compare LCA results of different diet scenarios, the results should be expressed on basis of a so-called functional unit (FU) (De Vries and De Boer, 2010). An FU represents the primary function of a system. Beside its social and psychological functions, a main function of food production is to satisfy the human body's need for energy and nutrients, such as protein, iron, fibre, vitamins and minerals. Studies that compared the environmental impact of food production focussed on its nutritional function, and, therefore, generally used 'meal' or 'daily diet' as FU. Meals and daily diets within studies were often comparable in terms of energy, protein and fat content. Studies that used a meal or daily diet as FU, however, did not account for the overall nutritional quality of a diet. Accounting for nutritional quality was done by Smedman et al. (2010). They compared greenhouse gas emissions relative to the so-called nutrient density score (NDS) of beverages. NDS is based on individual nutrient scores. The latter express the nutrient contents of food relative to the nutrient requirements (Hansen, 1973). By summing the individual nutrient scores, NDS represents the composite nutrient score of a product. Considering the nutrient density in environmental comparisons of food products may lead to different conclusions compared with traditional FUs, and, consequently, to different recommendations about how to alter consumer choices to the benefit of the environment (Smedman et al., 2010). To our knowledge, no study exists that compared LCA results of diet scenarios while accounting for overall nutritional quality.

The main objective of our study, therefore, was to explore if accounting for nutritional quality affects the comparison of the environmental impacts of human diets varying in their percentage of ASFP. We thus reviewed studies that used LCA to evaluate the environmental impact of diets varying in percentage of ASFP. We observed that these studies were generally based on a comparison among meals or daily diets. An additional objective, therefore, was to explore whether meals or daily diets are equally suitable to compare environmental impacts of diets.

To fulfil these objectives, we used the environmental impacts as published in selected studies and expressed these impacts relative to the protein concentration of the diet or the nutritional quality of the diet. The nutritional quality of the diet was computed based on the Nutrient Rich Food 9.3 (NRF9.3) score of a diet. We chose NRF9.3 out of available nutritional quality scores (Darmon et al., 2009; Fulgoni et al., 2009), as it was best validated against the Healthy Eating Index (Fulgoni et al., 2009). We computed the protein percentage or the nutritional quality score of the diet based on information given in published papers.

2. Material and methods

2.1 Selection of studies

We searched literature in Scopus and Web of Science. Our search terms were: diet, food, meal, human nutrition, consumption pattern, life cycle, footprint, environment, greenhouse, and land use. We defined the following inclusion criteria:

- the study contains more than one within-country diet scenario; Multiple scenarios per study were required, as studies define different system boundaries, and, hence, scenarios could only be compared within study.
- diet scenarios within studies vary in their percentage of animal source food product (ASFP);
- the weight of each food product included in the diet scenarios is given, or could be provided by the author(s);
- the study assesses global warming potential, land use or both;
- diet scenarios are not designed for specific groups (e.g. infants, people with health problems), and if diets are gender specific, we should be able to average these. Scenarios designed for specific groups were excluded, as individuals in these groups may have specific nutrient requirements, while the nutrient density score was computed using average nutrient requirements per person;
- the study is published in a peer-reviewed scientific journal.

2.2 Calculation of individual nutrient scores of each diet

For each diet scenario we quantified the daily intake of nine qualifying and three disqualifying nutrients when consuming the diets. The nine qualifying nutrients were (the recommended daily value (RDV) is given in brackets): protein (57 g) (EFSA, 2012), fibre (25 g) (EFSA, 2010), calcium (800 mg), iron (14 mg), magnesium (375 mg), potassium (2000 mg), and vitamins A (800 μg), C (80 mg) and E (12 mg) (EU, 2008). The three disqualifying nutrients were (the maximum recommended value (MRV) given in brackets): sodium (2400 mg), saturated fat (20 g) and total sugar (90 g) (EFSA, 2009). To quantify the daily intake of these 12 nutrients, we multiplied the daily intake of each food product in the diet scenario by the nutrient content of that food product. Contents of the 12 nutrients in various food products were derived from the Dutch nutrients database NEVO (NEderlands VOedingsstoffenbestand) (RIVM, 2011). The description of food products in the included studies was often less detailed than the description of food products in NEVO. We linked each food product in included studies to a product in NEVO, by using the following successive criteria:

- exact match between product description in the included studies and in NEVO. Unless the product was specified as 'raw' in the included studies, we chose the food product in NEVO in the form in which it would be consumed (e.g. boiled or otherwise prepared). When the product was only available in raw form, we chose this;
- match between product description in the included studies and in NEVO, with the additional description of 'average' in NEVO;
- the variant of the food product (e.g. 'apple juice' as a variant of the food group 'fruit juice') which had the highest consumption rate within the food group according to the Dutch National Food Consumption Survey (RIVM, 2010) in the age of 19-69 years.

Moreover, included studies did not mention the exact cut of meat consumed. We formulated, therefore, a composite meat cut per livestock species. The composite meat cut was created by combining cuts per livestock species that together sum up to a minimum of 60% of Dutch consumption volumes (RIVM, 2010), starting with the cuts that are consumed most. The consumption volumes of the various selected meat cuts form the weighing basis for computing the nutrient content of the composite cut. For the various selected meat cuts we chose the form in which it would be consumed when available in NEVO.

Diet scenarios found in literature could either be single meals, daily diets or accumulated annual diets. To enable a comparison of nutrient scores among these diet scenarios, we adjusted the scores of meals to daily scores, by expressing its nutrient score relative to a recommended daily energy intake of 2000 kcal (EFSA, 2009). In the following sections, we refer to meals that were adjusted to 2000 kcal daily diets as 'adjusted meals'. Nutrient scores of annual diets were adjusted to daily scores by dividing the intake of food products by 365 and were referred to as daily diets.

To evaluate the nutritional quality of each diet we computed the Nutrient Rich (NR) score for each qualifying nutrient (Eq. 1). The score expresses nutrient intake relative to the RDV of the nutrient. We either used the actual intake of nutrients, which we refer to as the uncapped intake (e.g. actual daily protein intake is Protein Uncapped (PU), or the so-called capped intake. Capping of the intake (defined as nutrient intake is equal to RDV if intake ≥ RDV) was applied to avoid crediting of overconsumption (Drewnowski, 2009). We refer to the capped daily intake of protein as Protein Capped (PC).

$$NR_{nutrient} = \frac{nutrient_{uncapped\,/\,capped}}{RDV_{nutrient}} \times 100 \qquad \text{Eq.1}$$

where $\mathrm{NR}_{\mathrm{nutrient}}$ is the Nutrient Rich score for a nutrient, $nutrient_{uncapped\,/capped}$ is the amount (in g) of daily nutrient intake with or without capping, and $RDV_{nutrient}$ is the Recommended Daily Value of the nutrient.

2.3 Calculation of the composite nutrient score of each diet

To quantify the composite nutritional quality of a food product, we used the principles of the Nutrient Rich Food 9.3 (NRF9.3) score (Drewnowski, 2009; Fulgoni et al., 2009). The NRF9.3 score reflects the composite nutritional quality of a food product per 100 kcal.
To quantify the composite nutritional quality of a diet, we adapted NRF9.3 into the Nutrient Rich Diet 9.3 (NRD9.3) score. In contrast to the NRF9.3 score, which is expressed per 100 kcal of a food product, NRD9.3 is not scaled to energy intake.
The NRD9.3 score consists of a total nutrient rich 9 sub-score (TNR9) and a total limiting 3 sub-score (TLIM3). The TNR9 sub-score (Eq. 2) sums the percentages of the RDV in the diet for each of the nine qualifying nutrients, while capping nutrient intake at 100% of RDV. The TLIM3 sub-score (Eq. 3) sums the percentages of the MRV for each of the three disqualifying nutrients. The NRD9.3 score (Eq. 4) of a diet was computed by subtracting the TLIM3 sub-score from the TNR9 sub-score.

$$TNR9 = \sum_{i=1}^{i=9} \frac{nutrient_{i,capped}}{RDV_i} \times 100 \qquad \text{Eq. 2}$$

$$TLIM3 = \sum_{i=1}^{i=3} \frac{nutrient_i}{MRV_i} \times 100 \qquad \text{Eq. 3}$$

$$NRD9.3 = TNR9 - TLIM3 \qquad \text{Eq. 4}$$

where $nutrient_i$ is the daily intake of nutrient i in the diet, RDV_i is the Recommended Daily Value of nutrient i, MRV_i is the Maximum Recommended Value of nutrient i.

2.4 Environmental impact of diets and FUs

Global warming potential (GWP) and land use (LU) used in our analysis were derived from studies reviewed. GWP was determined as the sum of emissions of greenhouse gasses, including e.g. CO_2, CH_4 and N_2O, weighed on their equivalence factor in terms of CO_2 equivalents (IPCC, 2007). Moreover, GWP and land use (LU) were expressed as GWP/meal and LU/meal, or as GWP/day and LU/day, or in other words, against 'meal' or 'day' as the FU. In our study, we expressed GWP and LU relative to four FUs: 'PU' (daily dietary protein intake uncapped, expressed in 100 g protein), 'PC' (daily dietary protein intake capped to a maximum of 57 g, expressed in 100 g protein), 'NRD9.3' and 'day'. Subsequently, we explored relations between GWP/FU and LU/FU (dependent variable), on the one hand, and the percentage of animal protein (AP%) in the human diet on the other (independent variable). AP% is the dietary protein supplied by ASFP expressed as percentage of the total dietary protein. In most studies, these relations were curvilinear. For further analysis across studies we defined for GWP/FU and LU/FU the Relative Increase from 0 to 65 AP% (RI_{65}). This was done by regressing GWP/FU and LU/FU for each study on AP% by a quadratic model (Eq. 5)

$$\text{GWP/FU or LU/FU} = a + b1 \times AP\% + b2 \times AP\%^2 \qquad \text{Eq. 5}$$

Subsequently, we indexed the data for each diet scenario within study for each FU such that scenarios with AP% =0 were fixed at a GWP/FU or LU/FU value of 100. This basically implies that we divided the left and right hand term of Equation 5 by a/100 (Eq. 6).

$$\text{GWP/FU}_{indexed} \text{ or LU/FU}_{indexed} = 100 + (100 \times b1/a) \times AP\% + (100 \times b2/a) \times AP\%^2 \qquad \text{Eq. 6}$$

Studies that did not include diet scenarios containing an AP% of less than 40 were excluded from environmental analysis to avoid unrealistic and negative estimates of the intercept as a result of over-extrapolation.

The RI_{65} was calculated as $\Delta y/\Delta x$ with Δy as the difference between the model estimate for AP% = 65 and the model estimate for AP% =0 and Δx is 65. The RI_{65}, consequently, represents the estimated difference of the GWP/FU or LU/FU per unit AP% between a diet with 65%AP and a (vegan) diet with 0%AP. The level of 65%AP was arbitrarily chosen. It represents the common AP% of diets that contain ASFP in the studies included.

2.5 Statistical analyses

For the comparison of homogeneity of variance between adjusted meals and daily diets, we also adjusted daily diets by expressing nutrient intakes of daily diets relative to an intake of 2000 kcal. The intake level of 2000 kcal was chosen as this was also the daily energy reference used to compute NRF9.3 scores of foods (Drewnowski, 2009; Fulgoni et al., 2009). We performed Levene's test for homogeneity of variance to test whether the variances for the 12 nutrients were similar for adjusted meals and adjusted daily diets. The outcomes of Levene's test were used to explore whether meals or daily diets are equally suitable to compare environmental impacts of diets varying in percentage of AP%.

To test whether $GWP/FU_{indexed}$ or $LU/FU_{indexed}$ differed between FUs: 'PU', 'PC' and 'NRD9.3', on the one hand, and 'day', on the other hand, we used paired t-tests to tests for the differences in RI_{65}. To test whether the average RI_{65} for an FU differed from 0 across studies we conducted a two-sided t-test. For both tests we assumed normal distributions.

Statistical analyses were done by the program SPSS Statistics 19. Differences with a p value of less than 0.05 were considered to be statistically significant.

3. Results

3.1 Characteristics of diets in selected studies

We found 50 studies from peer-reviewed scientific journals that examined the environmental impact of within-country diets. The following 38 studies did not meet our inclusion criteria: (Ascione et al., 2008; Baroni et al., 2007; Berners-Lee et al., 2012;

Bleken and Bakken, 1997; Browne et al., 2008; Calderón et al., 2010; Carlsson-Kanyama et al., 2003; Chen et al., 2010; Coley et al., 1998; Du et al., 2006; Eshel and Martin, 2006; Faist et al., 2001; Gerbens-Leenes et al., 2002; Gerbens-Leenes and Nonhebel, 2005; Klein-Banai and Theis, 2011; Kok et al., 2006; Kramer et al., 1999; Kytzia et al., 2004; Li et al., 2008; Liu and Savenije, 2008; Meier and Christen, 2012; Muñoz et al., 2010; Nijdam et al., 2005; Pimentel and Pimentel, 2003; Renault and Wallender, 2000; Risku-Norja and Maenpaa, 2007; Saarinen et al., 2012; Sonesson et al., 2005; Vieux et al., 2012; Vintila, 2010; Vintila, 2010; Virtanen et al., 2011; Wallen et al., 2004; Wallgren and Höjer, 2009; Wang et al., 2011; Weber and Matthews, 2008; Xue and Landis, 2010; Zufia and Arana, 2008).

Studies meeting our selection criteria are presented in Table 1. Most selected studies assessed western diets. The various scenarios within studies generally were described as having comparable levels of energy, protein, carbohydrates and/or fat.

The system boundary of the studies varied. Although all studies included crop cultivation in the field, upstream processes (e.g. production and transport of inputs for cultivation, such as seeds, fertilisers, pesticides and diesel) and downstream processes (e.g. transport to processing plants and retail) were not always included.

Diet scenarios described in the included studies were either individual meals or daily or yearly per capita diets (appendix Table A.1). Some studies formulated diet scenarios with specific characteristics, such as food products produced locally or abroad, or food products from conventional or organic production systems. To make scenarios comparable, we grouped scenarios according to these specific characteristics. The grouped scenarios within study were indicated with a capital letter behind the reference in Table A.1 in the appendix.

Table 1. Overview of some characteristics of studies included in this paper

	Reference	Country	Impact categories	Nutrients for which diets are claimed to be comparable[a]
1	Davis et al., 2010	ES, SE	GWP, EU, AP, EP, POFP, SODP	Energy, protein, fat[b]
2	Carlsson-Kanyama, 1998	SE	GWP	Energy and protein
3	Davis and Sonesson, 2008	SE	GWP, EU, AP, POCP	Energy, protein, fat, carbohydrates
4	Thibert and Badami, 2011	CA	LU, EU, WU	Energy, protein, fat
5	Carlsson-Kanyama and González, 2009	SE	GWP	None/unknown
6	Gerbens-Leenes and Nonhebel, 2002	NL	LU	None/unknown
7	Risku-Norja et al., 2009	FI	GWP	Energy, protein, fats, carbohydrates
8	Collins and Fairchild, 2007	UK	LU	14 nutrients[c]
9	Risku-Norja et al., 2008	FI	GWP, LU, AP, NB, PB, LD	Protein, fats, carbohydrates
10	Peters et al., 2007	US	LU	Energy
11	Pathak et al., 2010	IN	GWP	Energy, fat, carbohydrates[d]
12	Saxe et al., 2012	SE, DK	GWP	Energy, protein

[a]As described by the studies, [b]and overall size of the meals are reasonable, [c] Not further specified, [d] including other not further specified nutrients.
Note: ES=Spain, SE=Sweden, CA=Canada, NL=The Netherlands, FI=Finland, UK=United Kingdom, US=United States, IN=India, DK=Denmark, GWP=Global Warming Potential, EU=Energy Use, AP=Acidification Potential, EP=eutrophication potential, POFP=Photo Oxidant Formation Potential, SODP= Stratospheric Ozone Depletion Potential, POCP= Photochemical Ozone Creation Potential, LU=Land Use, WU=Water Use, NB=Nitrogen Balance, PB=Phosphorus Balance, LD=Landscape Diversity.

Some of the scenarios represented the actual, average or representative annual diet of a country or region, such as the average Finnish diet (Risku-Norja et al., 2008) or the actual diet of residents of Cardiff (Collins and Fairchild, 2007). Other scenarios were self-defined alternatives to these diets, such as the pork-and-poultry-free diet by Risku-Norja et al. (2008) and a diet in which food products with high land use per kg were replaced by those with lower land use (Collins and Fairchild, 2007). The number of scenarios within studies varied between two and three for studies on meals, and between two and seven for studies on daily diets (Table A.1). The number of products that the scenarios contained also varied. Meals contained a median of five food products whereas daily diets contained a median of 23 products. The percentages of ASFP in the diets ranged from 0% in vegan scenarios (diet without any ASFP) to over 85% in a meal scenario by (Thibert and Badami, 2011) and 78% in a daily diet scenario by (Peters et al., 2007).

3.2 Nutritional quality of adjusted meals

The adjusted meals showed an intake of fibre, vitamin A and vitamin C exceeding 500% of the RDV (results not shown). In the study of Carlsson-Kanyama (1998), vitamin A intake of adjusted meals even exceeded 1000% of RDV, which can be attributed to the carrots in the diets. Using Levene's test for homogeneity of variance, we found significantly ($p < 0.05$) higher variances for adjusted meals than for adjusted daily diets for protein, calcium, iron, vitamin A and vitamin C, and significantly lower variances for saturated fat and total sugar.

The extreme nutrient intakes and the large variances we found for adjusted meals resulted from the relatively small number of food products in meals (Table A.1). The inclusion of a nutrient dense product such as carrots highly affects the nutrient intake related to an adjusted meal. Adjusting meals to daily diets is also not reasonable, since a meal, e.g. breakfast or dinner, is not eaten three times a day.

We, therefore, concluded that the nutrient intake resulting from a meal cannot be used to assess the nutritional quality of a daily diet. Following this, we concluded that meals cannot be used to compare environmental impacts of human diets that vary in percentage of ASFP. As a results, our analysis of GWP and LU for diets focussed exclusively on daily diets.

3.3 Nutritional quality of daily diets

Table A.2 in the appendix shows the energy and nutrient intakes resulting from the daily diet scenarios. The table shows that most diets exceeded the energy recommendation of 2000 kcal, with a maximum of 3095 kcal for the first scenarios by Saxe et al. (2012). All diets exceeded the recommended daily intake of 57 g protein, except for the vegan scenario by Risku-Norja et al. (2008). Vitamin A was the nutrient with the largest variation, ranging from 52 µg (6.5% of RDV) in the vegetarian (diet without meat) diet by Pathak et al. (2010) to 5474 µg (684% of RDV) in the New Nordic Diet by Saxe et al. (2012). Most scenarios exceeded the MRV of total sugar while only four scenarios exceeded the MRV of sodium. The inclusion of sugar as single product was often reported whereas salt as single product (e.g. table salt) was often not. The MRV of sodium was especially exceeded in the scenarios by Saxe et al. (2012), which was the only study reporting salt as single product. In the scenarios by Saxe et al. (2012), salt contributed about 28 percent to total sodium content.

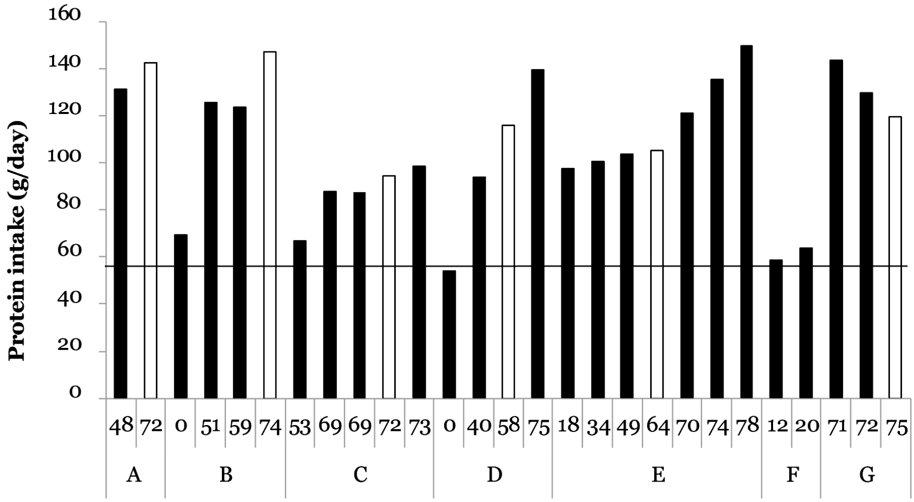

Figure 1. Daily intake of protein for each scenario. Capital letters on the x-axis indicate the reviewed studies. Numbers on the x-axis represent the AP% (= 100 x protein from animal source food products/total dietary protein). Within study, diets were ranked by increasing AP%. The horizontal line shows the recommended daily value of 57 grams protein. The blank bar within each study indicates the average, representative diet within the studied country or region. A: Gerbens-Leenes and Nonhebel (2002); B: Risku-Norja et al. (2009); C: Collins and Fairchild (2007); D:Risku-Norja et al. (2008); E: Peters et al. (2007); F: Pathak et al. (2010); G: Saxe et al. (2012).

Figure 1 shows the daily intake of protein for each diet scenario within studies (the different studies are indicated by capital letters at the horizontal bar). Within study, scenarios are ranked in order of AP%, ranging from 0 to 78%. The horizontal line indicates the RDV of protein of 57 g. Most studies have one blank bar, which represents the average, representative diet scenario for the studied country or region. Total protein intake ranged from 54 g (94% of RDV) in the vegan scenarios by Risku-Norja et al. (2008) to 150 grams (263% of RDV; (Peters et al., 2007). Within study, the total and excess intake of protein generally increased with increasing AP%. Figure 2 shows the association between daily intake of protein and AP% across studies. Across studies, protein intake is positively associated with AP% ($p = 0.000$).

Figure 3 shows the NRD9.3 score for each diet scenario. The relation between NRD9.3 score and AP% varied between studies. Some scenarios had a decreasing trend and others an increasing trend of the NRD9.3 score with increasing percentage of ASFP. Figure 4 shows the association between daily NRD9.3 intake and AP% across studies. Across studies, NRD9.3 intake is negatively associated with AP% ($p = 0.001$).

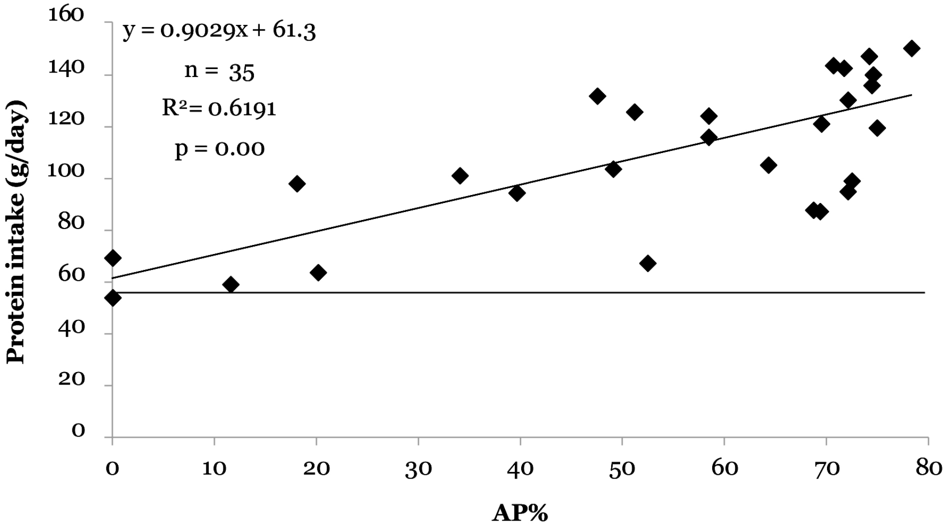

Figure 2. The association between protein intake (g/day) and AP% (= 100 x protein from animal source food products/total dietary protein) across studies. The horizontal line represents the recommended daily value of protein of 57 g.

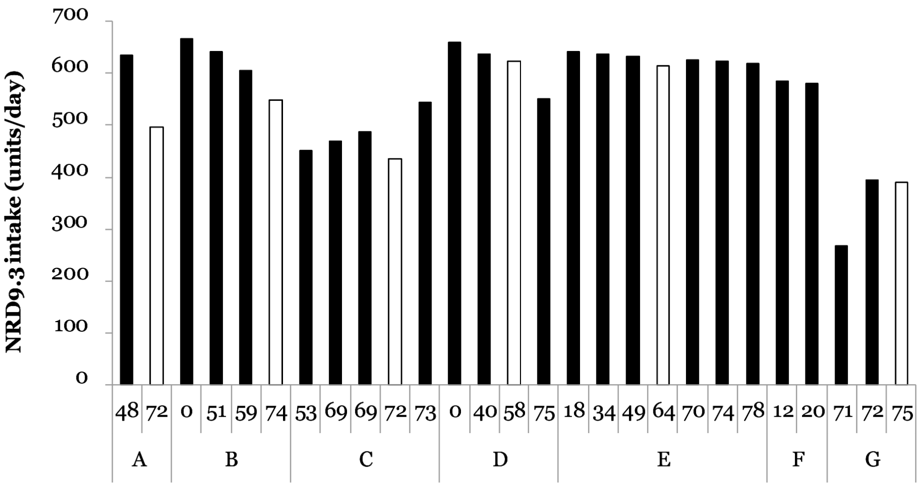

Figure 3. NRD9.3 scores for each diet scenario. Capital letters on the x-axis indicate the reviewed studies. Numbers on the x-axis represent the AP% (= 100 x protein from animal source food products/total dietary protein). Within study, diets were ranked by increasing AP%. The blank bar within each study indicates the average, representative diet within the studied country or region. A: Gerbens-Leenes and Nonhebel (2002); B: Risku-Norja et al. (2009); C: Collins and Fairchild (2007); D:Risku-Norja et al. (2008); E: Peters et al. (2007); F: Pathak et al. (2010); G: Saxe et al. (2012)

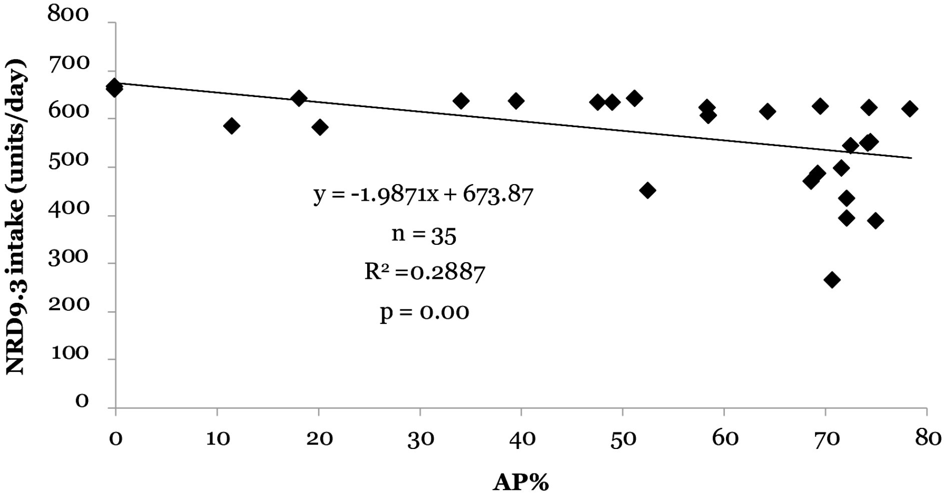

Figure 4. The association between NRD9.3 intake (units/day) and AP% (= 100 x protein from animal source food products/total dietary protein) across studies.

Table A.2. shows the intake levels of other nutrients. The association between intake of nutrients and AP% is given in Table 2. Significant positive associations were found for protein, calcium, vitamin A, sodium, saturated fat and total sugar. Significant negative associations were found for fibre, iron, magnesium and NRD9.3.

Table 2. The Pearson correlation coefficient between nutrient intake and AP%, and corresponding p-values

	Pearson correlation	p-value
Energy	0.16	0.35
Protein	0.79	0.00
Fibre	-0.57	0.00
Calcium	0.61	0.00
Iron	-0.46	0.00
Magnesium	-0.42	0.01
Potassium	0.12	0.49
Vitamin A	0.37	0.03
Vitamin C	-0.13	0.44
Vitamin E	-0.01	0.96
Sodium	0.48	0.00
Sat. fat	0.60	0.00
Total sugar	0.56	0.00
NRD9.3	-0.54	0.00

3.4 Global warming potential of diets

Four studies evaluated GWP of daily diets (Pathak et al., 2010; Risku-Norja et al., 2008; Risku-Norja et al., 2009; Saxe et al., 2012). Results taken from the articles by Risku-Norja et al. (2008, 2009) were averaged across diets containing food products from conventional agriculture (A) and across diets containing food products from organic (B) agriculture. Results from Saxe et al. (2012) were not included; to index data from this study we had to extrapolate for more than 40 AP% to estimate the intercept (Table A.1).

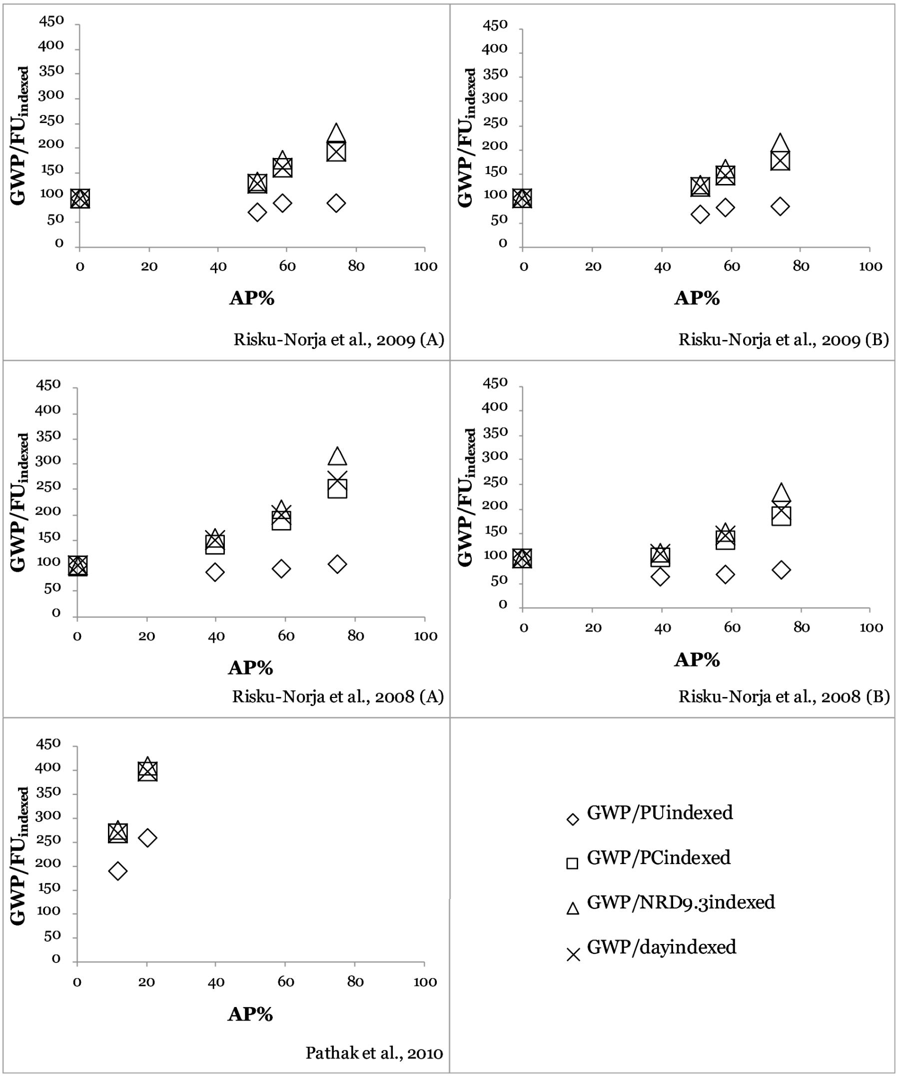

Figure 5. Associations between GWP/FU$_{indexed}$ and AP% A: conventional food products; B: organic food products

Figure 5 illustrates the associations between the GWP/$FU_{indexed}$ and AP%. The studies by Risku-Norja et al. (2009, 2008) covered diets with AP% ranging from 0 to 75, while those of Pathak et al. (2010) covered AP% from 12 to 20.

For each FU within studies the RI_{65} for GWP/$FU_{indexed}$ was estimated (see Table 3). Averaged across studies RI_{65} did not differ significantly ($p > 0.05$) from 0 for any of the FUs 'PU', 'PC', 'NRD9.3' and 'day'. This was because RI_{65} estimates for GWP/$FU_{indexed}$ showed large variation among studies from different authors (Table 3). Conclusions across studies regarding the increase of GWP/ $FU_{indexed}$ per AP%, therefore, require study on more harmonized diets.

Across studies the average RI_{65} GWP/$day_{indexed}$ was 3.93 (Table 3), and tended ($p = 0.08$) to be higher than the average RI_{65} GWP/$PU_{indexed}$, and was significantly ($p = 0.001$) lower than the average RI_{65} GWP/$NRD9.3_{indexed}$. These differences were, however, relatively small. No significant differences were found between the average RI_{65} for the FU 'day' and the RI_{65} of the FU 'PC'.

Table 3. RI_{65} of GWP/$FU_{indexed}$ for the FUs 'PU', 'PC' 'NRD9.3' and 'day'

	RI_{65} of GWP/$FU_{indexed}$			
	PU	**PC**	**NRD9.3**	**Day**
Risku-Norja et al., 2008 (A)	-0.07	1.73	2.38	1.93
Risku-Norja et al., 2008 (B)	-0.47	0.85	1.34	1.00
Risku-Norja et al., 2009 (A)	-0.22	1.06	1.44	1.06
Risku-Norja et al., 2009 (B)	-0.32	0.87	1.23	0.87
Pathak et al., 2010	7.92	14.75	15.35	14.75
RI_{65} averaged across studies	1.37	3.86	4.36	3.93
Significance of difference from 0	0.45	0.23	0.19	0.22
Significance of difference between FU and 'day '	0.08	0.18	0.00	--

3.5 Land use of diets

Four studies evaluated the LU of daily diets (Collins and Fairchild, 2007; Gerbens-Leenes and Nonhebel, 2002; Peters et al., 2007; Risku-Norja et al., 2008). Results from Gerbens-Leenes (2002) and Collins and Fairchild (2007) were not included; to index data from these studies we had to extrapolate for more than 40 AP% to estimate the intercept (Table A.1). The studies by Risku-Norja et al. (2008) and Peters et al. (2007) cover diets in a range from AP% 0 to nearly 80. Results taken from the article by Risku-Norja et al. (2008) were averaged across diets containing food products from conventional agriculture (A) and across diets containing food products from organic (B) agriculture.

Figure 6 illustrates the associations between LU/FU$_{indexed}$ and AP%.

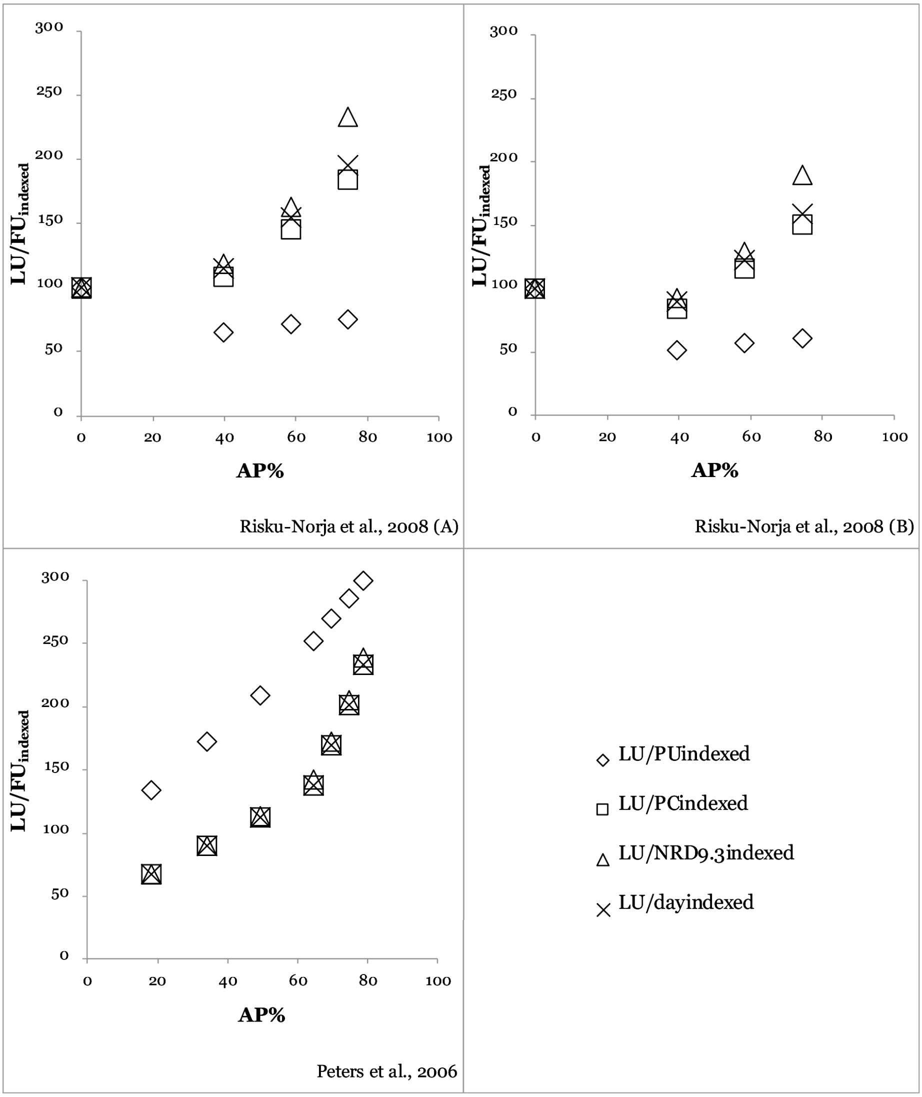

Figure 6. Associations between LU/FU$_{indexed}$ and AP%. A: conventional food products; B: organic food products

For each FU within studies the RI_{65} for LU/FU$_{indexed}$ was estimated (see Table 4). Averaged across studies RI_{65} differed significantly ($p < 0.05$) from 0 for FUs 'PC', 'NRD9.3' and 'day', but not ($p > 0.05$) for the FU 'PU'. This was because RI_{65} estimates for $LU/FU_{indexed}$ showed large variation among studies from different authors (Table 4). Conclusions across studies regarding the increase of LU/ $FU_{indexed}$ per AP%, therefore, require study on more harmonised diets.

Across studies the average RI_{65} $LU/day_{indexed}$ was 0.82 (Table 4), not significantly different ($p > 0.05$) from the average RI_{65} of the other PUs. The FU 'day' thus gave no different contrasts between diets differing in AP% with regard to $LU/FU_{indexed}$ than the FUs 'PU', 'PC', and 'NRD9.3'.

Table 4. RI_{65} of $LU/FU_{indexed}$ for the FUs 'PU', 'PC' 'NRD9.3' and 'day'

	RI_{65} of $LU/FU_{indexed}$			
	PU	**PC**	**NRD9.3**	**Day**
Peters et al., 2006	2.40	0.89	0.94	0.89
Risku-Norja et al., 2008 (A)	-0.45	0.91	1.39	1.06
Risku-Norja et al., 2008 (B)	-0.68	0.41	0.80	0.53
RI_{65} averaged across studies	0.42	0.73	1.04	0.82
Significance of difference from 0	0.71	0.05	0.03	0.03
Significance of difference between FU and 'day '	0.72	0.19	0.13	--

4. Discussion and conclusions

Although initially about 50 studies from peer-reviewed journals were found that examined environmental impacts of nationally oriented diets, only 12 studies met our inclusion criteria. The 38 studies that did not meet our inclusion criteria were excluded mainly because insufficient data were provided about diet composition or because only one diet scenario was assessed. Out of the 12 studies that did meet our inclusion criteria, five were not suitable to compare environmental impacts of diets that varied in AP% because they were based on meals rather than on diets. Our analysis, therefore, included only a limited number of studies.

To be conclusive about the effect of increasing AP% on environmental impacts, the representativeness of the dietary scenarios is essential; we recommend, therefore, studies that compare environmental impacts of actual diets consumed across a long period by individuals or groups with distinct consumption behaviour.

We adapted the NRF9.3 score (Drewnowski, 2009) into the NRD9.3 score, to compute the composite nutritional quality at the diet level. Smedman et al. (2010) applied a nutrient density score to compare the environmental impact of various beverages.

They implemented a weighing factor to benefit beverages contributing to more than 5% of the recommended intake of a nutrient. Scarborough and Rayner (2010) contested this approach as this arbitrary factor highly affected the environmental ranking of products. To our knowledge, this weighing factor is not implemented in other applications of the nutrient density score, and we therefore did not implement this factor.

The present nutrient density concept, moreover, does not account for nutrient quality and bioavailability. Compared to plant source foods, animal source foods are associated with higher quality of protein and higher bioavailability of iron (Drewnowski and Fulgoni, 2008; Hallberg, 1981; Otten et al., 2006). Accounting for nutrient quality and bioavailability, e.g. by weighing (Drewnowski and Fulgoni, 2008), could increase the NRF9.3 score of single food products. When diets are assessed, effects of dietary interactions on nutrient quality and bioavailability will have to be considered (FAO/WHO, 1991; Otten et al., 2006).

Nutrient intake

Our analysis showed that diets with higher AP% had higher excess intakes of protein compared to diets with lower AP%. If this can be extrapolated to diets that are deficient in protein (hence provide less than 57 g of protein daily), than an increase in consumption of ASFP could positively benefit health. For diets containing sufficient protein, however, an increase in consumption of ASFP does not lead to health benefits. In our analysis, iron intake was generally higher for diets with lower AP%. This is in contrast with what we expected, as iron intake was generally lower in vegetarian and vegan diets (FAO, 2009; O'Neill, 2010). Vegan diets were also associated with less intakes of fibre, magnesium and vitamin C compared with vegetarian and omnivorous diets (O'Neill, 2010), while in our study we found a decreasing trend for intake of these nutrients with increasing AP%. We found, however, an increasing trend for intake of calcium, vitamin A, potassium, sodium and saturated fat intake with increasing AP%, as we also expected based on O'Neill (2010). NRD9.3 intake was negatively associated with AP%. At higher AP%, the variance in NRD9.3 intake also increases. The significance in decreasing trend can be caused by a study-effect, e.g. the study by Saxe et al (2012) where diet scenarios have relatively low NRD9.3 intake levels, but it can also be caused by the risk of consuming higher levels of disqualifying nutrients.

Environmental impacts

Because of the negative aspect of overconsumption of protein, diets should not be credited for overconsumption when comparing environmental impacts. Crediting occurs when environmental impacts were expressed as GWP/PU or LU/PU. The increase of environmental impacts with increasing AP% was insignificant when expressed by these parameters. However, if only the nutritionally relevant protein was taken into account, hence when GWP/PC and LU/PC were estimated, impacts increased more strongly with increasing AP%. When expressing the environmental impact of a diet relative to the intake of qualifying nutrients in the diet, we thus propose capping to prevent crediting overconsumption. In our study, we did not credit overconsumption. We could, however, even have sanctioned overconsumption, but this requires further study since the impact of overconsumption differs between nutrients and literature quantifying such impacts is limited. Due to capping of protein, at intake levels higher than the RDV GWP/PC is similar to GWP/ (day*57) and therewith fully correlated with GWP/day. The same holds true for LU/PC and LU/day. Most of the included scenarios had protein intake levels above the RDV and thus little difference was found between RI_{65} between FUs 'PC' and 'day'. As stated before, the number of studies was limited. Our statistical analysis on 5 studies (of 2 scientific groups) for GWP and 3 studies only for LU revealed a non-significant increase for GWP/PC and GWP/day with increasing AP% and a significant increase of LU/PC and LU/day.

NRD9.3 is a holistic dietary quality parameter. This parameter credits qualifying nutrients up to their RDV and sanctions for disqualifying nutrients. Many of the dietary scenarios in the included studies had intake levels of qualifying nutrients above RDV and had, as a consequence, not much variation in their summed score of qualifying nutrients. Similarly, the summed score for disqualifying nutrients did not differ much between diets, partly because only few studies accounted for consumption of salt, snacks and (soft-) drinks. Nevertheless, Figure 4 showed reduced NRD9.3 with increasing AP% and consequently the contrast between diets with low and high AP% was slightly, but significantly higher for GWP/NRD9.3 than for GWP/day. However, this contrast was insignificant for LU.

The present study is not conclusive in the sense that it reveals the best, most relevant or most practically feasible functional unit for comparison of diet related environmental impacts. When protein is supplied in excess of the RDV, protein as FU gives similar results as the FU 'day' because of the capping. When all qualitative nutrients are in excess, the disqualifying nutrients will determine variation in NRD9.3 and the difference of NRD9.3 with the FU 'day'. In diets with nutritional deficiencies or with high levels of disqualifying nutrients, protein (as PU) and 'NRD9.3' may be more appropriate FUs than 'day' because they account for such deficiencies and undesirable components. Single meals, however, are not a good basis for study since they may have unrepresentative imbalances between nutrients, while author-defined diets may have a too well balanced nutritional profile because authors purposively aimed for this.

On the basis of the present study we therefore recommend that future research to study impacts of AFSP on environmental parameters be done on actual diets of individuals or groups with distinct consumption behaviour. In addition, the studies ignored the fact that besides food (meat, milk, and eggs), humans use other animal products, like leather for clothing and manure as a fertiliser or producer of bio-energy. If we did not produce animal products, we would then require plant-based or artificial sources to replace these non-food products and environmental impacts associated with such replacements should be taken into account. For further research, therefore, we recommend the application of a consequential life cycle approach in which the various food and non-food functions of animal production systems are taken into consideration.

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Appendices

Table A.1 The diet scenarios per study, number of products per scenarios, and the % of ASFP and weight of the various diet scenarios.

Reference	Functional unit	Scenario	Scenario number	# Products[a]	ASFP (%)	Weight (100 g)
Davis et al., 2010	Meal	Swedish meal Pea burger	1	4	9.12	7.45
Davis et al., 2010	Meal	Swedish meal soy pork chop	2	5	63.2	9.40
Carlsson-Kanyama, 1998 (A)	Meal	Vegan local meal	1	3	0.00	4.80
Carlsson-Kanyama, 1998 (A)	Meal	Animal local meal	2	3	61.2	4.80
Carlsson-Kanyama, 1998 (B)	Meal	Vegan global meal	1	3	0.00	2.50
Carlsson-Kanyama, 1998 (B)	Meal	Animal global meal	2	4	83.6	3.70
Davis and Sonesson, 2008	Meal	Semi-prepared meal	1	7	52.1	7.30
Davis and Sonesson, 2008	Meal	Home-made meal	2	7	69.8	7.65
Thibert and Badami, 2011 (A)	Meal	Vegan meal, season, local, Conv.	1	5	0.00	9.00
Thibert and Badami, 2011 (A)	Meal	Meat-based meal, local, Conv.	2	5	86.7	5.00
Thibert and Badami, 2011 (B)	Meal	Vegan meal, imp. N-A., Conv.	1	5	0.00	9.00
Thibert and Badami, 2011 (B)	Meal	Meat-based meal, imp., Conv.	2	5	86.7	5.00
Thibert and Badami, 2011 (C)	Meal	Vegan meal, local, Org.	1	5	0.00	9.00
Thibert and Badami, 2011 (C)	Meal	Meat-based meal, local, Org.	2	5	86.7	5.00
Carlsson-Kanyama and González, 2009	Meal	Meal A	1	4	0.00	5.50
Carlsson-Kanyama and González, 2009	Meal	Meal C	2	4	77.0	5.00
Carlsson-Kanyama and González, 2009	Meal	Meal B	3	4	79.0	6.00
Gerbens-Leenes and Nonhebel, 2002	Daily diet	Subsistence diet	1	12	47.6	24.1
Gerbens-Leenes and Nonhebel, 2002	Daily diet	Dutch consumption 1995	2	20	71.7	21.2
Risku-Norja et al., 2009 (A)	Daily diet	Vegan diet - Conv.	1	14	0.00	14.1
Risku-Norja et al., 2009 (A)	Daily diet	Dairy-beef-mutton-free diet - Conv.	2	21	51.3	16.1
Risku-Norja et al., 2009 (A)	Daily diet	Nutr. balanced diet - Conv.	3	24	58.5	21.7
Risku-Norja et al., 2009 (A)	Daily diet	Average Finnish diet - Conv.	4	24	74.2	21.4
Risku-Norja et al., 2009 (B)	Daily diet	Vegan diet - Org.	1	14	0.00	14.1
Risku-Norja et al., 2009 (B)	Daily diet	Dairy-beef-mutton-free diet - Org.	2	21	51.3	16.1

Risku-Norja et al., 2009 (B)	Daily diet	Nutr. balanced diet - Org.	3	24	58.5	21.7
Risku-Norja et al., 2009 (B)	Daily diet	Average Finnish diet - Org.	4	24	74.2	21.4
Collins and Fairchild, 2007	Daily diet	Vegetarian diet	1	35	52.6	18.5
Collins and Fairchild, 2007	Daily diet	Actual diet, high LU items repl.	2	37	68.7	18.5
Collins and Fairchild, 2007	Daily diet	Actual diet, medium LU items repl.	3	30	69.4	18.5
Collins and Fairchild, 2007	Daily diet	Actual diet	4	44	72.1	18.5
Collins and Fairchild, 2007	Daily diet	Actual diet, low LU items repl.	5	28	72.6	18.5
Risku-Norja et al., 2008 (A)	Daily diet	Vegan diet - Conv.	1	14	0.00	11.6
Risku-Norja et al., 2008 (A)	Daily diet	Mixed diet, no poultry/ pork - Conv.	2	19	39.7	19.1
Risku-Norja et al., 2008 (A)	Daily diet	Nutr. balanced diet - Conv.	3	23	58.5	20.2
Risku-Norja et al., 2008 (A)	Daily diet	Average diet - Conv.	4	23	74.6	20.8
Risku-Norja et al., 2008 (B)	Daily diet	Vegan diet - Org.	1	14	0.00	11.6
Risku-Norja et al., 2008 (B)	Daily diet	Mixed diet, no poultry/ pork - Org.	2	19	39.7	19.1
Risku-Norja et al., 2008 (B)	Daily diet	Nutr. balanced diet - Org.	3	23	58.5	20.2
Risku-Norja et al., 2008 (B)	Daily diet	Average diet - Org.	4	23	74.6	20.8
Peters et al., 2006	Daily diet	M00F40	1	44	18.2	14.7
Peters et al., 2006	Daily diet	M02F40	2	48	34.1	15.4
Peters et al., 2006	Daily diet	M04F40	3	48	49.1	16.1
Peters et al., 2006	Daily diet	M06F40	4	49	64.4	16.5
Peters et al., 2006	Daily diet	M08F40	5	49	69.6	16.8
Peters et al., 2006	Daily diet	M10F40	6	48	74.4	16.8
Peters et al., 2006	Daily diet	M12F40	7	48	78.4	16.8
Pathak et al., 2010	Daily diet	Vegetarian	1	10	11.6	10.8
Pathak et al., 2010	Daily diet	Non-vegetarian	2	12	20.2	10.3
Saxe et al., 2012	Daily diet	NND	1	>300	69.8	31.2
Saxe et al., 2012	Daily diet	NNR	2	>300	72.2	25.2
Saxe et al., 2012	Daily diet	ADD	3	>300	75.0	24.9

[a]As provided by included studies or background documentation. Collins and Fairchild (2007) and Saxe (2012) described yearly diets, which we converted to daily diets. Note: Conv = conventional, imp. = imported, N-A = North America, org = organic, nutr = nutritionally, NND = New Nordic Diet, NNR = Nordic Nutritional Recommendations, ADD = Average Danish Diet

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Table A.2 The energy and nutrient intake of daily diet scenarios as computed in this study

Reference	ASFP (%)	Energy (kcal)	Protein (g)	Fiber (g)	Ca (mg)	Fe (mg)	Mg (mg)	K (mg)	Vit. A (µg)	Vit. C (mg)	Vit. E (mg)	Na (mg)	Sat. fat (g)	Total sugar (g)
RDV and MRV		2000	57	25	800	14	375	2000	800	80	12	2400	20	90
Gerbens-Leenes and Nonhebel, 2002	48	2665	132	136	1249	38	1513	8939	753	178	12	911	22	98
Gerbens-Leenes and Nonhebel, 2002	72	3020	142	88	1242	27	1034	5967	1183	163	17	947	39	154
Risku-Norja et al., 2009	0	2447	69	105	504	32	1157	4976	617	227	13	173	11	101
Risku-Norja et al., 2009	51	2491	126	100	517	33	1156	5846	729	233	15	671	12	113
Risku-Norja et al., 2009	59	2468	124	90	1282	29	1090	6393	840	242	13	941	17	152
Risku-Norja et al., 2009	74	2383	147	72	1646	24	971	6071	638	95	12	1085	22	155
Collins and Fairchild, 2007	53	2086	67	24	1267	10	303	2637	1250	99	8	1710	29	137
Collins and Fairchild, 2007	69	2071	88	23	753	11	308	3243	1218	112	9	2126	23	136
Collins and Fairchild, 2007	69	1908	87	22	753	11	290	3118	1199	112	9	2116	19	124
Collins and Fairchild, 2007	72	2119	95	23	882	11	316	3338	1209	113	8	2438	26	137
Collins and Fairchild, 2007	73	1841	99	22	750	10	300	3278	1007	99	9	1268	16	122
Risku-Norja et al., 2008	0	2199	54	90	407	27	939	4184	634	185	12	173	14	77
Risku-Norja et al., 2008	40	2289	94	108	1372	31	1246	5984	737	162	11	765	16	123
Risku-Norja et al., 2008	58	2292	116	83	1255	27	1014	5931	818	227	12	854	16	147
Risku-Norja et al., 2008	75	2284	140	69	1635	23	934	5816	653	103	11	1003	21	157
Peters et al., 2006	18	2278	98	107	1153	30	1132	6141	987	43	35	372	25	64
Peters et al., 2006	34	2195	101	102	1111	29	1101	5791	1145	47	32	443	26	68
Peters et al., 2006	49	2113	104	96	1069	28	1069	5441	1303	52	29	513	26	72
Peters et al., 2006	64	2041	105	86	1012	26	996	4936	1416	55	26	579	26	87

Peters et al., 2006	70	2009	121	84	1022	27	999	5139	1448	55	25	647	27	74
Peters et al., 2006	74	1987	135	79	1020	27	964	5205	1439	53	23	711	27	70
Peters et al., 2006	78	1966	150	74	1016	27	927	5261	1428	51	21	774	27	67
Pathak et al., 2010	12	1702	59	103	572	26	1043	3689	52	42	8	130	10	54
Pathak et al., 2010	20	1739	64	102	465	27	1036	3617	101	41	9	150	12	49
Saxe et al., 2012	71	3095	144	51	1760	19	598	6636	5474	327	18	6319	40	155
Saxe et al., 2012	72	2886	130	45	1332	18	504	5453	2160	151	15	5019	30	130
Saxe et al., 2012	75	2726	119	34	1143	18	439	4504	1599	125	11	4751	32	135

Note: Ca = calcium; Fe = iron, Mg = magnesium, K = potassium, Na = sodium, sat. fat = saturated fat, RDV = daily recommended value, MRV = maximum recommended value

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Chapter 3

Saving land to feed a growing population: consequences for consumption of crop and livestock products

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Abstract

Purpose The expected increase in demand for food raises concerns about the expansion of agricultural land worldwide. To avoid expansion we need to focus on increasing land productivity, reducing waste, and shifting human diets. Studies exploring diet shifts so far have ignored competition for land between humans and animals. Our objective was to study the relation between land use, the share of animal protein in the human diet, population size, and land availability and quality.

Methods We used linear programming to determine minimum land required to feed a population a diet with 0-80% of the protein derived from terrestrial domestic animals. Populations ranged from 15 million to the maximum number of people that could be supported by the system. The agricultural system in the Netherlands was used as illustration, assuming no import and export of feed and food. Daily energy and protein requirements of humans were fulfilled by a diet potentially consisting of grain (wheat), root and tuber crops (potato, sugar beet), oil crops (rapeseed), legumes (brown bean), and animal protein from ruminants (milk and meat) and monogastrics (pork).

Results and discussion We demonstrated that land is used most efficiently if people would derive 12% of dietary protein from animals (% PA), especially from milk. The role of animals in such a diet is to convert co-products from crop production and the human food industry, into protein rich milk and meat. Below 12% PA, human-inedible products were wasted (i.e. not used for food production), whereas above 12% PA, additional crops had to be cultivated to feed livestock. Large populations (40 million or more) could be sustained only if animal protein was consumed. This results from the fact that at high population sizes, land unsuitable for crop production was necessary to meet dietary requirements of the population, and contributed to food production by providing animal protein without competing for land with crops.

Conclusions A land-use optimisation model including crop and animal production enables identification of the optimal % PA in the diet. Land use per capita was lowest at 12% PA, because at this level animals optimally consume co-products from food production. Larger populations, furthermore, can be sustained only with diets relatively high in % PA, as land unsuitable for crop production is needed to fulfil their food demand. The optimal % PA in the human diet depended on population size and the relative share of land unsuitable for crop production.

1. Introduction

Global food demand is projected to increase by 60% by 2050 (Alexandratos and Bruinsma 2012), because of a growing world population and increasing wealth. This increased demand for food has raised concerns about environmental impacts related to expansion of agricultural land worldwide (Foley et al. 2011). Pressure on land increases not only because of future food demands, but also because of land degradation (Stringer 2008) and increasing demands for biofuels (OECD/FAO 2014), biomaterials, housing and infrastructure.

Currently, agriculture already occupies about 38% of the terrestrial surface of the Earth, divided among1.5 billion ha of cropland and 3.4 billion ha of pastures (Alexandratos and Bruinsma 2012). Meeting the food demand projected for 2050 may require an additional 0.2 to 1 billion ha of land under agriculture (Tilman et al. 2011). This additional land will include land of relatively low fertility and productivity and will be partly located in currently forested or protected areas (Alexandratos and Bruinsma 2012; Foley et al. 2011; Ramankutty et al. 2002). Converting such forested lands to agricultural land conflicts with the need for nature preservation (Royal Society of London 2009; Smith et al. 2010; World Bank 2007) and leads to adverse environmental effects (DeFries et al. 2004; Gerber et al. 2013; Millennium Ecosystem Assessment 2005; Pielke Sr et al. 2002).

There is considerable agreement, therefore, that humans should minimise further expansion of agricultural land. Limiting global land expansion for food production, however, requires a combination of interventions on the production and consumption side (Foley et al. 2011). Proposed strategies include increasing yields on underperforming lands (Van Ittersum et al. 2013), reducing waste (Papargyropoulou et al. 2014) and shifting human diets (Stehfest et al. 2009; Wirsenius et al. 2010).

Studies exploring the potential contribution of dietary shifts generally conclude that (i) a vegan diet requires the least land (Hallström et al. 2015) and (ii) that land use decreases when ruminant meat is replaced by monogastric meat (Stehfest et al. 2009; Wirsenius et al. 2010). These studies, however, do not consider the competition between humans and animals for land. Animals fed with cereals, for example, directly compete with humans for land. No matter how efficiently produced, direct consumption of cereals by humans is ecologically more efficient than consumption of animal-source food produced by animals fed with these cereals (Foley et al. 2011; Godfrey et al. 2010). Compared to pigs or poultry, ruminants generally consume less feed that can be consumed directly by humans (De Vries and De Boer 2010; Vellinga et al. 2009). Ruminants, however, can still compete with humans for land, as some of the world's grasslands are also suitable for production of arable crops (Suttie et al. 2005). To limit global land use for food production, therefore, we should consume livestock products from systems that use land that is unsuitable or less suitable for crop production and/or that use co-products from food production (Van Zanten et al. in review). The objective of this study, therefore, was to identify which factors influence the relation between land use, the share of animal protein in the human diet, population size, and land availability and quality. We determined the minimum amount of land used to feed a growing population a diet varying in the percentage of the protein derived from terrestrial domestic animals. The agricultural system in the Netherlands was used as case-study, assuming no import and export of feed and food.

2. Material and methods

This study was based on a land-use optimisation model created in GAMS (General Algebraic Modelling System) version 24.2.

2.1 System definition

The system in our case-study consisted of production, processing and consumption of food in the Netherlands as a standalone system (Fig 1). The objective of this system was to produce human-edible energy and protein for domestic use. The model estimated the land area required to feed populations ranging from 15 million (close to the current population size) to the maximum number of people that could be supported by the system. Within this range, we increased population size by steps of five million people. As we approached the maximum number of people, we increased population size with steps of 0.1 million people. Daily per-capita requirements were defined as 2000 kcal and 57 g protein (EFSA 2009; EFSA 2012). Total sugar intake was limited to the maximum recommended intake level of 90 g per capita per day (EFSA 2009). We estimated land use for human diets varying in the percentage of the protein derived from terrestrial domestic animals ("protein derived from animals" or PA) between 0% PA (a vegan diet) to 80% PA. Within this range, we increased % PA by steps of 5% (and by steps of 1% where relevant). Land use was determined for cultivation of crops and forages.

2.2 Crop production system

The current Dutch agricultural area of 1,842 10^3 ha (CBS 2013) represented the maximum available area for production of crops and forages. This area consists of clay soils (839 10^3 ha), sandy soils (779 10^3 ha) and peat soils (224 10^3 ha) (Lesschen et al. 2012). Clay and sandy soils can be used for cultivation of crops and forages, whereas peat soils were assumed to be suitable only for cultivation of grass, since most of these are too wet for competitive crop production. For the fact that they are marginal for crop production, peat soils represent so-called "marginal" lands in this study.

Crops included in the modelling represented the following major groups: grains, root and tuber crops, oil crops and legumes (Online Resource I). In each food-crop group, the arable crop with the largest cultivated area in the Netherlands was chosen (LEI and CBS 2012). Grains were represented by wheat, root and tuber crops by potatoes and sugar beets, oil crops by rapeseed, and legumes by brown beans. Seven crop rotations were adopted from Van Ittersum et al. (1995), the length of rotations varying from one to six years (Online Resource I). Crop yields were based on current Dutch averages. To compute the annual yield of each crop within a rotation, we multiplied the crop's yield with its frequency in the rotation (i.e. years of cultivation in a rotation divided by total years of rotation). In the harvested crop, we distinguished the main crop product and human-inedible products (i.e. wheat straw, sugar beet tops and tails, and rapeseed straw). Wheat and maize stubble, potato haulms, sugar beet leaves and bean straw were left on the field as source of soil organic carbon. We lowered actual yield levels by 7% for potatoes, 5% for beans and 2% for wheat and rapeseed to account for production of seeds and seedlings (PPO 2009; PPO 2012). Hence, seed and seedling production was already accounted for in crop yields in our calculations ($Y_{i,j,l}$). In addition to crops, we considered production of maize silage and grass as forage for dairy cattle. We assumed no effects on yields of climatic differences across the Netherlands (Van Wart et al. 2013).

To determine total dry matter production (ton DM) of harvested product j (Q_j), we multiplied the land area allocated to crop rotation i on land type l ($X_{i,l}$) by the fresh matter yield of harvested product j from the same crop rotation and land type ($Y_{i,j,l}$), and by the dry matter content of harvested product j (DM_j), then summed across rotations (i=1,7) and land types (l=1,3; Eq.1).

$$Q_j = \sum_{i=1}^{7} \sum_{l=1}^{3} X_{i,l} \times Y_{i,j,l} \times DM_j \qquad \text{Eq.1}$$

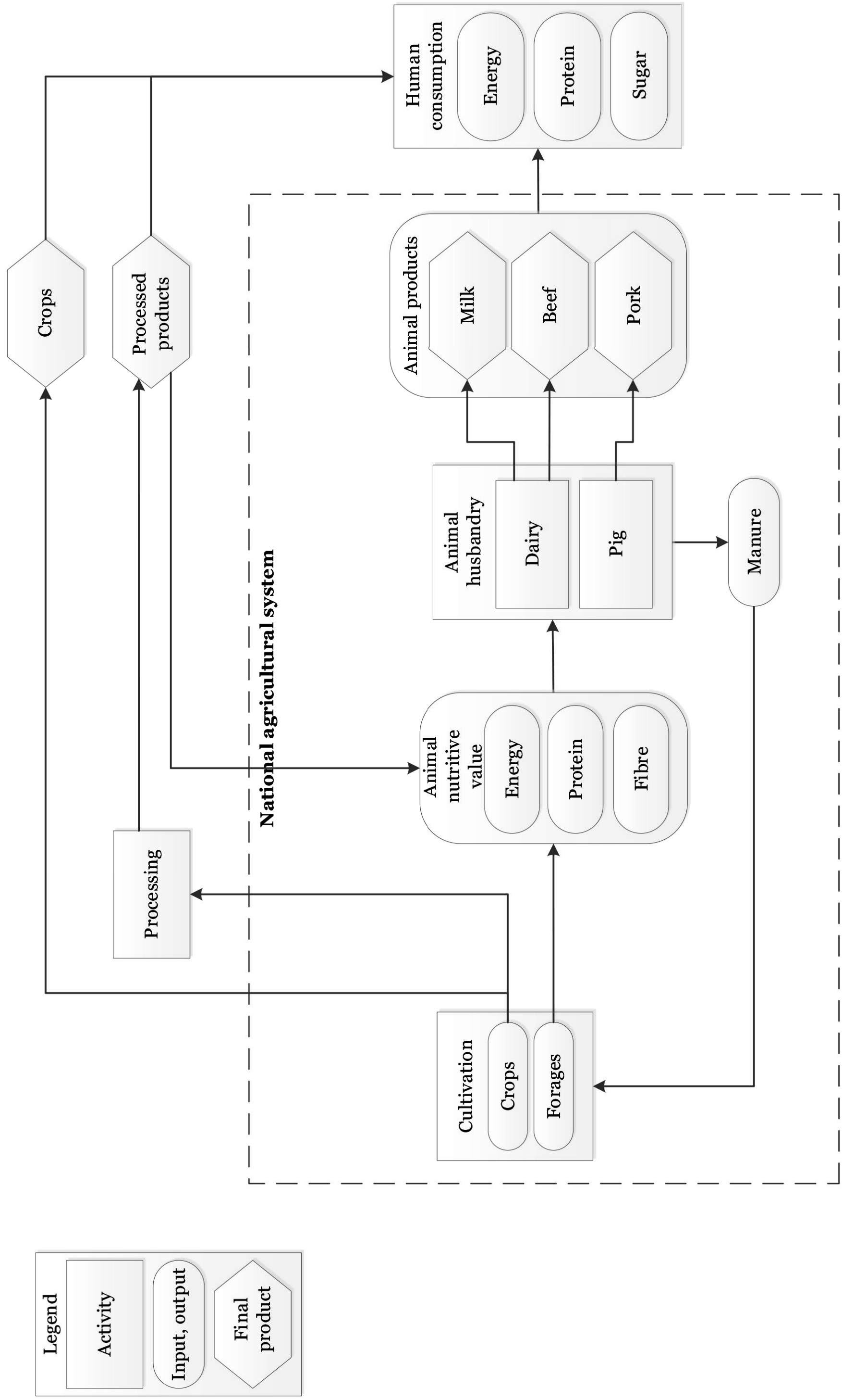

Figure 1 Diagram of the system

2.3 Processing of crops

Harvested products (e.g. rapeseed) were divided into food and feed products (e.g. oil and meal) following ratios of dry matter output and dry matter input of various processing steps (Online Resource II). These dry matter output/input ratios were calculated from fresh matter output/input ratios (Mattsson et al. 2001; Vellinga et al. 2013) and dry matter contents of food (RIVM 2013) and feed products (PDV 2011). To determine dry matter production of food or feed product k (Q_k), we multiplied production quantities of harvested product j assigned to process m ($Q_{j,m}$) with the output/input ratio of product k produced from harvested product j in process m ($C_{j,m,k}$) (Eq. 2). Processes that do not divide one product into multiple products (e.g. ensiling of grass and heating of beans) were assigned an output/input ratio of one.

$$Q_k = Q_{j,m} \times C_{j,m,k} \quad \text{Eq.2}$$

We converted production quantities of human-edible products into available energy, protein and sugar, using nutrient contents of products found in the Dutch nutrient database NEVO (RIVM 2013). Similarly, for animal feeds (see Section 2.4), we converted production quantities into nutrients using feed tables (PDV 2011).

2.4 Animal production system

We included two animal production systems with contrasting abilities to use marginal land. We chose pig production as representative for monogastrics, a system that derives its feeds from land suitable for cultivation of crops, and dairy production as representative for ruminants, able to use marginal land. We chose these systems as within the group of monogastrics and ruminants, pork and dairy products are the largest contributors to protein in the human diet (PPE/PVV 2013; RIVM 2011).

Production levels of animals were based on Dutch averages. We modelled pig and dairy production based on animal production units (PUs) per animal place per year. One pig PU consisted of 3.3 fattening pigs, 0.12 sows and 0.07 gilts (Online Resource III). One pig PU produced 171 kg pork per year, which corresponds to 1475 MJ and 55 kg human-edible protein per year. Net energy requirements per pig PU (equivalent to the weighted sum of net energy requirements (PDV 2012) for fattening pigs, sows and gilts) totalled 9901 MJ per year (Online Resource IV). The diet of one pig PU had a minimum of 16% and a maximum of 18% crude protein (Bikker 2014; Devendra and Clyde Parris 1970) and a digestibility coefficient of at least 80%

(Bikker 2014). Grass, maize silage, straw, sugar beet tops and tails, and sugar factory lime were excluded from consumption by pigs (Bikker 2014). We applied additional restrictions to create a plausible diet (Online Resource V).

One cow PU consisted of a dairy cow and its replacement stock, i.e. 0.31 heifers aged 1-2 years, and 0.34 calves aged 0-1 year (Online Resource VI). Surplus calves were excluded from our analysis. One cow PU produced 8502 kg fat-and-protein-corrected-milk and 74 kg meat per year, both derived only from the milking cow, corresponding to 22775 MJ, 303 kg human-edible protein and 383 kg total sugar per year (Online Resource VI). Net energy requirements for one cow PU (equivalent to the weighted sum of net energy requirements (PDV 2012) for the milking cow, replacement heifer and calf) totalled 51977 MJ and 606 kg intestinal digestible protein per year (Online Resources VII and VIII). Rumen degradable protein balance had a lower limit of 0 and an upper limit of 200 g per cow per day (Dijkstra 2014). To assure sufficient structure in the diet, the structure value of the diet (PDV 2012) was at least 1 per kg DM. Maximum feed intake capacity was limited to 14.9 saturation values per day for dairy cows, 3.2 for replacement heifers, and 5.9 for replacement calves (PDV 2012). Sugar factory lime was excluded from consumption by cows. We applied additional restrictions to create a plausible diet (Online Resource V).

2.5 Manure production and application

Nutrient (i.e. nitrogen and phosphorus) excretion by animals was computed as the difference between nutrient intake and nutrients retained in animals and their products. Nutrient intake was computed based on information about feed intake and nutrient content of feed ingredients (PDV 2011). Nutrient retention in growing pigs was computed from nutrient concentrations in body tissue (Groenestein et al. 2008) and production data (Online Resource III), and totalled 9.4 kg N and 2.0 kg P per pig PU per year. Nutrient retention in milk, and body tissue of culled cows and growing young stock was computed from nutrient concentrations in body tissue and milk (RVO 2010) and production data (Online Resource VI), and totalled 50.1 kg N and 9.64 kg P per cow PU per year.

In line with European Union (EU) legislation, application of manure to crop and grassland was limited to 170 kg nitrogen from animal manure per ha per year (RVO 2014). Additionally, we restricted total nitrogen application from manure and artificial fertiliser to the sum of crop- and soil-type-specific maximum nitrogen application rates allowed by EU legislation. The nitrogen fertiliser replacement value of manure was set at 60% (RVO 2014). Moreover, we restricted total phosphate application to the sum of soil-type-specific maximum phosphate application rates for grass and arable land (RVO 2014). These rates depended on the phosphate levels of the soil, as determined by Schoumans (2007).

2.6 Losses

We accounted for losses of food crop products, meat and milk by applying loss fractions (Gustavsson et al. 2011) during post-harvest handling, storage, processing, packing, distribution and consumption. We assumed that at most 21% of total food crop product losses could potentially be used as feed (Soethoudt and Timmermans 2013). We assumed 5% postharvest handling and storage losses for wheat straw, sugar beet tops and tails and rapeseed straw. In addition, we accounted for conservation and feeding losses for crop products allocated to animals. We assumed conservation losses of 4% for moist concentrates, 5-10% for potato peel, silage maize and beans, 15-17.5% for grass silage and potatoes, 20% for straw, and 25% for sugar beet tops and tails (Remmelink et al. 2012). Feeding losses were 2% for dried concentrates, 3% for moist concentrates, and 5% for roughages (Remmelink et al. 2012). No losses were assigned to fresh grass, as fresh grass yields represented net production (i.e. intake) by animals.

2.7 Objective function

The linear-programming model allocated crop products to humans or animals based on an objective function to minimise land use for all crop rotations *i* on all land types l ($X_{i,j}$) (Eq. 3) whilst meeting energy and protein requirements of the human population.

$$Min \sum_{i=1}^{7} \sum_{l=1}^{3} X_{i,l} \qquad \text{Eq.3}$$

2.8 Sensitivity analysis

We explored the impact of changes in the share of different soil types, in crop and forage yields and in the share of protein from meat in the animal protein consumed on final results of land use and human dietary composition (Table 1)

Changes in share of different soil types. In the reference situation, land consisted of 46% clay soils, 42% sandy soils and 12% peat soils. To determine the impact of decreasing the share of marginal land (peat soil), we studied a situation in which land consisted of 50% clay soils, 45% sandy soils and 5% peat soils (less peat). To determine the impact of increasing the share of marginal land, we studied a situation in which land consisted of 30% clay soils, 30% sandy soils and 40% peat soils (more peat).

Changes in crop and grass yields. To determine the impact of differences in relative productivity of clay, sand and peat soils, we decreased yields on sandy soils by 20% and on peat soils by 50%.
Changes in meat content of the diet. In the reference situation, we did not set requirements for meat consumption. One possible outcome, therefore, was that PA could come mainly from milk. To determine the impact of meat consumption, we forced meat (pork and/or beef) to constitute at least 50% of PA, as this is the current situation in the Netherlands (RIVM 2009).

Table 1. Characteristics of the reference situation and, assessed in sensitivity analyses, alternative situations

Sensitivity	Reference situation	Alternative situations
Soil type	12% peat soils, 46% clay soils, 42% sandy soils	A: 5% peat soils, 50% clay soils, 45% sandy soils
		B: 40% peat soils, 30% clay soils, 30% sandy soils
Yield	Average Dutch yields under current practices	20% lower yields on sandy soils and 50% lower yields on peat soils compared to the reference situation
Meat content	No pre-defined requirements for meat consumption	Meat constitutes at least 50% of dietary protein from animals

3. Results

3.1 Land use

Reference situation

The relation between the minimum amount of land needed to feed a specific population and the percentage of the protein derived from animals (% PA) in the diet was non-linear (Figure 2). As % PA increased, land use initially decreased up to about12% PA, and subsequently increased. Diets with about 12% PA, therefore, systematically had the lowest land use. Furthermore, as population size increased, the possible range of % PA in the diet became more narrow. This implies that larger populations could not be supported by a vegan diet or a diet containing a high % PA.

The amount of land needed per capita increased as population size increased (Table 2). Per-capita land use increased with population size because high yielding soils, i.e. clay soils, were cultivated first, followed by sandy soils (Figure 3). This order follows from the generally higher yields at rotation level on clay soils than on sandy soils. As population size increased, therefore, the relative contribution of lower yielding soils increased, explaining the increase in per-capita land use (Table 2 and Figure 3).

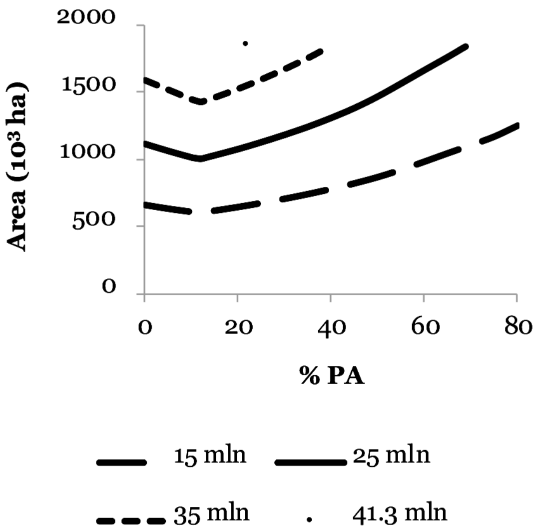

Figure 2. Minimum land (10^3 ha) needed for feeding the total population with diets varying in percentage of dietary protein from animals (% PA) in the reference situation. Mln = million.

Table 2. Per capita land use index for diets varying in percentage of dietary protein from animals (% PA) and various populations in the reference situation. Index = 100 for a diets with 0% PA and a population of 15 million people. Mln = million.

	Population (mln)			
% PA	**15**	**25**	**35**	**40**
0	100	102	104	
10	92	93	94	
15	94	94	95	96
20	98	99	100	100
30	107	108	109	
40	119	120		
50	132	134		
60	150	152		
70	169			
80	191			

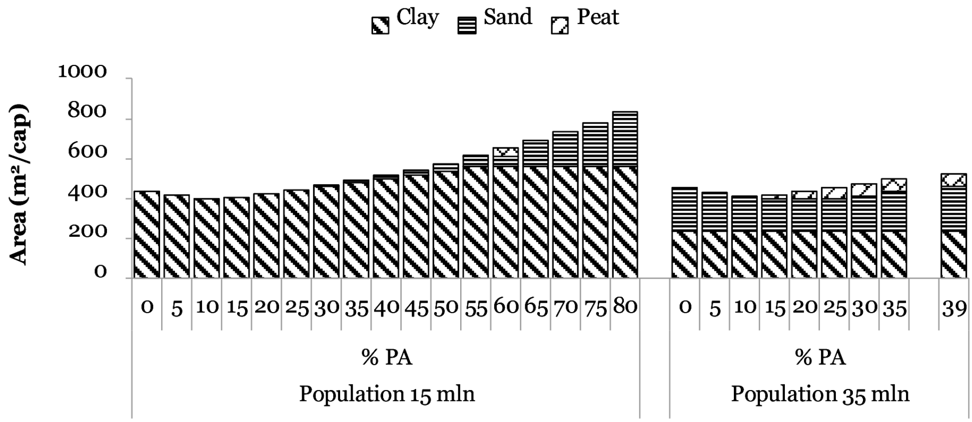

Figure 3. Land use (m²/capita) per soil type for diets varying in percentage of dietary protein from animals (% PA) and populations of 15 million (left) and 35 million (right) people in the reference situation. For a population of 35 million people, 39% PA was the last feasible option. Mln = million.

Food production on clay soils was sufficient to feed a population of 15 million people a diet with 0-25% PA. In the range from 30 – 60 % PA, in addition to clay soils, sandy soils were used for the production of silage maize, as silage maize had higher yields on sandy soils than on clay soils. If PA exceeded 60%, sandy soils were predominantly used for the production of crops, as all clay soils were fully used. Feeding a population of 35 million, however, required all clay soils and most of the sandy soils, even at low % PA. From 15% PA upwards, in addition to clay and sandy soils, peat soils were used to produce grass (see Online Resource X for diet composition per cow PU). Diets with more than 39% PA were not feasible.

Impact of changes in share of different soil types

Decreasing the share of peat soils (i.e. from 12% in the reference situation to 5%) increased the maximum number of people that could be fed from the land, whereas increasing the share of peat soils (i.e. from 12% in the reference situation to 40%) decreased the maximum number of people that could be fed from the land (Figure 4). This difference in the number of people that can be fed can be explained by the higher productivity of clay and sandy soils than peat soils. Furthermore, in the situation with a smaller share of peat soils, the maximum number of people (i.e. 43.6 million) consumed diets with about 15% PA, whereas in the situation with a larger share of peat soils, the maximum number of people (i.e. 31.5 million) consumed diets with about 44% PA. In other words, when population size increases in a region with a larger share of marginal land, this population can be supported only if a relatively high percentage of its protein comes from animal-sources. Moreover, a vegan diet is only feasible for smaller populations in such a situation, i.e. larger populations can only be sustained when animal protein is consumed. The feasible share of animal protein in the human diet, therefore, depends on the population size in combination with the share of marginal land.

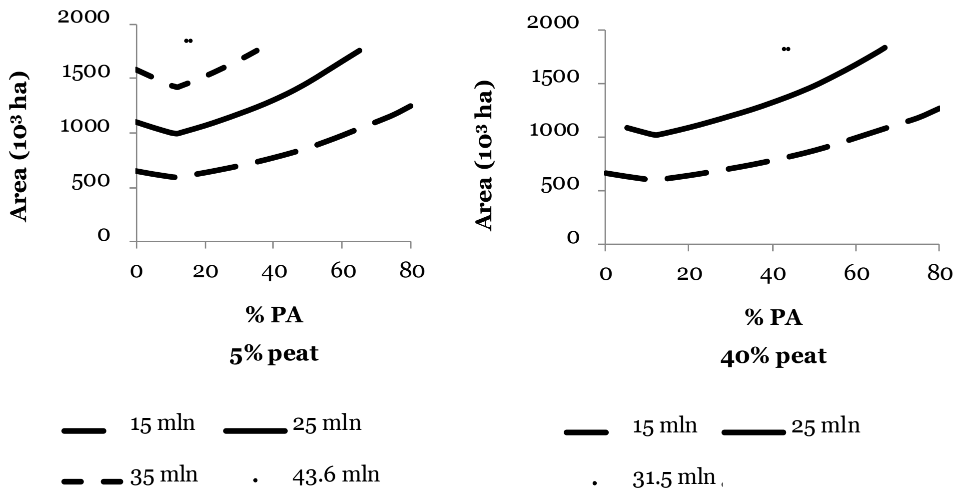

Figure 4. Minimum land (10^3 ha) needed for feeding the total population with diets varying in percentage of dietary protein from animals (% PA) in alternative situations with 5% (left) and 40% (right) of total land area underlain by peat soils. Mln = million.

Impact of changes in crop and grass yields

Decreasing crop yields on sandy and peat soils did not increase per capita land use compared to the reference situation as long as % PA was less than 30%, because only clay soils were used in that range to feed a population of 15 million in both the reference and alternative situation (Figure 5). When % PA was 30% or more, per-capita land use increased relatively quickly compared to the reference situation (Figure 3), because of the lower availability of highly productive land.

We expected decreasing crop yields on sandy soils to result in higher land use on these soils compared to the reference situation, under high population pressure or high % PA. For a population of 15 million, this was indeed the case for diets with 50% PA and more (Figure 5). For diets containing less than 50% PA, however, sandy soils were not used in the alternative situation. This resulted from the relatively small difference in yield of maize silage between clay and sandy soils in the reference situation. After reducing yields by 20% on sandy soils, yield of maize silage was higher on clay soils, which postponed the use of sandy soils to a higher % PA.

To feed 35 million people, more sandy soils were used then in the reference situation, due to their lower yields. From 20% PA upwards, sandy soils were fully used and peat soils were needed to produce animal protein. The maximum feasible % PA for this population was lower than that in the reference situation.

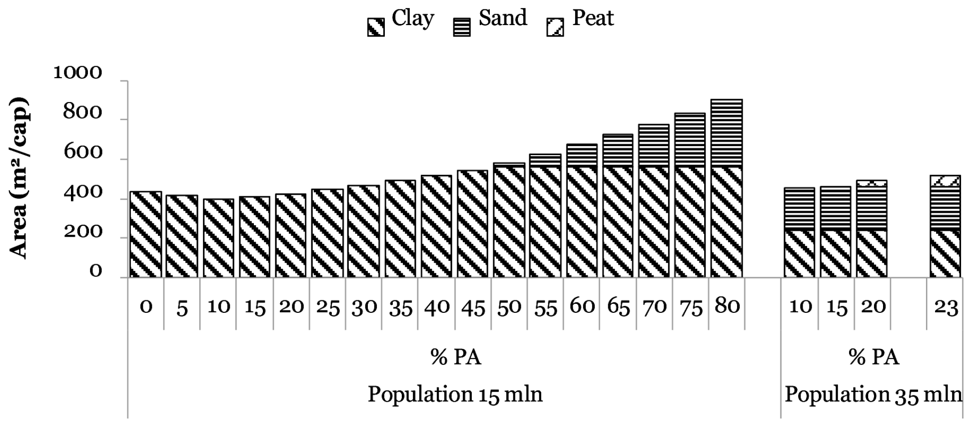

Figure 5. Land use (m²/capita) per soil type for diets varying in percentage of dietary protein from animals (% PA) and populations of 15 million (left) and 35 million (right) people in the alternative situation with 20% lower yields on sandy soils and 50% lower yields on peat soils compared to the reference situation. For a population of 35 million people, 23% PA was the last feasible option. Mln = million.

3.2 Consumption of animal protein

Reference situation

When % PA was less than 10%, daily protein intake per capita equalled the recommended intake level of 57 grams, but this recommended level was often exceeded when % PA exceeded 10% (Figure 6) (see Online Resource IX for human-diet composition). Our simulations also show that animal protein was mainly provided by milk (fixed ratio of protein from milk and beef of 14:1) (Figure 6), which is due to higher protein productivity of dairy cows than of pigs (De Vries and de Boer, 2010).

Impact of changes in meat content of the diet

When requiring that at least 50% of the dietary protein of a population of 15 million came from meat (in our model, beef or pork), the percentage of dietary protein from pork gradually increased from about 2% (i.e. PA = 5%) to about 37% (PA = 80%) (Figure 7), at the expense of dietary protein from milk in particular. Land use in this alternative scenario, therefore, is slightly higher than that in the reference scenario. Hence, replacing dietary protein from milk with that from meat, implies that we can eat less protein derived from animals. For a population of 35 million people, maximum %PA in the diet decreased to 35% (Figure 7), compared to 39% in the reference situation (Figure 6) (see Online Resource XI for diet composition per pig PU).

Figure 6. Per-capita protein intake (g/ day) from crops, milk, beef and pork for diets varying in percentage of dietary protein from animals (% PA) and populations of 15 million (left) and 35 million (right) people in the reference situation. For a population of 35 million people, 39% AP was the last feasible option. The horizontal line indicates the daily protein requirement of 57 g/cap. Mln = million.

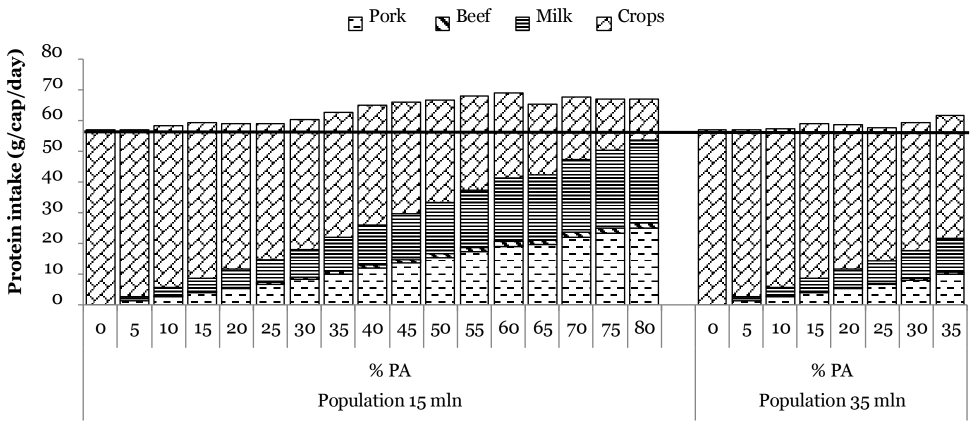

Figure 7. Per-capita protein intake (g/ day) from crops, milk, beef and pork for diets varying in percentage of dietary protein from animals (% PA) and populations of 15 million (left) and 35 million (right) people in the alternative situation in which meat contributed at least 50% of PA. For a population of 35 million people, 35% PA was the last feasible option. The horizontal line indicates the daily protein requirement of 57 g/cap. Mln = million.

4. Discussion and conclusions

Our model provides insights into relations between land use, the share of animal protein in the human diet, population size and land availability and quality. We demonstrated that at lower population sizes consumption of about 12% of dietary protein from animal-source foods resulted in the most efficient use of agricultural land. In the range from 0 to 20% PA, land use remains more or less stable, whereas beyond 20% PA land use increases more strongly. At the highest population size that could be supported by the land, however, the optimal percentage of dietary protein derived from animal-source foods ranged from 15- 45%.

Minimising land use resulted in per-capita land use of 400-800 m^2/ year, values lower than actual land-use values reported by Meier and Christen (2013), Terluin et al. (2013) and Van Oorschot et al. (2012). This implies that humans can use land more efficiently if they would accept austere diets. Diets resulting from our analysis consisted of a limited variety of products because we used proxies for the five major groups of crop production, and for monogastric and ruminant production. We think, however, that including a wider variety of plant-based or terrestrial domestic animal-based products would not have affected our conclusions. Products from fisheries were not considered, as these systems do not use land.

In our results, animal protein in the human diet consisted mainly of milk, and beef was consumed as a co-product of milk production (milk:beef ratio of 14:1). When requiring that at least 50% of the animal protein consumed should come from meat, pork was added to the human diet. Let us consider, however, what would have happened had we included beef production from suckler-beef systems. Suckler cows can exploit marginal lands by producing beef via grazing. Beef from suckler cows, however would have been included in the human diet only if we had defined minimum requirements for beef consumption, or if marginal lands had been suitable only for grazing of suckler cows. This is because from a land-use perspective grazing of dairy cows is preferred to grazing of suckler cows because dairy cows produce animal protein more efficiently (De Vries and De Boer 2010). Furthermore, feed produced on clay or sandy soil is converted more efficiently to animal protein by pigs than by beef cattle. We realize, however, that producing beef or mutton on marginal lands unsuitable for grazing of dairy cattle can be of utmost importance in other areas, and this will result in an increase of per-capita land use.

Another important finding of our study is that a vegan diet always required more land than a diet with small amounts of animal protein. In other words, land is used most efficiently if people consume small amounts of animal protein, which is also referred to as the "default livestock diet" (Fairlie 2010). The role of animals in a default livestock diet is to convert co-products from arable production (e.g. straw) and the human food industry (e.g. beet pulp) that cannot be consumed directly by humans into protein-rich milk and meat. When no animal protein is produced, as suits a vegan diet, these human-inedible products are wasted (i.e. not used for food production) or used as a bio-energy source, and additional crops will have to be cultivated to meet nutritional energy and protein requirements of the population. Consequently, larger populations could not be supported by a vegan diet and a population cannot exceed a certain size unless animal protein is consumed.

Larger populations also could not be supported by a diet with a high percentage of protein derived from animal-source foods. A population of 35 million people, for example, could not be supported from a diet containing 40% PA or more. When demand for animal protein exceeds the default livestock diet, additional crops will have to be cultivated, resulting in higher land use. At higher population sizes this land is not available, which limits consumption of high amounts of animal protein, and, thus, the possible percentages of dietary protein derived from animal-source foods decreased. Moreover, at the highest population size that could be supported by the land, the optimal percentage of dietary protein derived from animal-source food ranged from 15-45%, the exact value depending on assumed crop yields and the share of marginal land. Peat soils, although relatively productive in the Netherlands, were considered marginal land as they are suitable only for grass production. Also, peat soils are somewhat less productive than clay soils in The Netherlands. Thus, increasing the share of peat soils increased the optimal percentage of dietary protein from animal-source foods at the highest population size, but at the same time decreased the maximum number of people that could be fed. In contrast, decreasing the relative share of peat soils, and hence increasing the relative share of arable soils, would increase the number of people that could be fed and lower the optimal percentage of dietary protein from animal-source foods at higher population sizes. The optimal percentage of dietary protein from animals in future diets, therefore, depends on the share of marginal land in the world, together with the productivity of these marginal lands (which is atypically high in The Netherlands). Moreover, the optimal percentage of dietary protein from animals also depends on the type of crops and the extent to which co-products are harvested. A higher availability of co-products for feed would shift optimum land use to higher % PA but also reduce carbon inputs to the soil.

A final important conclusion is that our results contradict results of life cycle assessment (LCA) studies that explored land use of diets differing in the percentage of protein derived from animals. These LCA studies suggest that vegan diets require the least amount of land, followed by vegetarian diets (Hallström et al. 2015; Meier and Christen 2013). Optimisation, as employed in our study, accounts for the unsuitability of marginal lands to grow crops, the suitability of animals to use human-inedible products, and the co-production of meat and milk. These aspects are not included in LCA studies, and explain the different results. Our land-use optimisation model could be extended to the use of other limited resources such as fossil energy and phosphorous, and the emission of, for example, greenhouse gases.

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Online resource I Crop yields per hectare per crop rotation by soil type, and dry matter fraction of harvested crops

	G	M	WOWB	PWSW	PBSW	WOWBS	WOWBWP
ton FM/ha				Clay			
Wheat grain			4.31	4.31	2.16	3.45	4.31
Wheat straw			2.19	2.19	1.10	1.75	2.19
Potato tubers				11.8	11.8		7.88
Sugar beet				17.8	17.8	14.2	
Sugar beet tops and tail				1.07	1.07	0.85	
Rapeseed			0.97			0.77	0.65
Rapeseed straw			0.64			0.51	0.42
Beans			0.76		0.76	0.61	0.51
ton DM/ha							
Silage maize		16.1					
Silage grass	8.14						
Fresh grass	2.95						

Note: G=grass, M=maize, W=wheat, O=oilseed, B=beans, P=potato, S=sugar beet. Unless mentioned otherwise, yields of harvested crops were averages from PPO (2009) and PPO (2012). The yield of sugar beet tops and tails was computed as 6% of the yield of sugar beet (Corré and Langeveld 2008; Van Zeist. et al. 2012). The yield of rapeseed on sandy soils was taken from (Van Geel and Borm 2007). The production of rapeseed straw on sandy soils was computed based on the ratio between seeds and straw on clay. The yield of beans on sandy soils was computed using its yield on clay soil and the yield ratio between field beans grown on clay and sandy soils (WRR 1992). Yield of grass and silage maize were taken from Hooijboer et al. (2014; 2013) and Buis et al. (2012). Dry matter percentages were taken from PPO (2009) (for wheat grain), from Vellinga et al.(2013) (for wheat straw, and beans), from Productboard for arable production (2013) (for potato tubers), and from Marinussen et al. (2012) (for rapeseed). Dry matter percentage of sugar beet was based on Corré and Langeveld (2008). For rapeseed straw we assumed the same dry matter percentage as for wheat straw.

Continued - Online resource I Crop yields per hectare per crop rotation by soil type, and dry matter fraction of harvested crops

	G	M	WOWB	PWSW	PBSW	WOWBS	WOWBWP	G	
ton FM/ha					Sand			Peat	DM fraction
Wheat grain			3.70	3.70	1.85	2.96	3.70		0.84
Wheat straw			1.89	1.89	0.94	1.51	1.89		0.90
Potato tubers				12.32	12.32		8.22		0.21
Sugar beet				16.61	16.61	13.29			0.24
Sugar beet tops and tail				1.00	1.00	0.80			0.24
Rapeseed			1.06			0.84	0.70		0.86
Rapeseed straw			0.70			0.56	0.46		0.90
Beans			0.75		0.75	0.60	0.50		0.86
ton DM/ha									
Silage maize		16.37							
Silage grass	7.74							7.72	
Fresh grass	2.59							2.94	

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3

Online resource II Output/input ratio (DM weight basis) of various processes of crops

Process	Input	Output	Output/input ratio
Sugar beet processing	Sugar beet	Sugar	0.61
		Sugar factory lime	0.11
		Sugar beet molasses	0.10
		Sugar beet pulp	0.18
Dry milling of wheat	Wheat grain		
		Wheat middlings	0.12
		Wheat germ	0.02
		Wheat bran	0.12
		Wheat flour	0.73
Crushing of rapeseed	Oilseed	Rapeseed oil	0.44
		Rapeseed meal	0.56
Peeling of potato	Potato	Potato tuber	0.94
		Potato peel	0.06

Sources: adapted from Vellinga et al. (2013) and Mattsson et al. (2001)

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Online resource III Herd and meat production data per Pig production unit (PU) per year

	Value	Unit	Reference
Fattening pig			
Length of fattening pig period	112	days	PDV (2012)
Number of slaughtered fattening pigs per PU per year	3.3		
Death rate piglets till weaning	0.13	fraction	Agrovision (2013)
Death rate piglets after weaning	0.02	fraction	Agrovision (2013)
Death rate fattening pig period	0.02	fraction	Vermeij (2012)
Average live weight piglet at birth/death	2.1	kg	Groenestein et al. (2008)
Live weight piglet at end phase	25	kg	PDV (2012)
Live weight at slaughter fattening pig	113	kg	PDV (2012)
Slaughter weight fattening pig	88	kg/animal	PDV (2012); Agrovision (2013)
Meat fraction fattening pig	0.58	fraction	Agrovision (2013)
Meat weight fattening pig	51	kg/animal	
Meat production per fattening pig PU per year	167	kg	
Sow			
Replacement rate sow	0.42	fraction	Vermeij (2012)
Dry period (interval suckling-heat)	21	days	Vermeij (2010)
Length of suckling period	25	days	Vermeij (2010)
Length of gestation period	115	days	Vermeij (2010)
Number of gestations per sow per year	2.27		
Litter size (alive and dead) first litter	14		PDV (2012)
Litter size (alive and dead) 2nd and later litter	15		PDV (2012)
Average litter size	15		
Number of sows per year per slaughtered fattening pig	0.04		
Selection rate sows (= selection for slaughter)	0.37	fraction	Vermeij (2012)
Meat fraction sow	0.50	fraction	Bikker (2014)
Live weight at start phase	130	kg	Jongbloed (2010), Bikker (2014)
Live weight at slaughter	214	kg	
slaughter weight per sow	167	kg	Vermeij (2012)
Meat weight per sow	84	kg	
Meat production from sow per fattening pig PU per year	3.61	kg	

	Value	Unit	Reference
Gilt			
Fraction gilts not for replacement	0.27	fraction	Vermeij (2012)
Number of gilts per year per slaughtered fattening pig	0.02		
Selection rate gilts (= selection for slaughter)	0.25	fraction	Vermeij (2012)
Live weight at start phase	25	kg	Jongbloed (2010), Bikker (2014)
Live weight at slaughter	113	kg	
Slaughter weight per gilt	88	kg	Bikker (2014)
Meat fraction gilts	0.58	fraction	Agrovision (2013)
Meat weight per gilt	51	kg	
Meat production from gilt per fattening pig PU per year	0.86	kg	

Note: The energy and protein contents of pork were 8.6 MJ and 0.32 kg protein per kg of fresh pork (Van Kernebeek et al. 2014). Pork did not contain any sugar.

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Online resource IV Net Energy (NE) requirements per animal per period

		Total NE (MJ)	# days in period	NE/day
Piglet		286		
Fattening pig		2345		
Day within period	1		7	10.12
	8		7	11.66
	15		7	13.20
	22		7	14.74
	29		7	16.28
	36		7	17.82
	43		7	19.36
	50		7	20.90
	57		7	22.44
	64		7	23.98
	71		7	25.30
	78		7	26.49
	85		7	27.68
	92		7	28.29
	99		7	28.38
	106		7	28.38
Dry sow		409		
			6	30.80
			15	14.96
Gestating sow		2580		
Days of gestation	0-14		14	17.42
	15-28		14	18.83
	29-56		28	20.86
	57-84		28	22.62
	85-98		14	24.64
	99-115		17	27.02
Lactating sows		1540	25	61.60
Gilts		1977		
Age (weeks)	11		42	8.80
	17		42	14.08
	23		42	20.24
	29		7	23.76

Source: (PDV 2012). Energy requirements per day for dry sows were based on the assumption that dry sows are fed 3.5 EW (energy value for pigs; EW = NE/8.8) per day during the first 6 days of the dry period, and are fed comparable to gestating sows during the remainder of days within the dry period (Hoste 2013).

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Online resource V Additional ration restrictions for cows and pigs

	CowPU	PigPU	Source
Max. feed intake (ton DM) per aniPU/year			
Potato peel	0.45		De Jong (1987)
Sugar beet tops&tails	1.2		De Jong (1987)
Sugar beet pulp	1.4		De Jong (1987)
Max. fraction of fresh matter			
Sugar	0.05		De Jong (1987)
Sugar beet molasses	0.08		Subnel (1997)
Rapeseed meal	0.25		Subnel (1997)
Beans	0.15		Subnel (1997)
Wheat-combination	0.4		Subnel (1997)
Max. fraction of dry matter			
Sugar		0.15	Bikker (2014)
Sugar beet pulp		0.075	OPVN (2014)
Sugar beet molasses		0.06	Bikker (2014)
Wheat middlings		0.2	Feedipedia (2014)
Wheat germ		0.2	Feedipedia (2014)
Wheat bran		0.2	Feedipedia (2014)
Wheat flour		0.2	Feedipedia (2014)
Potato peel		0.1	OPVN (2014)
Rapeseed oil		0.04	Bikker (2014)
Rapeseed meal		0.15	Bikker (2014)
Beans		0.2	Feedipedia (2014)

Note: PU = production unit. One cow PU consisted of a dairy cow and its replacement stock, i.e. 0.31 heifers aged 1-2 years, and 0.34 calves aged 0-1 year . One PigPU consisted of 3.3 fattening pigs, 0.12 sows and 0.07 gilts.

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Online resource VI Herd and milk and meat production data per cow production unit (PU) per year

	Value	Unit	Reference
Dairy cow			
Live weight	650	kg	PDV (2012)
Replacement rate	0.3	fraction	Vermeij (2012)
Death rate	0.02	fraction	Vermeij (2012)
# Slaughtered dairy cows/dairy cow PU/ year	0.28		
Calving interval	419	days	Vermeij (2012)
Fat in milk	4.4	%	Vermeij (2012)
Protein in milk	3.5	%	Vermeij (2012)
Meat weight/dairy cow	264	kg	Van Middelaar (2014)
Meat production/dairy cow PU/year	74	kg	
Average milk yield/dairy cow/year	8120	kg	LEI (2013)
Milk consumption by replacement calve/ dairy cow PU/year	79.2	kg	
Milk production/dairy cow PU/year	8502	kg FPCM	
Replacement heifers 1-2 yr			
Death rate replacement heifer age 1-2 years	0.02	fraction	Vermeij (2012)
Number of calves/cow PU/year	0.31		
Replacement calves 0-1 yr			
Death rate at birth	0.07	fraction	Vermeij (2012)
Death rate within 2 months	0.03	fraction	Vermeij (2012)
Death rate between 2 months and 1 year	0.02	fraction	Vermeij (2012)
Number of calves/cow PU/year	0.34		

Note: Milk production (kg) was a three-year average over the years 2009-2011 (LEI 2013). We subtracted milk consumption by calves (231 kg during their first two months) (Remmelink et al. 2012) from milk production. Milk production (kg) was converted to Fat and Protein Corrected Milk (FPCM) according to PDV (2012). Energy and protein contents of whole milk were 2.58 MJ and 0.03 kg protein per kg of milk (RIVM 2013). Energy and protein contents of beef were 11.3 MJ and 0.3 kg protein per kg of beef (Van Kernebeek et al. 2014). Beef contained 0.002 kg sugar per kg (Van Kernebeek et al. 2014).

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Table VII Energy and protein requirements for maintenance of dairy cows

		First calver		Second calver		Third calver	
		VEM/ day	DVE/ day	VEM/ day	DVE/ day	VEM/ day	DVE/ day
Milk production							
Maintenance and grazing		5855	131	5855	131	5855	131
Age premium		660	37	330	19		
Gestation premium (month of gestation)							
	6th	450	60	450	60	450	60
	7th	850	105	850	105	850	105
	8th	1500	180	1500	180	1500	180
	9th	2700	280	2700	280	2700	280
Total per year		2464485	74541	2373900	69600	2283315	64384
Average per day		6752	204	6504	191	6256	176

Source: (PDV 2012). VEM = feed unit milk production. DVE = intestinal digestible protein

References

PDV (2012) Tabellenboek Veevoeding 2012. Product Board Animal Feed, The Hague, The Netherlands

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Online resource VIII Energy and protein requirements for maintenance of replacement calves and replacement heifers

	Replacement calves 0-1 yr		Replacement heifers 1-2 yr	
	VEM/day	DVE/day	VEM/day	DVE/day
Maintenance and growth				
2-3 months	2500	225		
4-5 months	3200	255		
6-7 months	3850	285		
8-9 months	4600	305		
10-11 months	4850	290		
12 months	5400	310		
13 months			5400	310
14-15 months			5900	330
16-17 months			6100	335
18-19 months			6650	355
20-21 months			7300	415
22 months			7500	460
23 months			7500	460
24 months			7500	460
Grazing premium				
2-3 months	125			
4-5 months	175			
6-7 months	225			
8-9 months	275			
10-11 months	300			
12 months	325			
13 months			325	
14-15 months			375	
16-17 months			400	
18-19 months			425	
20-21 months			475	
22 months			525	
23 months			550	
24 months			575	
Total per year	1404463	92753	2588263	138620
Average per day	3848	254	7091	380

Source: (PDV 2012). VEM = feed unit milk production. DVE = intestinal digestible protein

References

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Online resource IX Diet composition (grams per capita per day) for a population of 15 million people, for diets varying from 0 to 80% PA in the reference situation

% PA	**0**	**5**	**10**	**15**	**20**	**25**	**30**	**35**	**40**	**45**	**50**	**55**	**60**	**65**	**70**	**75**	**80**
gr/cap/day																	
Wheat germ	6.08	5.81	5.59	5.46	0	0	0	0	0	0	0	0	0	0	0	0	0
Wheat bran	39.3	37.5	36.1	35.3	35.1	24.4	11.7	22.8	0	0	0	0	0	0	0	0	0
Wheat flour	239	228	220	215	213	213	222	208	212	202	191	169	144	68.0	45.0	0	0
Potato	591	581	569	564	576	591	614	601	618	613	608	627	653	823	842	913	824
Sugar	62.5	71.5	76.0	72.2	68.4	64.5	61.8	56.4	52.5	46.5	39.9	34.0	27.8	29.9	23.1	21.8	8.49
Rapeseed oil	17.3	14.9	13.3	12.2	10.6	8.96	9.54	6.28	5.80	3.08	0.06	0	0	15.2	14.3	23.1	16.1
Beans	87.1	75.0	67.0	61.3	53.1	45.2	0	0	0	0	0	0	0	0	0	0	0
Pork	0	0	0	0	0	0	0	0.55	0.51	0.27	0.01	0	0	0	0	0	0
Beef	0	0.62	1.26	1.94	2.63	3.34	3.90	4.85	5.55	6.61	7.78	8.85	9.98	9.80	11.0	11.4	13.7
Milk	0	80.7	163	252	341	433	506	630	721	858	1010	1149	1296	1272	1431	1474	1774

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Online resource X Diet composition per cow production unit (PU) (gram DM per day) when population is 15 million and human diets vary from 0-80% PA in the reference situation

% PA	**0**	**5**	**10**	**15**	**20**	**25**	**30**	**35**	**40**	**45**	**50**	**55**	**60**	**65**	**70**	**75**	**80**
gr DM/cow PU/day																	
Wheat middlings	0	0	5709	3607	2653	2085	1863	1380	1229	992	804	727	672	972	862	967	702
Wheat germ	0	0	62.0	39.0	427	336	300	202	182	153	129	117	108	157	139	156	113
Wheat bran	0	0	220	245	180	745	1326	515	1247	1007	816	739	683	987	876	982	713
Wheat flour	0	4815	2290	1447	1065	837	747	563	500	401	322	870	1402	4561	4425	5798	4207
Wheat straw	0	6867	6682	4491	3179	2988	3316	2257	2116	1616	1218	0	145	3467	3076	3450	2504
Potato	0	0	0	912	689	556	495	372	335	285	245	222	206	264	240	253	190
Potato peel	0	1230	1159	743	561	453	403	317	285	237	200	181	167	215	195	206	154
Sugar	0	1767	904	1547	2299	2755	2659	2890	2907	3012	3096	2969	2896	2268	2326	1824	2380
Sugar beet molasses	0	0	0	1382	1158	1023	902	810	746	687	640	581	537	447	427	343	390
Sugar beet pulp	0	2972	3712	2530	2120	1872	1650	1483	1365	1257	1171	1063	983	818	781	629	714
Sugar beet tops&tails	0	0	0	749	628	554	489	439	404	372	347	315	291	242	231	186	211
Rapeseed oil	0	88.0	39.0	23.0	15.0	10.0	9.00	0	0	0	0	0	0	6.00	5.00	7.00	4.00
Rapeseed meal	0	5129	2291	1356	869	581	530	272	219	98.0	2.00	0	0	336	280	441	255
Rapeseed straw	0	0	0	0	830	555	506	268	216	96.0	2.00	0	0	321	268	421	244
Beans	0	137	61.0	36.0	23.0	15.0	650	344	277	124	2.00	0	0	393	321	514	285
Silage maize	0	0	0	0	0	0	563	1517	2191	2777	3242	3631	3462	687	1034	1142	1704
Silage grass	0	0	0	2397	4038	4964	4378	5973	5846	6523	7062	6727	6543	4043	4383	3025	4733
Fresh grass	0	0	0	1075	1812	2227	1964	2680	2623	2927	3168	3021	2961	1814	1966	1357	2124

Note: One cow PU consisted of a dairy cow and its replacement stock, i.e. 0.29 heifers aged 1-2 years, and 0.32 calves aged 0-1 year .

Online resource XI Diet composition per pig production unit (PU) (gram DM per day) when population is 15 million and human diets vary from 0-80% PA in the scenario where meat is required

% PA	**0**	**5**	**10**	**15**	**20**	**25**	**30**	**35**	**40**	**45**	**50**	**55**	**60**	**65**	**70**	**75**	**80**
gr DM/pig PU/day																	
Wheat middlings	0	504	506	497	495	514	532	530	495	474	461	438	424	478	445	436	423
Wheat germ	0	45.0	21.0	209	162	133	110	93.0	80.0	76.0	74.0	71.0	68.0	77.0	72.0	70.0	68.0
Wheat bran	0	63	135	0	0	186	374	450	503	482	469	445	430	486	452	443	429
Wheat flour	0	504	506	497	405	332	304	530	541	539	536	536	535	541	536	535	534
Potato	0	378	380	329	260	217	183	156	134	127	123	117	113	119	113	88.0	66.0
Potato peel	0	0	0	40.0	0	0	0	0	0	0	0	0	0	0	0	0	0
Sugar	0	308	249	373	372	385	399	377	312	334	370	370	385	403	402	401	400
Sugar beet molasses	0	151	152	68	149	154	160	0	162	162	113	161	161	130	130	129	131
Sugar beet pulp	0	189	190	186	186	193	200	199	203	202	201	201	201	203	201	201	200
Rapeseed oil	0	0	13.0	8.00	6.00	4.00	3.00	0	2.00	2.00	3.00	3.00	3.00	4.00	4.00	5.00	6.00
Rapeseed meal	0	378	380	277	143	137	178	142	122	134	150	155	164	250	200	154	107
Beans	0	0	0	0	300	312	219	175	150	163	179	184	192	13.0	128	213	303

One pig PU consisted of 3.3 fattening pigs, 0.12 sows and 0.07 gilts

Chapter 4

Closing the phosphorus cycle in a food system: insights from a modelling exercise

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Abstract

Mineral phosphorus (P) used to fertilise crops is derived from phosphate rock, which is a finite resource. Preventing and recycling mineral P waste in the food system, therefore, are essential to sustain future food security and long-term availability of mineral P. The aim of our modelling exercise was to assess the potential of preventing and recycling P waste in a food system, in order to reduce the dependency on phosphate rock. To this end, we modelled a hypothetical food system designed to produce sufficient food for a fixed population with a minimum input requirement of mineral P. This model included representative crop and animal production systems, and was parameterised using data from the Netherlands. We assumed no import or export of feed and food. We furthermore assumed small P soil losses and no net P accumulation in soils, which is typical for northwest European conditions. We first assessed the minimum P requirement in a baseline situation, i.e. 42% of crop waste is recycled, and humans derived 60% of their dietary protein from animals (PA). Results showed that about 60% of the P waste in this food system resulted from wasting P in human excreta. We subsequently evaluated P input for alternative situations to assess the (combined) effect of: 1) preventing waste of crop and animal products, 2) fully recycling waste of crop products, 3) fully recycling waste of animal products, and 4) fully recycling human excreta and industrial processing water. Recycling of human excreta showed most potential to reduce P waste from the food system, followed by prevention and finally recycling of agricultural waste. Fully recycling P could reduce mineral P input by 90%. Finally, for each situation, we studied the impact of consumption of PA in the human diet from 0 to 80%. The optimal amount of animal protein in the diet depended on whether P waste from animal products was prevented or fully recycled: if it was, then a small amount of animal protein in the human diet resulted in the most sustainable use of P; but if it wasn't, then the most sustainable use of P would result from a complete absence of animal protein in the human diet. Our results apply to our hypothetical situation. The principles included in our model however, also hold for food systems with, for example, different climatic and soil conditions, farming practices, representative types of crops and animals, and population densities.

1. Introduction

Sustainable food security has become a prominent research topic (West et al., 2014). The urge to produce safe and nutritious food in a sustainable way is mainly driven by two challenges: feeding a growing and more prosperous world population, and reducing the environmental impact of food production. The current food production system largely depends on supplies of mineral phosphorus (P). Mineral P is derived from rock phosphate, which is a finite resource. Mineral P is an essential nutrient for crop and grass growth, and, hence, is essential for food security (Smil, 2000). However, use of P in the global food system is rather inefficient, and sustainable food security requires a more sustainable use of mineral P (Cordell and White, 2015). This can be achieved by preventing waste of crop and animal products, disposal of industrial processing water and human excreta, and leaching and run-off from agricultural land (Cordell et al., 2009, Smit et al., 2015), and also by changing human consumption patterns towards diets that contain less animal-source food (Schmid Neset et al., 2008, Bai et al., 2016).

The aim of our modelling exercise was to assess the potential of preventing and recycling P waste in a food system, in order to reduce the dependency on phosphate rock. To this end, we modelled a hypothetical food system designed to produce sufficient food for a fixed population with a minimum requirement of mineral P input. This model included representative crop and animal production systems, and was parameterised using data from the Netherlands. We assumed no import or export of feed and food. We furthermore assumed small P soil losses from run-off and leaching and no net P accumulation in soils, which is typical for northwest European conditions nowadays (Sattari et al., 2012). We also explored the effect of this assumption. We assessed mineral P input for a baseline situation and six alternative situations. These alternative situations were designed to assess the effect of 1) preventing waste of crop and animal products, 2) fully recycling waste of crop products, 3) fully recycling waste of animal products, 4) fully recycling human excreta and industrial processing water, 5) a combination of prevention and recycling as applied in alternative situations 1 and 4, and 6) a combination of recycling as applied in alternative situations 2, 3 and 4. Within each situation, we moreover studied the impact of consumption of protein from animals (PA) on P input requirement of the food system.

2. Material and methods

We compared mineral P input requirements between a baseline situation and six alternative situations. Within each situation, we also studied the impact of consumption of ASF (meat and milk), by varying PA from 0 to 80%. To quantify the use of mineral P in each situation, we extended the optimisation model developed by (Van Kernebeek et al., 2016). This extended model had the objective function to minimise mineral P input from feed additive for all animal types *k*, and mineral fertiliser for all crop rotations *i* on all land types *l* (Eq. 1), while producing sufficient food for a fixed population. The crop and animal production system were parameterised using data from the Netherlands. The related constraints of the model are land availability, crop rotation (*i*=1...7), land type (*j*=1...3), and type of livestock (*k*=1,2) (see sections on crop production system and animal production system for further details).

$$Min(\sum_{k=1}^{2} P_{Feed\ additive,k} + \sum_{i=1}^{7}\sum_{j=1}^{3} P_{Mineral\ fertiliser,i,j}) \qquad \text{Eq. 1}$$

We first describe key features of this extended model, and then define the baseline situation and alternative situations. Finally, we describe the extended model in more detail.

2.1 System definition

Our hypothetical food system comprises the following processes: crop cultivation, post-harvest crop storage, feed processing, food processing, animal husbandry, processing of animals and their products (slaughtering, pasteurisation of milk etc.), manure storage, human consumption, and waste water treatment (Figure 1). The purpose of the system was to produce enough nutritional energy and protein to feed a population of 17 million people, which is approximately the current population size in the Netherlands. Daily per capita nutritional requirements were defined as 2 000 kcal and 57 g protein (EFSA, 2009 and 2012). Total sugar intake was limited to the maximum recommended intake level of 32.9 kg per capita per year (EFSA, 2009). Crop products available for human consumption (Supplementary Material S1) could be consumed without any further restrictions. In addition to crop products, humans could consume milk and beef from dairy cows, and pork from pigs. The selection of crop and animal products resulted in a hypothetical and sober diet. We computed energy, protein, sugar and P intake based on nutrient contents presented by the Dutch nutrient database NEVO (RIVM, 2013). We assumed that all P consumed was excreted. Waste of P (Efflux; (E) in Figure 1) occurred through losses and waste of crop and animal products (including animal meal), human excreta, waste water from industrial processing of crops, and through leaching and run-off of P from cropland. To simplify our writing, we refer to waste for all above-mentioned losses and waste, except for P loss through leaching and run-off, as these were fixed, and not considered for prevention or recycling.

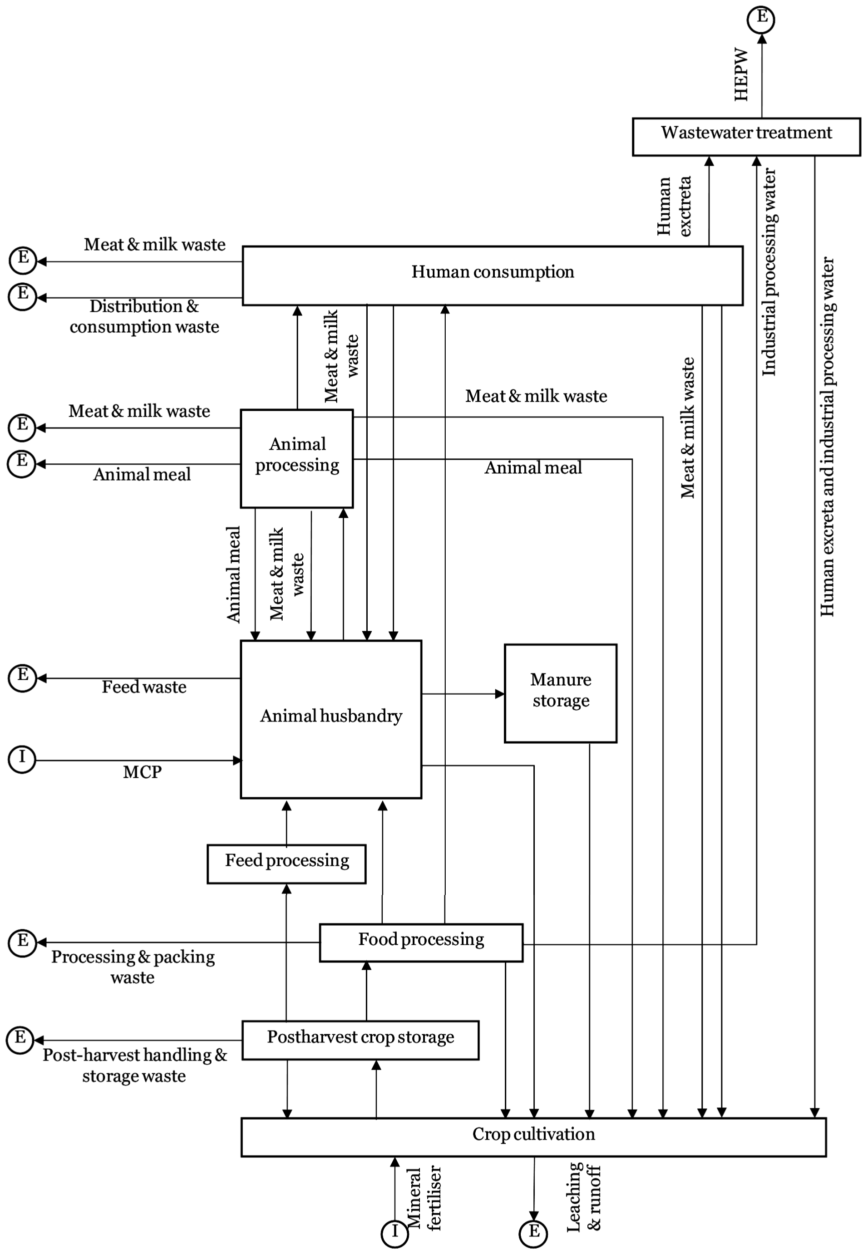

Figure 1. Diagram of the system. Note: I = influx, E= efflux, MCP = monocalcium phosphate, HEPW = human excreta and industrial processing water. P flows are incorporated in crops or crop products, unless specified otherwise.

Depending on the situation (i.e. baseline or alternative situation), we allowed for recycling of wasted P. When recycled, we assumed that waste of crop and animal products could be converted into animal feed or crop fertiliser, and that human excreta and industrial processing water could be converted into crop fertiliser only (Figure 1). Inputs of mineral P in the system included monocalcium phosphate (MCP) as feed additive, and mineral fertiliser P.

2.2 Prevention and recycling of wasted phosphorus to close the phosphorus cycle

In the baseline situation, we assumed that at most 42% of waste of crop products could potentially be recycled (Soethoudt and Timmermans (2013) and Supplementary Material S2). Animal wastes, including slaughterhouse wastes (i.e. animal meal), were not recycled, as this is restricted by EU legislation (European Commission, 2009). We furthermore assumed no recycling of P from human excreta and industrial processing water, as sewage sludge from communal processing water treatment plants is currently not reused in agriculture, and sewage sludge from industrial processing water treatment plants is only reused in agriculture to a limited extent (10%) (Smit et al., 2015, CBS, 2016a).

We subsequently explored the impact of prevention and recycling of wasted P in six alternative situations (Table 1). In the first alternative situation (P_Waste_Crop_Animal) we explored the impact of prevention of all waste of crop and animal products, including the full utilisation of animal meal. If waste of crop and animal products cannot be prevented, than the most promising strategy to reduce food waste according to the food waste hierarchy was recycling into feed or fertiliser (Papargyropoulou et al., 2014). In the second and third alternative situation (R_Waste_Crop and R_Waste_Animal), therefore, we assumed that waste of respectively crop and animal products (meat, milk, and animal meal) were fully recycled. In the fourth alternative situation (R_Humexc_ProcWater), we explored the impact of fully recycling human excreta and industrial processing water. This situation was chosen given the substantial waste of P from sewage sludge in the Netherlands (Smit et al., 2015), as human excreta, in particular urine, is rich in phosphorus, and, hence, recycling human excreta can substantially reduce the use of mineral P fertiliser (Jönsson et al., 2004). In the fifth and sixth alternative situation, we explored the impacts of a combination of prevention and recycling as applied in alternative situations 1 and 4 (Combi_1), and the impacts of a combination of recycling as applied in alternative situations 2, 3 and 4 (Combi_2).

Table 1. Characteristics of the baseline situation and alternative situations

	Waste of crop products		Waste of ASF		Slaughterhouse waste	HEPW
Situation	**Amount**	**Recycling**	**Amount**	**Recycling**	**Recycling**	**Recycling**
Baseline	current %	42%	current %	0%	0%	0%
P_Waste_Crop_Animal	0%	-	0%	-	100%	0%
R_Waste_Crop	current %	100%	current %	0%	0%	0%
R_Waste_Animal	current %	42%	current %	100%	100%	0%
R_Humexc_ProcWater	current %	42%	current %	0%	0%	100%
Combi_1	0%	-	0%	-	100%	100%
Combi_2	current %	100%	current %	100%	100%	100%

Note: ASF = animal-source food, HEPW = human excreta and processing water, P = prevention, R = recycling, Humexc_ProcWater = Human excreta and processing water. Crop products, animal products and HEPW can be recycled as feed or fertiliser.

2.3 Crop production system

Production of crops and forage were limited by the current Dutch agricultural areas of clay soils (839 10^3 ha), sandy soils (779 10^3 ha) and peat soils (224 10^3 ha) (Lesschen et al., 2012). We assumed that clay and sandy soils can be used for cultivation of crops and forage, whereas peat soils were assumed suitable only for cultivation of grass (Van Kernebeek et al., 2016). Crops in our model included wheat, potato, sugar beets, rapeseed and brown beans (Van Kernebeek et al., 2016). In addition to crops, we considered production of maize and grass silage, and fresh grass as forage for dairy cattle. Crops were cultivated in rotations (Van Kernebeek et al., 2016). We considered all above-ground biomass of the crops as potential food and/or feed ingredients, except for wheat and maize stubble, potato haulms, sugar beet leaves and bean straw. These parts of the crops were assumed to stay behind on the field as source of soil organic carbon, and as such also contributed to fertilisation with P. In the following section, we briefly describe crop fertilisation with P. We assumed nitrogen and potassium fertilisation such that P can be used efficiently by the crops and grassland. More details on fertilisation with P are provided in Supplementary Material S3.

2.4 Crop fertilisation

We assumed long-term stable P contents of soils and, hence, no net accumulation of P in soils. We consider this assumption justified and feasible for the situation in the Netherlands with large soil stocks of P (Verloop et al., 2010, Sattari et al., 2012). The effect of this assumption will be assessed in the results section for the baseline situation. Total amount of P required per ha for each crop rotation was computed from the P content of all crops in that rotation and unavoidable losses through leaching and run-off (Supplementary Material S3). The latter was assumed 2.2 kg P ha^{-1} on all soil types and crop rotations.

The required P was provided by variable sources, i.e. mineral fertiliser, animal manure, variable crop residues (defined here as co-products that could either be left on the field or be harvested as feed, i.e. wheat straw, sugar beet tops and tails, and rapeseed straw), crop products returned back to the land (either or not after they are wasted), human excreta and industrial processing water, wasted ASF and animal meal (Supplementary Material S3). For all recycled and organic fertiliser sources we assumed a P fertiliser replacement value relative to mineral fertiliser of 100% (De Haan and Van Geel, 2013, Severin et al., 2014). Availability of these resources depended, logically, on the situation explored and PA%.

2.5 Animal production system

We included two animal production systems with contrasting abilities to use marginal land: pig production as representative for monogastrics, who generally consume feed from land suitable for cultivation of crops, and dairy production as representative for ruminants, who can value marginal grassland. The dairy production system was a non-grazing system, as to avoid grass uptake inherent to grazing. We modelled dairy and pig production based on animal production units (PUs). One pig PU consisted of 3.3 fattening pigs, 0.12 sows and 0.07 gilts, and produced 171 kg pork per year (Van Kernebeek et al., 2016). One cow PU consisted of a dairy cow and its replacement stock, i.e. 0.31 replacement heifers aged 1-2 years, and 0.34 replacement calves aged 0-1 year (Van Kernebeek et al., 2016). We assumed that surplus calves were slaughtered directly after birth. One cow PU produced 8502 kg fat-and-protein-corrected-milk (FPCM) and 74 kg meat per year, both derived only from the milking cow (Van Kernebeek et al., 2016). Dietary requirements and intake restrictions of each PU are provided in Supplementary Material S4. As our feed ingredients contained relatively low digestible-P contents for pigs, we included a mineral source of phosphorus as potential P additive in the pig ration, to better enable a positive P balance in pigs. To treat cows and pigs equally, we also allowed this P additive in rations of cows. We chose MCP as P additive, with P digestibility of 83% for pigs (PDV, 2011) and 100% for cows. In situations where animal meal was recycled, animal meal could be consumed by animals with P digestibility of 74% for pigs (PDV, 2011) and 100% for cows.

2.6 Phosphorus excretion by animals and phosphorus retention in animal products

We computed P excretion by animals as the difference between P intake and P retention in animals and their products (excluding P retained in milk consumed by replacement calves, which eventually ends up in manure). P retention per animal PU was computed from P concentrations in body tissue and milk, and animal production data (Table 2 and Supplementary Material S5). We distinguished between P retained in ASF, which is either eaten or wasted, and in non-edible animal products. P retained in ASF that is consumed by humans (i.e. non-wasted ASF) finally ends up in human excreta, and is wasted in case human excreta are not recycled. P retained in wasted ASF was lost in situations where animal products were not recycled. Besides bones, organs, blood etc., non-edible products also included the bodies of surplus calves, as we assumed that these were eliminated after birth, and of dead animals. We assumed that all non-edible products from animals were converted into animal meal. P retained in animal meal was lost in situations where animal products were not recycled. Moreover, in situations where waste of ASF is prevented (P_Waste_Crop_Animal and Combi_1), more HEP per PU is available for human consumption (Table 2).

2.7 Crop processing

Harvested crops were assigned to industrial food processing or industrial feed processing, or were ensilaged (maize and grass silage) (Supplementary Material S1). We defined industrial food processing as resulting in multiple crops products, of which at least one is edible for humans (Van Kernebeek et al., 2016), whereas industrial feed processing resulted in a crop product that is edible for animals only. We assured closed P balances in industrial food processing; P content of harvested crop was equal to P content in the sum of output products (including waste) plus P content of processing water (Supplementary Material S6). In those cases where the P content in the sum of output products was lower than in the harvested crop, we assumed that the remaining P were dissolved in processing water. This was the case for potato and sugar beet processing.

2.8 Waste of crop and animal products

To account for waste of crop products and ASF, we applied waste fractions as estimated by Gustavsson et al. (2013) and Remmelink et al. (2012). Waste of various crops and crop products during post-harvest handling and storage ranged between 1 to 9% of dry matter (DM), during food processing and packing between 5 to 15% of DM, during distribution and human consumption between 5 to 27%, and during feeding between 2 to 29%. Waste of meat and milk during post-harvest handling and storage ranged between 0.5 to 0.7%, during processing and packing between 1.2 to 5%, during distribution and human consumption between 0.7 to 14.5% (Supplementary Material S2).

Table 2. Production of human edible protein (HEP) per year for a pig and a cow production unit (PU) (kg HEP year^{-1}), P retention in non-edible products, wasted ASF, and in non-wasted ASF (kg P kg^{-1} HEP), and P waste (kg P kg^{-1} HEP) for the baseline situation and alternative situations

	PigPU					CowPU				
	HEP[a]	**P retention**			**P waste**	**HEP**[a]	**P retention**			**P waste**
		Non-edible products	Wasted ASF	Non-wasted ASF			Non-edible products	Wasted ASF	Non-wasted ASF	
Baseline	44	0.034	0.002	0.009	0.045	269	0.0058	0.0028	0.027	0.036
R_Waste_Crop	44	0.034	0.002	0.009	0.045	269	0.0058	0.0028	0.027	0.036
R_Waste_Animal	44	0.034	0.002	0.009	0.009	269	0.0058	0.0028	0.027	0.027
R_Humexc_ProcWater	44	0.034	0.002	0.009	0.036	269	0.0058	0.0028	0.027	0.009
P_Waste_Crop_ Animal	55	0.027	0.000	0.009	0.009	298	0.0053	0.0000	0.027	0.027
Combi_1	55	0.027	0.000	0.009	0.000	298	0.0053	0.0000	0.027	0.000
Combi_2	44	0.034	0.002	0.009	0.000	269	0.0058	0.0028	0.027	0.000

Note: PU = production unit, HEP = human edible protein, ASF = animal-source food, [a]HEP available for consumption was computed from meat and milk production, N-content of meat and milk, and fractions of meat and milk waste along the chain.

4

3. Results

3.1 Baseline situation

Figure 2 shows the P flows through the food system in the baseline situation, in which at most 42% of crop waste was recycled, and diets contained 60% PA (the current average PA% in the Dutch diet (RIVM, 2009). The external input of mineral P into this food system was 16 103 ton per year, all in the form of mineral fertiliser. The amount of mineral P input equalled the sum of all wasted P in the food system, and P lost through leaching and run-off, as we assumed no accumulation of P in soils. The majority of the wasted P in the food system resulted from wasting valuable P in human excreta (8 509 ton), followed by P loss through leaching and run-off (2 773 ton), P waste along the crop production chain (2 660 ton), or the animal production chain (1 824 ton), and P waste in industrial processing water (337 ton). Recycling of P in human excreta in the food system, therefore, shows great potential to save mineral P.

When the %PA was increased in the baseline situation, the input of mineral P into the food system also increased (Figures 3 and 4). To unravel the observed relation between %PA and mineral P input, we present key parameters describing the P flow in the food system (Figure 2), for varying %PA in Table 3. We do this in two steps; first we explain the difference in P flows between a diet with 0% PA (i.e. a vegan diet) and a diet with 10% PA. Second, we explain the difference in P flows between a diet with 10% PA and one with 60% PA. The increase in P waste from a diet that contained 0% PA to a diet that contained 10% PA is a result of two opposite effects. On the one hand, a diet with 10% PA required less land, and therefore had lower associated P losses through leaching and run-off, than a diet with 0% PA (Table 3). The lower land use of a diet with about 10% PA is in agreement with results of Van Kernebeek et al. (2016), who demonstrated that land use was most efficient if people would consume a small amount of ASF derived from so-called default livestock (Fairlie, 2010). Default livestock converts co-products from crop production that are inedible for humans, such as wheat straw and sugar beet pulp, into protein-rich meat and milk. In a vegan diet, these human inedible co-products are not used for food production, and, hence, additional cropland is required to meet the energy and protein requirements of the population. A vegan diet, furthermore, results in higher P waste from processing and human consumption of crop products than a diet with 10% PA, in which crop products are partly displaced by animal products. On the other hand, a diet with 10% PA resulted in higher P waste during post-harvest storage of crop products used for feed, and higher feeding wastes. This can be explained as follows. In a vegan diet, co-products from crop production, such as wheat straw, were not harvested from the

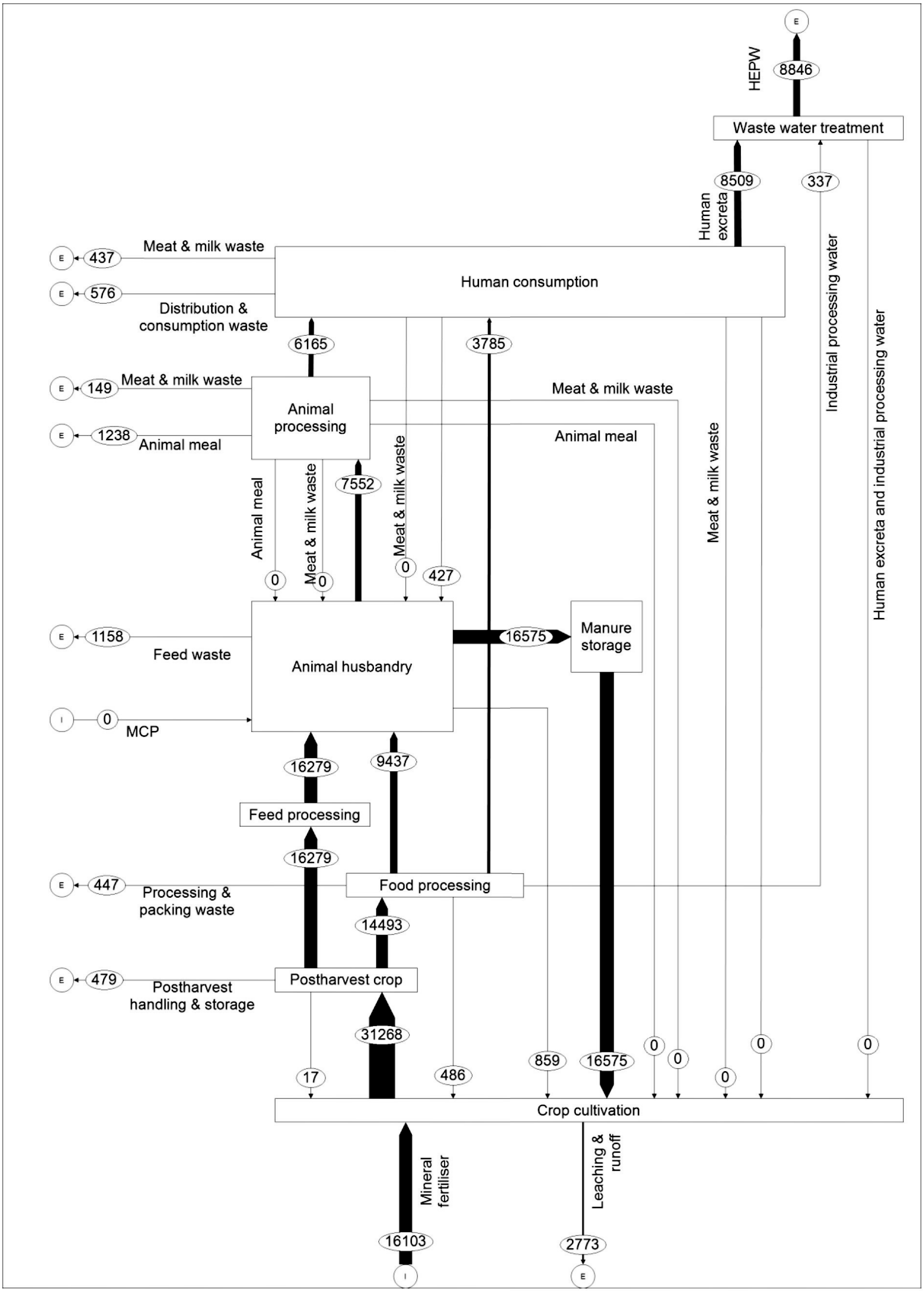

Figure 2. Phosphorus flows through the system (ton P) in the baseline situation with 60% PA. NOTE: MCP = monocalcium phosphate, HEPW = Human excreta and industrial processing water.

4

Table 3. Mineral P input requirement (ton), P input (ton), P waste (ton), number of cow and pig production units (PU) (1 000 PU), and land use (1 000 ha) in the baseline situation for diets varying in percentage of protein from animals (%PA)

		%PA	
	0	**10**	**60**
Mineral P input requirement	11 898	12 425	16 103
Fertiliser P	11 898	12 425	16 103
MCP	0	0	0
P loss			
Leaching and run-off	2 500	1 927	2 773
P waste			
Post-harvest storage			
Crop products	378	390	479
Processing			
Crop products	684	597	447
Industrial processing water	0	595	337
Animal husbandry			
Feed waste	0	104	1 158
Animal processing			
Meat and milk	0	25	149
Animal meal	0	206	1 238
Human consumption			
Meat and milk	0	73	437
Crop products	1 228	937	576
Human excretion	7 108	7 570	8 509
Number of animal units			
CowPU	0	132	790
PigPU	0	0	0
Land use	1 147	884	1 272

Note: %PA = percentage of protein from animals, MCP = monocalcium phosphate, PU = production unit land, but, instead, were left on the field as source of P. Harvesting these co-products to feed the animals, therefore, resulted in higher wastes during storage and during the feeding process on the farm. Logically, a diet with 10% PA also resulted in higher waste of animal products during processing of animals and their products, and higher waste of ASF during human consumption than a vegan diet. Finally, P waste through human excretion increased as the %PA increased. This increase in P waste has two causes. First, milk, being the main source of animal protein in a diet with 10% PA, has a higher P:N ratio than crop products. Second, human excreta were wasted in the baseline situation, implying that a diet with ASF resulted in a larger waste through human excreta than a vegan diet.

Overall, the mineral P input requirement of the food system increased from 0 to 10% PA, because the positive effects of a lower land use (i.e. lower P losses through leaching and run-off) and reduced processing and human consumption of crop products, were outweighed by the negative effects of producing and consuming ASF.

When increasing PA from 10 to 60%, the same trends in P waste were observed, except for P losses through leaching and run-off. These P losses increased as the demand for animal protein exceeded the amount that can be obtained from default livestock (Van Kernebeek et al., 2016). Unlike in a diet with 10% PA, where default livestock is fed merely on co-products from food production and processing, a diet with 60% PA required specific cultivation of feed crops to feed the animals (Van Kernebeek et al., 2016).

Cows appeared more P efficient than pigs in the baseline situation, explaining why milk and associated beef were the main source of animal protein in the human diet (Supplementary Material S7). This higher P efficiency of cows had two causes. First, cows were better able to convert available co-products, such as wheat straw, into protein-rich milk and meat (Van Kernebeek et al., 2016). Second, P waste via cows was lower than via pigs in the baseline situation, because P retained in animal products (non-edible products, wasted ASF, and non-wasted ASF) were not recycled. The amount of P retained in the sum of these products per kg of animal protein was lower for cows than for pigs (Table 2).

The above mentioned results hold for a situation with small P losses through leaching and run-off (2.2. kg P ha^{-1} $year^{-1}$). The mineral P input requirement would have increased with up to a factor two (67 to 105%) if we would have assumed leaching and run-off, or equivalent enhanced accumulation of P in soils, of 13 kg P ha^{-1} $year^{-1}$ (Supplementary Material S8). In that case, moreover, mineral P input was relatively constant between 0 to 20% PA, with a minimum at 10% PA, and subsequently increased with increasing PA%. In this situation, minimum mineral P input for a given PA% was achieved by minimising land use (Supplementary Material S7).

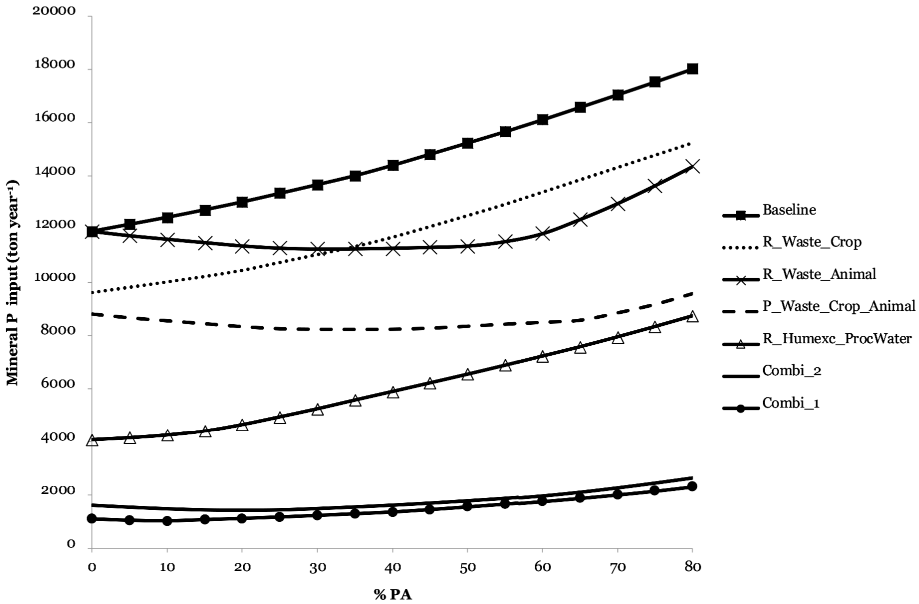

Figure 3. Mineral P input (ton year^{-1}) for the baseline situation and the alternative situations, for diets varying in their contribution of protein from animal (%PA). Note: See Table 1 for description of the alternative situations.

3.2 Alternative situations: impacts of prevention and recycling phosphorus waste

Figure 3 shows the mineral P input in the food system of prevention and recycling P waste to reduce the dependency on phosphate rock. We first describe the effect of prevention and recycling individually. Thereafter, we describe the effect of combinations of prevention and recycling. As expected based on results of the baseline situation (Figure 2; Table 3), recycling of P in human excreta and in processing water (R_Humexc_ProcWater) had greatest potential to reduce mineral P input requirement (Figure 3). Recycling of human excreta implied that P retained in edible plant products and ASF was not lost, but instead could be used to fertilise crops. In this situation, cows were more P efficient than pigs, and, hence, milk and associated beef were consumed as the source of animal protein (Supplementary Material S7). The higher P efficiency of cows followed from the fact that only the P retained in human non-edible products of animals and wasted ASF were lost. The amount of P retained in the sum of these products per kg edible protein was lower in cows than in pigs (Table 2). The increase in P input with increasing PA% was mainly due to increased losses of P through leaching and run-off, and animal and feeding wastes.

The second most promising option was prevention of waste along the crop and animal supply chain (P_Waste_Crop_Animal) (Figure 3). Because in this situation no wastes of crop products occurred, less feed crops were needed to meet nutritional requirements of animals than in the baseline situation. Similarly, less crops and ASF were required to meet nutritional requirements of the human population. In this situation, pigs appeared more P efficient than cows, and, hence, pork was consumed as the source of animal protein (Supplementary Material S7). This higher P efficiency of pigs had two causes. First, P retention in non-wasted ASF from cows was higher than from pigs. Second, P retained in non-wasted ASF was not recycled, and, thus, lost through human excreta. In this situation, mineral P input to the system decreased in the range from 0 to 35% PA, and subsequently increased (Figures 3 and 4). Over the full PA-range, mineral P input to the system was mainly determined by two opposite effects. On the one hand, when increasing PA%, P consumption by humans decreased. The decrease in P consumption was due to the increased displacement of crop products by pork; pork has a low P:N ratio compared to the P:N ratio of the human edible crop products included in our model. As a result of decreased P consumption by humans, P waste through human excreta also decreased. On the other hand, land use increased from a PA% of 20 upwards, and, consequently, P loss through leaching and run-off increased.

When recycling waste of all animal products (R_Waste_Animal), mineral P input by the system decreased for diets in the range from 0 to 30% PA, and subsequently increased. We will discuss mineral P input first for diets in the range from 0 to 60% PA, and subsequently for the range from 65 to 80% PA. For diets in the range from 0 to 60% PA, two opposite effects resulted in a relatively constant mineral P input. On the one hand, pigs were more efficient than cows (see P_Waste_Crop_Animal for explanation) (Supplementary Material S7), and, hence, pork was consumed as the source of animal protein. As a result, P waste through human excreta decreased due to decreased P consumption by humans (see P_Waste_Crop_Animal). On the other hand, P loss through leaching and run-off increased due to increased demand for feed. For diets in the range from 65 to 80% PA, mineral P input increased. In this range, pigs were still more P efficient than cows. However, from 65% PA upward, not enough cropland was available for sufficient production of feed for pigs. Due to this scarcity of cropland, pigs were partly displaced by cows, as cows can value grassland on peat soils (Van Kernebeek et al., 2016). Consequently, P loss through leaching and run-off increased only very slightly. However, following from the production of cows, milk and beef were included in the human diet. Consumption of milk resulted in relatively high P waste through human excretion, as milk has a high P:N ratio compared to other edible products. The increase in P input for diets in the range 60 to 80% PA was mainly caused by increased waste of P through human excretion.

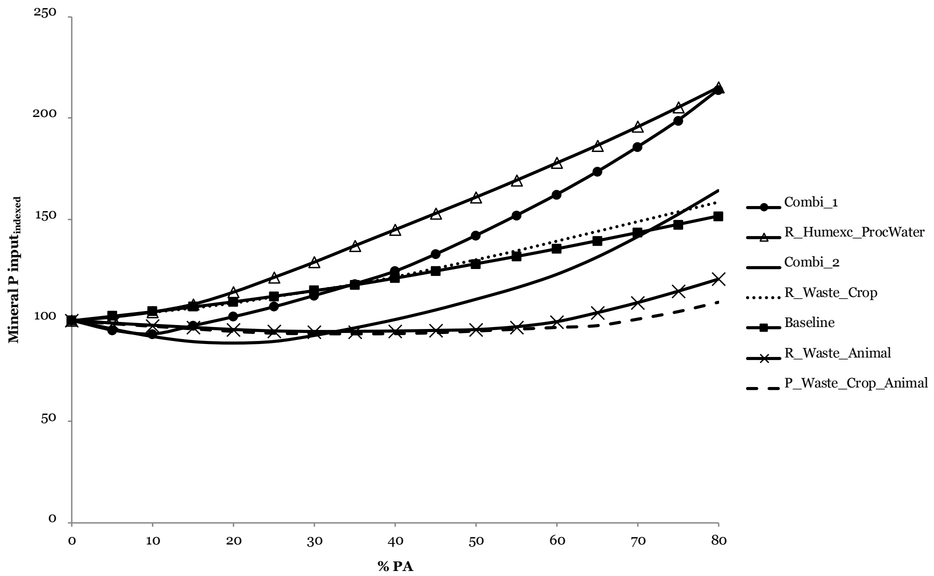

Figure 4. Mineral P input (indexed) for the baseline situation and the alternative situations, for diets varying in their contribution of protein from animal (%PA). Note: See Table 1 for description of the alternative situations.

The greatest potential for reducing mineral P input was when prevention of waste of crop and animal products and recycling of human excreta and processing water were combined (Combi_1) (Figure 3). This result is in line with the waste hierarchy (Papargyropoulou et al., 2014). When applying this combination of prevention and recycling, leaching and run-off was the only source of P loss, and, hence, mineral P input was only determined by land use. Combi_2, the combination of recycling of waste of crop products, waste of animal products, and human excreta and processing water, was less efficient than Combi_1; in Combi_2 waste of crop and animal products were recycled while these were prevented in Combi_1. Because these wastes were available for livestock in Combi_2, the default livestock diet was at higher PA% (20%) compared to the default livestock diet in Combi_1 (10%) (Figure 4).

4. Discussion

4.1 Strategies to lowering phosphorus input requirements

We assessed mineral P input requirement in the food system using an optimisation model. As our model minimised the mineral P input in a hypothetical food system, P input for our diets was lower, and P use efficiency higher, compared to estimates found in other studies. Metson et al. (2012), for example, estimated a mineral P input of 5 kg P cap^{-1} $year^{-1}$ for the production of the average diet in the Netherlands, whereas we found P inputs ranging from 0.7 to 1.0 kg P cap^{-1} $year^{-1}$. Metson et al. (2012), however, assumed that the P required by the system was provided by mineral P only, and excluded P provisioning by manure. The P input into our baseline food system would have been 2 kg P cap^{-1} $year^{-1}$ in case we would have excluded P recycling by manure. Recycling of manure was accounted for by most studies that estimated P efficiency of national food systems. P use efficiencies ranged from 6% in China (Bai et al., 2016) to 14 to 29% in the US and west-European countries (Suh and Yee, 2011, Jedelhauser and Binder, 2015). In our baseline situation with 60% PA, our food system yielded a P use efficiency of about 50%. An important explanatory factor for this relatively high P use efficiency was our assumption of small P losses through leaching and run-off (2.2 kg P ha^{-1} $year^{-1}$). The P use efficiency in our baseline food system would have ranged between 27 to 38%, depending on the % PA, in case we would have assumed that leaching and run-off, or equivalent enhanced accumulation of P in soils, was 13 kg ha^{-1} $year^{-1}$.

Our modelling exercise of a hypothetical food system provided valuable insights into the potential of prevention and recycling to reduce mineral P input requirements of the food system. In our baseline situation, in which waste of crop products was recycled to a limited extent, and P in waste of animal products, human excreta and processing water were not recycled, a vegan diet had lower P input compared to diets that contained PA. Recycling P from human excreta and industrial processing water showed most potential to reduce mineral P input requirements of the food system. By recycling human excreta and industrial processing water, mineral P requirements could be reduced by approximately 55 to 65%, depending on the % PA. This reduction potential was mainly due to the high waste of P through human excreta compared to other P waste of the system. When combining prevention of waste of crop and animal products with recycling of human excreta, mineral P input requirements were reduced by approximately 90%. We also demonstrated that, within our baseline situation, reducing animal protein consumption from the current rate of 60% towards a vegan diet (0% PA) reduced mineral P input requirements by approximately 25%. A vegan diet was, however, not most P use efficient in situations that included full prevention or recycling of animal products.

We demonstrated that P use efficiency was determined not only by the wastes and recycling rates, but also by P:N ratios in human edible products, and by the ability of animals to convert human inedible crop products. For example, in the alternative situation in which we recycled animal products, pig production was preferred in the 0 to 60% range for animal protein. At higher animal protein percentages, cows partly displaced pigs, as cows can value grassland on peat soils. As a result, P waste of the system increased because of the high P content in milk. If no waste of crop and animal products, and no waste through human excreta occurred, i.e. P was lost only by leaching and run off, P loss was minimised by minimising land use.

4.2 Implications of a hypothetical food system

Insights gained from our modelling exercise are that both preventing and recycling wasted P, and changing consumption of animal protein can reduce mineral P input to a system. Another insight gained is that the optimal (in terms of P input) consumption level of animal protein depends on the applied (combination of) prevention and recycling of P. Furthermore, we gained insight into the effect of prevention and recycling of P waste on the efficiency of animal production and consumption. We acknowledge that other food systems are bounded to other constraints, resulting from for instance differences in climatic and soil conditions, in different shares of land that can only be used as grassland, in farming practices, in representative types of crops and animals, and in population densities. These differences may lead to other P use efficiencies of crop and animal production systems, and may affect the optimal consumption level of animal protein and the (relative) importance of prevention and recycling to reduce mineral P input. The principles included in our model, however, also hold for other food systems. To illustrate this, we modelled P input for a system with higher P surpluses or P accumulation in the soils, and we concluded that in this situation mineral P input, as well as the optimal (in terms of P input) consumption level of animal protein, were higher compared to our baseline situation. We furthermore acknowledge that human diets in our system were constrained only by their energy, protein and sugar content, and were composed of a specific selection of crop and animal products. We included, for example, pigs as representative for monogastrics, assuming that poultry would behave similar to pigs in terms of P use. Inclusion of micronutrients and a wider range of crop and animal products would be required to conclude on P use efficiency of a system providing healthy and socially acceptable diets.

4.3 Other factors that determine phosphorus waste in food systems

The aim of this paper was to assess the potential of prevention and recycling P waste in the food system, in order to reduce the dependency on phosphate rock. To this end, our alternative situations included full prevention or full recycling of P waste. We did not account for the (technical) feasibility, legal aspects, and social acceptance of full prevention or recycling of these wastes. In comparison, the UN Sustainable Development Goal is to reduce waste by 50% by 2030, while the current P recovery rate from municipality processing water treatment plants in the Netherlands is over 80% (CBS, 2016b). Recovered P from municipality processing water treatment plants ends up mostly in building material such as asphalt (Luesink *et al.*, 2013). Use of recycled and recovered P from these sources in the food system is restricted by legislation (LNV/VROM, 1997). Moreover, the use of animal meal in feed for farmed animals is banned by European Union regulation as a measure to prevent, amongst others, the spread of Transmissible Spongiform Encephalopathies (TSE). Novel enzyme-based methods to destroy prion infectivity in animal meal have shown potential in degrading infectious prion proteins (Gupta *et al.*, 2013, Okoroma *et al.*, 2013). The potential to reduce mineral P input by recycling animal meal should be weighed against the (perceived) risk of the occurrence of TSE.

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Supplementary Material S1 Available crop products for humans, cows and pigs
Crop products available in this study, and an overview of whether or not products are edible or restricted for humans, cows and pigs

		Humans	Cows	Pigs
Industrial food processing				
Dry milling of wheat				
	Wheat middlings	n.c.	restr[1]	restr[1]
	Wheat germ	n.r.	restr[1]	restr[1]
	Wheat bran	n.r.	restr[1]	restr[1]
	Wheat flour	n.r.	restr[1]	restr[1]
Peeling of potato				
	Potato tuber	n.r.	n.r.	n.r.
	Potato peel	n.c.	restr[1]	restr[1]
Sugar beet processing				
	Sugar	restr	restr[1]	restr[1]
	Sugar factory lime	n.c.	n.c.	n.c.
	Sugar beet molasses	n.c.	restr[1]	restr[1]
	Sugar beet pulp	n.c.	restr[1]	restr[1]
Crushing of rapeseed				
	Rapeseed oil	n.r.	n.r.	restr[1]
	Rapeseed meal	n.c.	restr[1]	restr[1]
Industrial feed processing				
Grinding of wheat	Ground wheat grain	n.c.	restr[2]	restr[2]
Chopping of wheat straw	Chopped wheat straw	n.c.	n.r.	n.c.
Heating of potatoes	Potatoes	n.c.	restr[2]	restr[2]
Cutting of sugar beet	Cut sugar beet	n.c.	restr[2]	restr[2]
Cutting of sugar beet tops & tails	Cut sugar beet tops & tails	n.c.	restr[2]	n.r.
Grinding of rapeseed	Ground rapeseed	n.c.	restr[2]	restr[2]
Chopping of rapeseed straw	Chopped rapeseed straw	n.c.	restr[2]	n.c.
Feed or food processing				
	Brown beans	n.r.	restr[1]	restr[1]
Ensilaging				
	Silage maize	n.c.	n.r.	n.c.
	Silage grass	n.c.	n.r.	n.c.
No processing				
	Fresh grass	n.c.	n.r.	n.c.

Note: n.c. = not consumed, we did not allow this product to be consumed; n.r. = not restricted, this product could be consumed without dietary restriction, restr = restriced, consumption of this product was restricted; [1]Van Kernebeek et al. (2016); [2]Section Animal production system in Supplementary Material S4.

Supplementary Material S2. Waste of crop and animal products along the chain
Waste of crops and crop products are provided in Table S1. In addition, during animal processing we assumed 6% waste of meat and 2% waste of milk (Gustavsson et al., 2011). Moreover, during human consumption we assumed 15% waste of meat and 8% waste of milk (Gustavsson et al., 2011).

Table S1 Post-harvest waste (%) of crop products during various steps in the food and feed chain

	Post-harvest storage	Processing	Human consumption	Animal husbandry
Prior to processing				
Wheat grain	4			
Wheat straw	5			
Potato	9			
Sugar beet	9			
Sugar beet tops&tails	5			
Rapeseeds	1			
Rapeseed straw	5			
Beans	1			
Industrial food processing				
Dry milling of wheat				
Wheat middlings		5	27	2
Wheat germ		5	27	2
Wheat bran		5	27	2
Wheat flour		5	27	2
Peeling of potato				
Potato tuber		15	23	22
Potato peel		15		10
Sugar beet processing				
Sugar		15	23	2
Sugar factory lime		15		
Sugar beet molasses		15		2
Sugar beet pulp		15		7
Crushing of rapeseed				
Rapeseed oil		5	5	2
Rapeseed meal		5		2

	Post-harvest storage	Processing	Human consumption	Animal husbandry
Industrial feed processing				
Grinding of wheat				
Ground wheat grain				2
Chopping of wheat straw				
Chopped wheat straw				24
Heating of potatoes				
Potatoes				22
Cutting of sugar beet				
Cut sugar beet				7
Cutting of sugar beet tops & tails				
Cut sugar beet tops & tails				29
Grinding of rapeseed				
Ground rapeseed				2
Chopping of rapeseed straw				
Chopped rapeseed straw				24
Feed or food processing				
Brown beans			5	15
Ensilaging				
Silage maize				12
Silage grass				19

Note: based on Remmelink et al. (2012) and Gustavsson et al. (2011)

Supplementary Material S3 Crop fertilisation

Total amount of P required per ha for each crop rotation was computed from the P content of all crops in that rotation and assumed unavoidable losses through leaching and run-off (Eq. 1) (Table S2). Wheat and maize stubble, potato haulms, sugar beet leaves and bean straw were not included, as we assumed that these parts of the crops stayed behind on the field as a source of P for the subsequent crop.

$$\mathrm{TR_{i,l}} = \sum_{\mathrm{j=1}} \mathrm{Y_{j,i,l}} \times \mathrm{DM_j} \times \mathrm{Pcont_j} + \mathrm{UL_{i,l}} \qquad \text{Eq. 1}$$

Where $TR_{i,l}$ is the total requirement of P per ha (in kg ha^{-1}), for crop rotation (i) on land type (l), based on the sum of all harvested products (j) from that rotation, including main and co-products (Supplementary Material S1); Y is the fresh matter yield of a harvested product (ton ha^{-1}) (Online resource I in Van Kernebeek et al., (2016)), DM is the dry matter content of a harvested product (Online Source I in Van Kernebeek et al., (2016)), Pcont is the nutrient content of a harvested product (kg ton^{-1} DM) (PDV, 2011), and UL is the unavoidable P loss (kg ha^{-1}) through leaching and run-off, which was assumed 2.2 kg P ha^{-1} on all soil types and crop rotations (Rijksoverheid, 2014).

Total amount of P required per ha for each crop rotation was provided by variable sources according to Eq. 2. For all recycled and organic fertiliser sources we assumed a P fertiliser replacement value relative to mineral fertiliser of 100% (De Haan and Van Geel, 2013, Severin et al., 2014).

$$\begin{aligned}\mathrm{TR_{i,l}} = {} & \mathrm{MF_{i,l}} + \sum_{\mathrm{a=1}}^{2}\sum_{\mathrm{b=1}}^{41} \mathrm{Man_{i,l,a,b}} \times \mathrm{ManConc_{a,b}} + \sum_{\mathrm{j=1}} \mathrm{VCR_{i,l,j}} \times \mathrm{Nutrcont_j} \\ & + \sum_{\mathrm{k=1}} \mathrm{Crp_{i,l,k}} \times \mathrm{Nutrcont_k} + \mathrm{HumanexcProcWater_{i,l}} + \mathrm{WasteAnimal_{i,l}} \\ & + \mathrm{Animalmeal_{i,l}}\end{aligned} \qquad \text{Eq. 2}$$

Where $TR_{i,l}$ is the total fertiliser requirement of P for crop rotation (i) and soil type (l) (kg ha^{-1}), $MF_{i,l}$ is the amount of P from mineral fertiliser (triple superphosphate) (kg ha^{-1}). $Man_{i,l,a,b}$ is the volume of applied manure (ton DM) of manure type (b) produced in animal production system type (a). Manure types (b) differed in their nutrient concentrations. $ManConc_{a,b}$ is the P concentration in manure (kg ton^{-1} DM) per manure type and animal production system type, $VCR_{i,l,j}$ is the amount of variable crop residue (ton DM) (j) left for crop rotation (i) on soil type (l). $Nutrcont_{j,}$

is the P content (kg ton^{-1} DM) in variable crop residue (j), $Crp_{i,l,k}$ is the volume of crop product (k) returned back to the land (ton DM ha^{-1}), $Nutrcont_k$ is the P content of crop product (k) returned back to the land. $HumanexcProcWater_{i,l}$ is the amount of P (kg ha^{-1}) from recycled human excreta and industrial processing water. $WasteAnimal_{i,l}$ is the amount of P (kg ha^{-1}) from recycled waste of ASF, and $Animalmeal_{i,l}$ is the amount of P (kg ha^{-1}) from recycled animal meal. We did not allow fertilisation of grassland by crop residues or crop products returned back to the land.

Table S2 Total requirement (TR) of phosphorus (P) by crop rotation and soil type (kg ha^{-1})

		TR (kg ha^{-1})
Rotation[a]	**Land type**	**P**
G	Clay	47
M	Clay	32
WOWB	Clay	25
PWSW	Clay	29
PBSW	Clay	25
WOWBS	Clay	26
WOWBWP	Clay	26
G	Sand	44
M	Sand	33
WOWB	Sand	24
PWSW	Sand	27
PBSW	Sand	24
WOWBS	Sand	24
WOWBWP	Sand	25
G	Peat	45

[a]G=grass, M=silage maize, W= wheat, O= oilseed, B=beans, P = potato, S = sugar beet.

Supplementary Material S4 Dietary requirements and intake restrictions of animals

Dietary requirements of each PU regarding energy and protein intake, digestibility, structure and intake restrictions are described in detail in Van Kernebeek et al. (2016). In addition to these feed restrictions, we also accounted for feed restrictions for products that resulted from feed processing (Table S3).

Table S3 Feed restrictions per cow and pig production unit (PU) for products that resulted from feed processing

	CowPU	PigPU	Based on source
Max. feed intake (ton DM) per animal PU year^{-1}			
Potato	1.78	0.43	Feedipedia (2017)
Wheat grain	3.67		Feedipedia (2017)
Sugar beet tops&tails	1.23		Feedipedia (2017)
Rapeseed	0.90		Emanuelson et al. (1991) and Rymer and Short (2003)
Rapeseed straw	0.14		Vestjens (2017)
Max. fraction of total dry matter intake			
Wheat grain		0.4	Feedipedia (2017)
Rapeseed		0.05	Pharazyn (2016)
Beans		0.2	Feedipedia (2017)
Sugar beet	0.4	0.056	Feedipedia (2017)

Supplementary Material S5 Phosphorus retention in animals

P retention per animal PU was fixed, and was computed from P concentrations in body tissue and milk (Groenestein et al., 2008, RVO, 2010), and production data (Van Kernebeek et al., 2016) (Supplementary Table S4). P retention in body tissue per cowPU included retention in replaced dairy cow, surplus calves, and deceased replacement calves. Retention in human non-edible products was computed as 'retention in body tissue minus retention in meat'. Retention in milk for human consumption was computed as 'retention in raw milk minus retention in milk for replacement calves'.

Table S4 Production of meat and milk per pig and cow production unit (PU) per year, phosphorus (P) retention in body tissue, meat, milk, and human non-edible products, and P content of meat and milk

		Retention (kg)	**Content (g kg^{-1})[b]**
	kg[a]	**P**	**P**
PigPU			
Body tissue		2.0	
Of which meat	171	0.51	3.0
Of which human non-edible products		1.5	
CowPU			
Body tissue		1.8	
Of which meat	74	0.20	2.7
Of which human non-edible products		1.6	
Raw milk	8 120	7.9	
Of which for replacement calves	79.2	0.08	
Milk for human consumption (FPCM)	8 502	7.8	0.92

Note: FPCM = Fat and protein corrected milk. [a]See Online resources III and VI in Van Kernebeek et al. (2016) for herd composition and meat and milk production per animal PU, [b]P contents of meat were taken from RIVM (2013). P content of milk for human consumption was computed from production (kg) and P retention. P content of our milk for human consumption was comparable with the content of full fat milk as presented by the Dutch Food Composition Table NEVO (RIVM, 2013) (i.e. 1.02 g P kg^{-1}).

Supplementary Material S6 Nutrient balances in crop processing

To assure nutrient balances in crop processes that involved separation of harvested crop into multiple crop products, we compared the nutrient content (kg of P and N per ton DM) of each harvested crop before processing (PDV, 2011) with the nutrient content of the sum of output products (including wastes), which we computed from nutrient content per ton dry matter (PDV, 2011) and output/input ratios (Van Kernebeek et al., 2016). We computed N content as 16% of crude protein (PDV, 2012). In those cases where the nutrient content in the sum of output products was lower than in the harvested crop, we assumed that the remaining nutrients were dissolved in industrial processing water. This was the case for two processes, i.e. potato and sugar beet processing. During potato processing, 0.46 kg of 2.50 kg P ton^{-1} DM, and 1.42 kg of 16.32 kg N ton^{-1} DM potato tuber ended up in industrial processing water. During sugar beet processing, these quantities were 0.51 kg of 1.6 kg P ton^{-1} DM, and 1.04 kg of 6.56 kg N ton^{-1} DM sugar beet. In those cases where nutrient content in the sum of output products was higher than in the harvested crop, we lowered the nutrient content of output products by solving a system of linear equations such that the initial nutrient ratio (PDV, 2011) between output products remained unchanged. This was the case for the remaining two food processes that involved separation of harvested crop into multiple crop products, i.e. dry milling of wheat, and crushing of rapeseed. The nutrient contents of the output products of the dry milling of wheat were lowered from 2.8% to 2% N, and from 0.68% to 0.35% P. The nutrient contents of the output products from rapeseed crushing were lowered with less than 1%. To account for the relation between N and protein, we lowered the contents of intestinal digestible protein and rumen degradable protein in feed ingredients for cows with the same percentage as the percent-change in N.

Supplementary Material S7
Number of cow and pig production units (PU) (1 000 PU), land use (1 000 ha), phosphorus (P) loss through leaching and run-off (ton) and P waste through human excreta (ton) for the baseline situation, the alternative situations, and for the situation assuming higher P-surplus of 13 kg ha^{-1} $year^{-1}$ (instead of 2.2 kg ha^{-1} $year^{-1}$). Results presented for various percentages of protein from animals (%PA).

	%PA in human diet	**0**	**10**	**15**	**20**	**25**	**60**	**65**	**80**
Baseline	Number of cowPU	0	132	197	263	329	790	855	1 053
	Number of pigPU	0	0	0	0	0	0	0	0
	Land use	1 147	884	895	899	916	1 272	1 351	1 662
	P loss through leaching and run-off	2 500	1 927	1 952	1 959	1 998	2 773	2 945	3 622
	P waste through human excreta	7 108	7 570	7 622	7 675	7 728	8 509	8 639	9 028
P_Waste_Crop_ Animal	Number of cowPU	0	0	0	0	0	0	0	0
	Number of pigPU	0	648	972	1 296	1 620	3 888	4 211	6 162
	Land use	869	847	830	813	823	1 493	1 611	1 618
	P loss through leaching and run-off	1 894	1 847	1 810	1 773	1 794	3 255	3 512	3 526
	P waste through human excreta	6 894	6 506	6 329	6 152	6 000	4 776	4 594	5 108
R_Waste_Crop	Number of cowPU	0	132	197	263	329	790	855	1 053
	Number of pigPU	0	0	0	0	0	0	0	0
	Land use	1 147	1 131	1 124	1 099	1 048	1 242	1 336	1 620
	P loss through leaching and run-off	2 500	2 466	2 450	2 396	2 284	2 707	2 912	3 531
	P waste through human excreta	7 108	7 241	7 307	7 397	7 531	8 509	8 639	9 028
R_Waste_Animal	Number of cowPU	0	0	0	0	0	0	90	550
	Number of pigPU	0	804	1 206	1 608	2 010	5 177	5 318	4 216
	Land use	1 147	1 092	1 062	1 043	1 119	1 618	1 619	1 688

4

	%PA in human diet	0	10	15	20	25	60	65	80
	P loss through leaching and run-off	2 500	2 382	2 315	2 273	2 438	3 526	3 529	3 679
	P waste through human excreta	7 108	6 695	6 493	6 279	5 982	5 126	5 582	7 367
R_Humexc_ ProcWater	Number of cowPU	0	132	198	267	342	790	855	1 100
	Number of pigPU	0	0	0	0	0	0	0	0
	Land use	847	772	789	803	845	1 219	1 286	1 499
	P loss through leaching and run-off	1 847	1 683	1 721	1 750	1 842	2 657	2 803	3 269
	P waste through human excreta	0	0	0	0	0	0	0	0
Combi_1	Number of cowPU	0	119	179	250	322	909	1 017	1 419
	Number of pigPU	0	0	0	0	0	0	0	0
	Land use	498	465	486	508	532	809	865	1 066
	P loss through leaching and run-off	1 087	1 014	1 059	1 107	1 160	1 763	1 885	2 324
	P waste through human excreta	0	0	0	0	0	0	0	0
Combi_2	Number of cowPU	0	133	206	275	289	1 067	1 223	1 496
	Number of pigPU	0	0	0	51	495	0	0	0
	Land use	740	680	661	656	662	908	972	1 216
	P loss through leaching and run-off	1 614	1 483	1 442	1 431	1 444	1 980	2 118	2 651
	P waste through human excreta	0	0	0	0	0	0	0	0
Higher P surplus	Number of cowPU	0	132	197	263	329	790	863	1 108
	Number of pigPU	0	0	0	0	0	0	0	0
	Land use	740	688	702	726	756	1 174	1 237	1 493
	P loss through leaching and run-off	9 624	8 944	9 126	9 439	9 825	15 262	16 084	19 408
	P waste through human excreta	7 924	7 999	7 793	7 769	7 818	8 761	8 987	9 653

Supplementary Material S8 The effect of assuming higher phosphorus surplus (13 kg ha^{-1} year^{-1})

MacDonald et al. (2011) reported that top quartile fields with surpluses globally had P surpluses of more than 13 kg P ha^{-1} year^{-1}. We explored the effect of assuming P surpluses or P accumulation of 13 kg P ha^{-1} year^{-1} while all other parameters were equal to that in the baseline situation (Figure S1). In this situation, mineral P input varied roughly between 21 000 ton (at 10% PA) and 35 000 ton (at 80% PA). The absolute difference in mineral P input between our baseline situation (P loss through leaching and run-off is 2.2 kg ha^{-1} year^{-1}) and the situation with 13 kg P loss ha^{-1} year^{-1} through leaching and run-off or equivalent enhanced accumulation of P in soils, varies roughly between 8 400 and 18 000 ton P (i.e. 67 to 105% more mineral P input requirement in the situation with higher P surplus). In P-saturated soils, an important strategy therefore would be to lower P surplus. If P surplus is not lowered, mineral P input was minimised by minimising land use (Supplementary Material S7). The P use efficiency (P consumed by humans (Supplementary Material S7) over mineral P input) in the situation with higher P surplus ranged between 27% (at 80% PA) and 38% (at 15% PA).

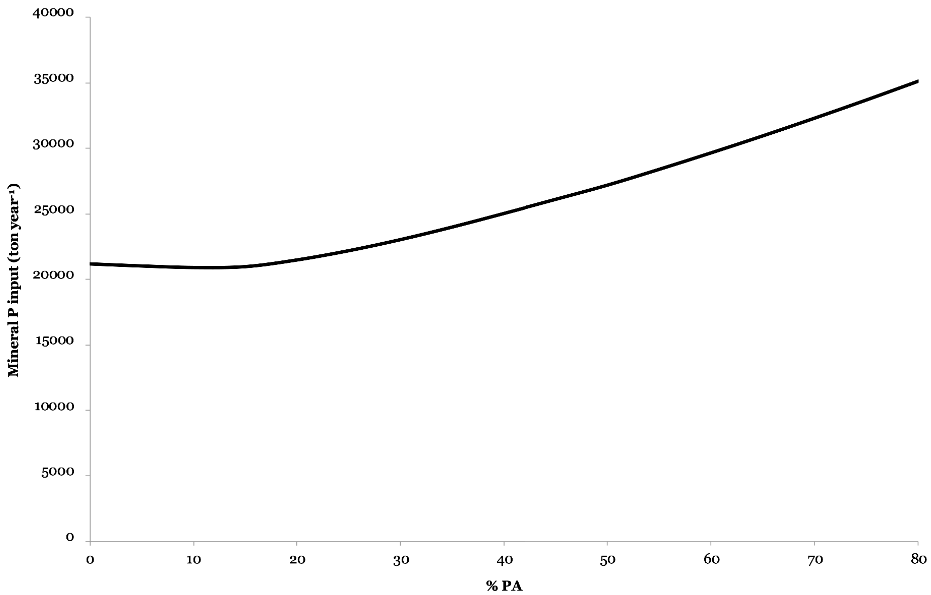

Figure S1. Mineral phosphorus (P) input (ton year^{-1}) in relation to percentage of protein from animals (% PA) assuming P surpluses of 13 kg P ha^{-1} year^{-1} while all other parameters are equal to that in the baseline situation.

4

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Chapter 5

Understanding the energy input into a food system – the combined effects of waste prevention, anaerobic digestion and dietary shifts

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Submitted

Abstract

Our food system highly depends on the input of fossil energy. As using fossil energy contributes to climate change, and fossil energy is becoming increasingly scarce, reducing the input of fossil energy to the food system is essential to sustain food security. The aim of our modelling exercise was to assess the potential of preventing waste, recycling waste as animal feed or fertiliser and recovering waste as bioenergy, via anaerobic digestion, to reduce the overall energy input in the food system. We assessed the potential of these strategies across human diets differing in the percentages of protein from animals (%PA). We defined energy input as the difference between energy that is used during activities in the food system, and energy that is recovered through anaerobic digestion. We modelled a hypothethical food system designed to produce sufficient food for a fixed population with minimum energy input. This model included representative crop and animal production systems, and was parameterised using data from the Netherlands. We assessed a baseline situation, and an alternative situation in which waste was prevented. To avoid the cultivation of crop biomass exclusively for bio-energy production, we first minimised land use for food self-sufficiency in both the baseline and the alternative situation without anaerobic digestion. Subsequently, we assessed the impact of introducing anaerobic digestion in both the baseline and the alternative situation.

We concluded that energy input into the food system was reduced by anaerobic digestion and waste prevention as single interventions. If waste was not prevented, the effect of anaerobic digestion was strongest in situations where animals did not compete for food waste and inedible crop products (at 0% PA, i.e. a vegan diet), and feed waste (i.e. at 80% PA) with anaerobic digestion. If waste was prevented, the relatively high potential to recover bio-energy from waste at 0 and 80% PA was lacking. We furthermore concluded that in situations with anaerobic digestion and/ or waste prevention, energy input continuously increased with increasing %PA, and, hence, a vegan diet was most energy efficient. In the baseline situation where none of these strategies were applied, however, energy input showed a minimum at about 15% PA. This implies that, depending on whether or not strategies of anaerobic digestion and waste prevention are applied, a vegan diet or a diet with a modest amount of animal protein is most energy-efficient. Our study shows that, to reduce energy input to a food system, it is essential to account for the combined effects of waste prevention, anaerobic digestion and dietary shifts.

1. Introduction

The use of fossil energy raises increasing concerns about global energy security and environmental impacts, such as climate change. The food sector is responsible for about one quarter of the total energy use within the European Union (EU), the main part of which is fossil energy (Monforti-Ferrario et al., 2015; Sims and Dubois, 2011). To reduce the fossil energy input into the food system, several strategies have been proposed, such as preventing food waste, recycling of food waste as animal feed or fertiliser, recovering food waste as bio-energy via, for example, anaerobic digestion and eating less animal-source food in high income countries (Papargyropoulou et al., 2014).

As significant amounts of our food produced in the EU are wasted along the chain, prevention of food waste is a first priority to reduce fossil energy use and other environmental impacts (Papargyropoulou et al., 2014; Vandermeersch et al., 2014). At the same time, food waste that is unavoidable can be recycled as animal feed and fertiliser, or recovered for the production of bio-energy (Tonini et al., 2016). Further, consumption of animal-source food (ASF) is often seen as a major contributor to the energy use of the EU food system (Monforti-Ferrario et al., 2015). It is argued that because of the biomass losses during the conversion of plant biomass into ASF, the production of ASF is less energy-efficient than the production of plant-source foods (Pelletier et al., 2011; Pimentel and Pimentel, 2003).

Studies that address the potential of preventing waste, recycling food waste as animal feed, fertiliser or bio-energy source, or dietary shifts, however, do not include all consequences for the food system that are relevant for a sound evaluation (Tonini et al., 2016; Truong et al., 2019; Tufvesson et al., 2013; Van Stappen et al., 2016). A sound evaluation of above mentioned strategies should include the following consequences.

First, to evaluate the potential of prevention of food waste, we must also consider the fact that, once prevented, this waste is no longer available as animal feed, fertiliser or biomass source for anaerobic digestion.

Second, unavoidable food waste can be recycled as animal feed. Animals indeed do have the ability to convert biomass that humans cannot or do not want to consume, such as food waste, into animal-source food (ASF), such as meat and milk and manure. Feeding animals with these human inedible products can increase the land-use efficiency of a food system (Van Kernebeek et al., 2016). We expect that feeding animals with human inedible crop products, therefore, could also reduce energy use in the food system. However, as mentioned above, animal production and bio-energy production compete for the same human inedible biomass. Feeding these resources to animals, and not to the anaerobic digester, therefore reduces the potential to produce biogas.

Third, besides from food waste we can also produce biogas via anearobic digestion from manure and other co-products. Biomass suitable for anaerobic digestion includes crop residues, co-products from the food and feed industry, food waste, and animal manure (Achinas et al., 2017; RVO, 2013). Currently, some crops are also cultivated specifically for the production of renewable energy via anaerobic digestion (CBS, 2016). However, crop products used for energy production compete for land with crop products that can be consumed directly by animals or humans (De Vries et al., 2012; Tonini et al., 2016; Valin et al., 2015; Van Stappen et al., 2016), and it may be argued that crops should not be cultivated just for energy production. Besides bio-energy, anaerobic digestion also yields digestate, which can replace mineral fertiliser (Miranda et al., 2015; Tufvesson et al., 2013). The production of mineral fertiliser is among the main contributors to energy use in crop cultivation (Bos et al., 2014; Pelletier et al., 2011; Williams et al., 2010). Substitution of mineral fertiliser by digestate, therefore, could reduce energy use in crop cultivation.

So far, it is unclear how preventing waste, recycling waste as animal feed or fertiliser and recovering waste as bioenergy, via anaerobic digestion, will ultimately affect the overall energy input in the food system. We deem this a relevant knowledge gap as energy input in the food system is mainly acquired from fossil energy sources (Monforti-Ferrario et al., 2015; Sims and Dubois, 2011). The aim of our modelling exercise, therefore, was to assess the potential of preventing waste, recycling waste as animal feed or fertiliser, and recovering waste via anaerobic digestion to reduce energy input to the food system. We defined energy input as the difference between energy that is used during activities in the food system, and energy that is recovered through anaerobic digestion. To investigate our knowledge gap, we modelled a hypothethical food system designed to produce sufficient food for a fixed population with minimum energy input. This model included representative crop and animal production systems, and was parameterised using data from the Netherlands. We assumed no import or export of feed and food.

2. Material and methods

We compared energy input to the food system between a baseline situation, and an alternative situation in which waste was prevented (Table 1). Within both situations, we studied the effects of anaerobic digestion, and of consumption of ASF. To quantify energy input in each situation, we extended the optimisation model developed by Van Kernebeek et al. (2018) This extended model had the objective function to minimise energy input while producing sufficient food for a fixed population. We defined energy input as the difference between, on the one hand, energy use during crop production, crop processing, animal production, production of monocalcium phosphate (MCP), animal processing, storage in warehouse and retail, home consumption, anaerobic digestion and transport, and, on the other hand, energy recovery through anaerobic digestion (Figure. 1, Eq 1). We quantified energy as primary energy (Supplementary material S1).

$$Min\left(\left(\sum_{i=1}^{7}\sum_{l=1}^{3} EU_{crop\ production,i,l} + \sum_{m=1}^{16}\sum_{j=1}^{8} EU_{crop\ processing,m,j} + \sum_{k=1}^{2} EU_{animal\ production,k} + \sum_{k=1}^{2} EU_{MCP\ production} + \sum_{o=1}^{3} EU_{animal\ processing,o} + \sum EU_{WHR} + \sum_{p=1}^{4} EU_{home\ consumption,p} + \sum_{r=1}^{2} EU_{anaerobic\ digestion,r} + EU_{transport}\right) - \sum_{r=1}^{2} ER_{anaerobic\ digestion,r}\right) \quad \text{Eq. 1}$$

Where $EU_{crop\ production,i,j}$ is energy use for production of crops on crop rotation *i* on soil type *l*, $EU_{crop\ processing,\ m}$ is energy use during crop processing *m* of crop product *j*, $EU_{animal\ production,k}$ is energy use during animal production in animal production system *k*, $EU_{MCPproduction}$ is energy use for production of monocalcium phosphate (MCP) for use in animal production system *k* $EU_{animal\ processing,o}$ is energy use during animal processing of animal product *o*, EU_{WHR} is energy use in warehouse and retail, $EU_{home\ consumption,p}$ is energy use during home processing *p*, $EU_{anaerobic\ digestion,r}$ is energy use during anaerobic digestion in anaerobic digester type *r*, $EU_{transport}$ is energy use during transport of all products throughout the system, and $ER_{anaerobic\ digestion,r}$ is energy recovery during anaerobic digestion in anaerobic digester type *r*. All energy was expressed in TJ.

Table 1. Characteristics of the Baseline situations and alternative situations in which waste is prevented (WPREV)

	Waste of crops, meat and milk					**Slaughterhouse waste**				**Manure**	
		Recycling options				**Recycling options**				**Recycling options**	
Situation	Amount	Recycling	Feed	Fertiliser	Energy	Recycling	Feed	Fertiliser	Energy	Fertiliser	Energy
Baseline_no_AD	current %	50%	yes	yes	no	50%	yes	yes	no	yes	no
Baseline_AD	current %	50%	yes	yes	yes	50%	yes	yes	yes	yes	yes
WPREV_no_AD	0%	-	-	-	-	100%	yes	yes	no	yes	no
WPREV_AD	0%	-	-	-	-	100%	yes	yes	yes	yes	yes

Note: AD = anaerobic digestion, WPREV = waste prevention. Current percentages of waste of crops, meat and milk are presented in Van Kernebeek et al., 2018. Production of slaughterhouse waste is inherent to animal production. To prevent wasting these products, they are fully recycled in situations WPREV_no_AD and WPREV_AD.

2.1 System definition

Our hypothetical food system included the following processes: crop production, crop processing, animal production, MCP production, animal processing, storage in warehouse and retail, home consumption, anaerobic digestion, and transportation (Figure 1). The system was designed to produce sufficient dietary energy and protein to feed a population of 17 million people, which is approximately the current population size in the Netherlands. Daily per capita nutritional requirements were defined as 2,000 kcal and 57 g protein (EFSA, 2009, 2012), and sugar intake was limited to the maximum recommended intake level of 32.9 kg per capita per year (EFSA, 2009). The model included seven types of crops, i.e. wheat, potato, sugar beet, rapeseed, beans, silage maize and grass. These crops were included as they have the largest cultivated area per food crop group (i.e. grains, roots and tubers, oil crops and legumes) in the Netherlands (Van Kernebeek et al., 2016). Furthermore, the model included two types of animals, i.e. pigs and dairy cows. These animals were included as they have the largest contribution to protein in the Dutch human diet within the groups of monogastrics and ruminants respectively (Van Kernebeek et al., 2016). The hypothetical food system was parameterised using data for the Netherlands (Van Kernebeek et al., 2016). Our system was assumed to be a closed, self-sufficient system, hence we assumed no imports and exports of food and feed.

2.2 Strategies to reduce energy use in the food system

We computed energy input to the system for a baseline situation, and an alternative situation in which waste was prevented (Table 1). In the baseline situation, we assumed that crop products, meat and milk were wasted at rates equal to current rates in the Netherlands (Van Kernebeek et al., 2018). In compliance with current recycling rates of food waste in the Netherlands (Soethoudt and Timmermans, 2013), we assumed that 50% of wasted crops, meat and milk were recycled. These waste streams could be recycled as animal feed, fertiliser, or bio-energy source. Wasted feed could only be recycled as fertiliser or bio-energy source. To treat crop and animal production sub-systems equally, we also assumed 50% recycling of slaughterhouse waste as feed, fertiliser or bio-energy source, despite EU regulation hindering this (European Commission, 2009). To avoid trade-offs between energy production and land use, we first minimised land use for the baseline and alternative situation. We subsequently minimised energy input for both situations while keeping the number of hectares per type of crop rotation equal to that when minimising land use.

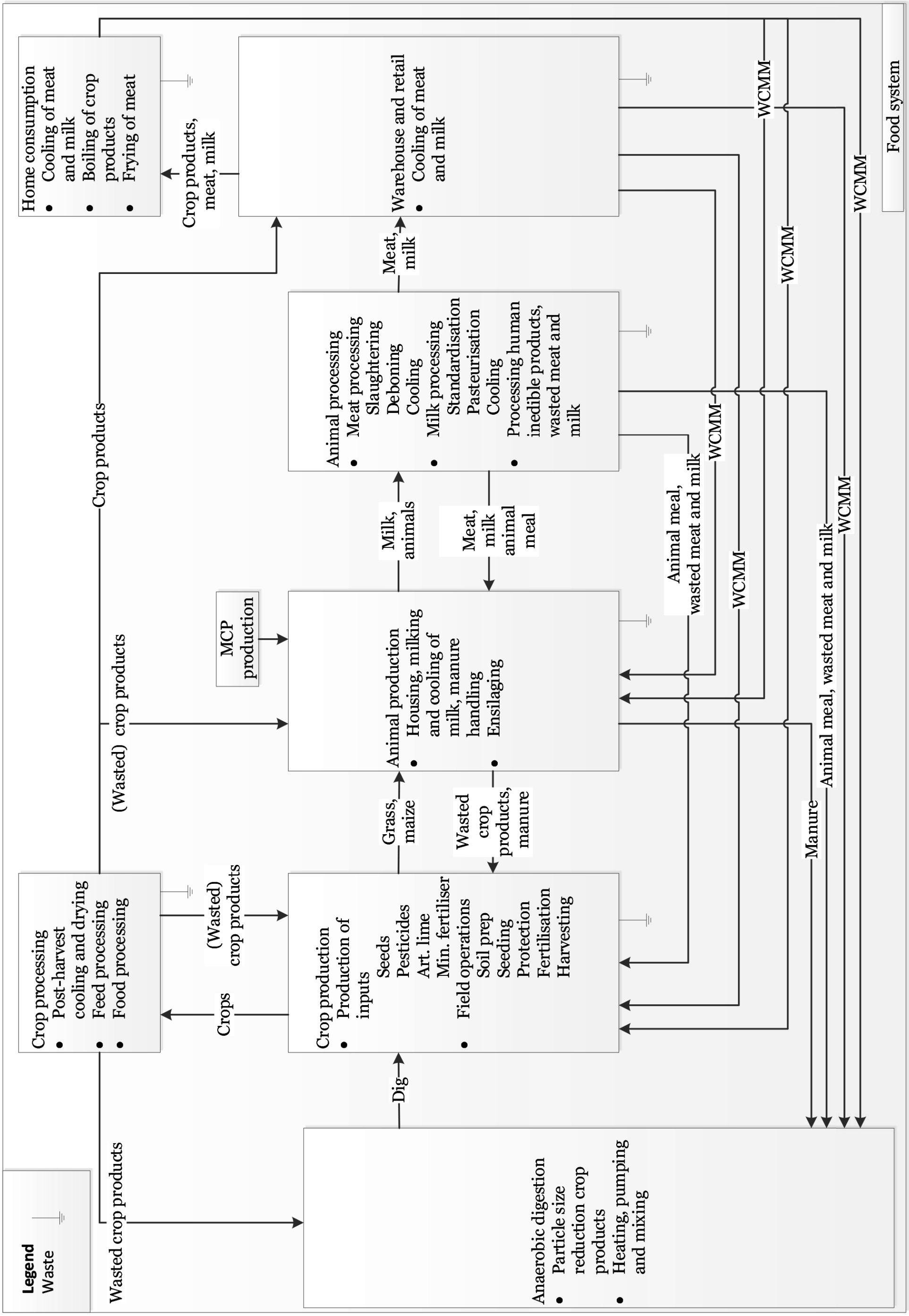

Figure 1. Included activities in the food system. Arrows indicate transport. Note: Dig = digestate, MCP = monocalcium phosphate, WCMM = wasted crop products, meat and milk

2.3 Crop production

Our model included five food crops cultivated in rotations, i.e. wheat, potato, sugar beet, rapeseed and brown bean, and two feed crops for dairy cattle, i.e. silage maize and grass (Van Kernebeek et al., 2016). All above ground biomass could potentially be harvested, except for wheat and maize stubble, potato haulms, sugar beet leaves and bean straw. We assumed that these crop residues stay behind on the field as source of soil organic carbon and nutrients. Crops were fertilised with nitrogen (N), phosphorus (P) and potassium (K) at levels that are normally sufficient to achieve the yield levels presented by Van Kernebeek et al. (2016). This implies that the amount of nutrients that are lost from one ha, whether that is through harvesting of crops, or through (unavoidable) losses via leaching, run-off, ammonia volatilisation or denitrification, were compensated by inputs (Supplementary Material S2). The various sources of inputs included animal manure, crop residues, crop products recycled as fertiliser, wasted ASF, animal meal, digestate and mineral fertiliser (Supplementary Material S2).

We accounted for energy use for the production of inputs, including seeds, pesticides, artificial lime, and mineral fertiliser (Figure 1). We assumed fixed quantities and, hence, fixed energy use for the production of seeds and pesticides per ha of crop rotation (Supplementary material S3). The required quantities, and, hence, energy use for the production of artificial lime and mineral fertiliser were variable, and depended on the availability of lime and N, P and K by other sources (Supplementary material S3).

We accounted for energy use during field operations, i.e. soil preparation, seeding, crop protection, fertilisation and harvesting (Figure 1). We assumed fixed field operations for soil preparation, seeding, crop protection and harvesting per ha of crop rotation, including production and maintenance of machinery for field operations (Vellinga et al., 2013) (Supplementary material S3). As a result, energy use for these operations is also fixed (Supplementary material S3). Field operations for fertilisation of crops were variable, and dependent on the types of fertilisers applied to the land (Supplementary Material S2 and S3).

2.4 Crop processing

We assumed that harvested crops (i.e. wheat grain, potatoes, sugar beets, rapeseeds and beans) were cooled and dried during post-harvest storage for conservation purposes. Storage of wheat grain requires 169 MJ ton^{-1} DM, whereas storage of rapeseed requires 178 MJ ton^{-1} DM (Williams et al., 2010). We assumed that energy use for storage of sugar beets and beans are equal to that for wheat grain. We furthermore assumed that storage of potatoes requires 3143 MJ ton^{-1} DM (Williams et al., 2010).

We assumed that all crop products, except for grass and maize, were industrially processed. We distinguished between feed and food processing. We defined feed processing as an activity resulting in only feed for animals. Included processes were grinding, chopping and cutting of crop products to reduce particle size (Supplementary material S4). We also included heating of potatoes fed to pigs, and heating of beans fed to both cows and pigs (Van der Poel et al., 1991; Whittemore et al., 1975; Yu et al., 1998) (Supplementary material S4). Moreover, we included pelleting of 85% of processed feed fed to pigs and cows (dry matter basis) Supplementary material S4) (Bikker, 2016; Dijkstra, 2016). We find it reasonable to include the above described feed processes, as they are assumed to increase digestibility of feed and growth performance of animals (Liu et al., 2013; Wondra et al., 1995; Woyengo et al., 2014). We defined food processing as an activity resulting in at least one human edible crop product. Included processes were milling of wheat grain, peeling of potatoes, sugar beet processing and crushing of rapeseed (Supplementary material S1 in Van Kernebeek et al. (2018)), and Supplementary material S4). Some output products from food processing can also be consumed by animals (Supplementary material S1 in Van Kernebeek et al. (2018)). Dry milling of wheat, for example, resulted in wheat middlings (which we assumed available for animals only), and wheat germ, bran and flour (which we assumed edible for humans and animals) (Supplementary material S1 in Van Kernebeek et al. (2018)). For each food process, we assured N, P and K balances between the input crop product and the sum of output crop products (Supplementary Material S5).

2.5 Animal production

Our model included dairy production as representative for ruminants, and pig production as representative for monogastrics (Van Kernebeek et al., 2016). We modelled dairy and pig production based on animal production units (PUs). One pig PU consisted of 3.3 fattening pigs, 0.12 sows and 0.07 gilts (Van Kernebeek et al., 2016). One cow PU consisted of a dairy cow and its replacement stock, i.e. 0.31 replacement heifers aged 1-2 years, and 0.34 replacement calves aged 0-1 year (Van Kernebeek et al., 2016). Moreover, one cow PU provided 8120 kg unprocessed milk (Van Kernebeek et al., 2016). Feed requirements for pigs and cows were described by Van Kernebeek et al. (2016) and Van Kernebeek et al. (2018). These feed requirements were met by consumption of (recycled) crop products (Supplementary Material S1 in Van Kernebeek et al. (2018)), recycled waste of milk and meat, recycled animal meal, and monocalcium phosphate (MCP) (Van Kernebeek et al., 2018) (Figure 1). The assumed nutritional values of these feed ingredients, as well as feed restrictions, are presented in Van Kernebeek et al. (2016), Van Kernebeek et al. (2018), and Supplementary Material S6. To determine N, P and K concentrations in manure, we computed nutrient intake and nutrient retention in animals, and assumed fixed quantities of manure production per PU per year (Supplementary material S6 and S7). We assumed non-grazing systems, and, hence, all manure was produced inside (Van Kernebeek et al., 2018).

Housing of pigs required 761 MJ primary energy per pig PU per year for heating, ventilation, lighting, mechanical feeding and manure handling (Blanken et al., 2017). Housing of cows, including milking, cooling of the milk tank, and manure handling, required 3641 MJ primary energy per cow PU per year (Blanken et al., 2017). In addition, we accounted for on-farm energy costs related to ensilaging of grass (744 MJ primary energy ha^{-1}) and maize (710 MJ primary energy ha^{-1}) (Vellinga et al., 2013).

2.6 Animal processing

Meat processing included slaughtering, removal of hides, hair and offal, cutting, deboning and cooling (Supplementary material S4). One slaughtered pig PU yielded 171 kg pork (of which part will be wasted, see section on waste), and 208 kg human inedible products (Hejnfelt and Angelidaki, 2009; Van Kernebeek et al., 2016). One slaughtered cow PU provided 74 kg beef (of which part will be wasted, see section on waste), and 161 kg human inedible products (Van Kernebeek et al., 2016). We assumed that human inedible animal products, and wasted meat and milk, require processing before being recycled as animal feed or crop fertiliser (Supplementary material S4). Once human inedible animal products were processed, we refer to it as animal meal. Milk processing included standardisation, pasteurisation and cooling (Supplementary material S4). Standardisation of unprocessed milk resulted in 8502 kg fat-and-protein-corrected-milk (FPCM) per cow PU per year (Van Kernebeek et al., 2016). We included pasteurisation as pasteurised milk is the most dominant type of milk consumed in the Netherlands (Productschap Zuivel, 2010).

2.7 MCP production

Monocalcium phosphate (MCP) was potentially available as feed additive for pigs and cows to better enable a positive P balance in feed (Van Kernebeek et al., 2018). Production of MCP required 60.7 MJ/kg P (Nielsen and Wenzel, 2007).

2.8 Warehouses and retail

We assumed that milk and meat were cooled in warehouse and retail for conservation purposes (Figure 1). We accounted for 0.432 MJ kg^{-1} fresh matter (FM) for cooling of milk in warehouses and retail (Broekema and Kramer, 2014). We assumed equal energy use for cooling of meat (MJ kg^{-1} FM). We furthermore assumed that crop products were stored at ambient temperatures without requiring energy.

2.9 Home consumption

We assumed that milk was cooled in the refrigerator at home for conservation purposes, requiring 0.17 MJ kg^{-1} FM, and we took equal energy use for cooling of meat (Figure 1) (Broekema and Kramer, 2014). We furthermore assumed that crop products were stored at ambient temperatures without requiring additional energy (Foster et al., 2006).

We assumed that wheat products, potatoes, beans and meat require home cooking before consumption. Energy use for home cooking of food vary largely, depending on e.g. portion size and type of cooking facility (Braschkat et al., 2003; Carlsson-Kanyama and Boström-Carlsson, 2001; Foster et al., 2006; Grönroos et al., 2006). We assumed generic energy use of 3.5 MJ kg^{-1} FM for boiling of crop products, and 7.5 MJ kg^{-1} FM for frying of meat (Foster et al., 2006). We further assumed that wasting of food at home occurred prior to food preparation, based on Van Westerhoven (2013).

2.10 Anaerobic digestion

We distinguished two types of digesters, i.e. a high-productive (HP) and a low-productive (LP) digester. We made this distinction to account for the relation between the carbon to nitrogen (C:N) ratio of the input mixture and biogas yield (Wang et al., 2014). Biogas yield is optimal for an input mixture with C:N ratio between 20 and 30 (Wang et al., 2014). For the HP digester, therefore, we restricted the C:N ratio of the input mixture to this range, whereas for the LP digester we did not restrict the C:N ratio of the input mixture. We assumed that the biogas yield of the LP digester was 50% of that of the HP digester, based on Wang et al. (2014). Energy recovery from digestion of crop products and meat and milk was computed from their digestible carbohydrates, crude fat and crude protein (Supplementary Material S8). In the HP digester, anaerobic digestion yielded 886 litre biogas (with 50% methane content) per kg of digested carbohydrates, 1535 litre biogas (with 70% methane content) per kg of digested fat, and 587 litre biogas (with 84% methane content) per kg of digested protein (Chandra et al., 2012). The energy density of methane gas equalled 36 MJ per m^3 of methane (Banks, 2009). Energy recovery from anaerobic digestion of crops, meat and milk, manure and slaughterhouse waste in a HP digester is presented in Supplementary Material S8. We assumed that sugar factory lime could not be allocated to the digester. We furthermore assumed that 62% of the produced energy was actually used (RVO, 2013). We refer to this as "recovered energy". Twelve percent of the recovered energy was used by the digester for processes such as particle size reduction, heating, pumping and mixing of biomass (RVO, 2013).

In addition to biogas, the anaerobic digestion also yielded digestate, a residual that can be applied to crops as a source of fertiliser (Supplementary material S2). We assumed that all digestate was applied to crops without separation of the dried and liquid fraction, as is most common in The Netherlands (RVO, 2013). The amount (in ton dry matter) of digestate was computed from the organic matter fraction, and one minus the digestibility rate of organic matter, plus ash content. We assumed that all N, P and K entering the digester ended up in the digestate (Jones and Salter, 2013; Möller, 2015).

2.11 Transport

To account for energy use during transport, we distinguished three transport distances, i.e. 20 km, 35 km and 100 km: Assuming a high density of crop production, animal production and anaerobic digesters in the Netherlands, we accounted for a 20 km distance between crop and animal production, between animal processing and crop production, and between any step in the chain and an anaerobic digester, based on Zwart et al. (2006). We also assumed a distance of 20 km between human consumption and crop and animal production for recycled waste of crop products, meat and milk. We furthermore accounted for a distance of 35 km between animal production and animal processing (Head et al., 2011). Moreover, we took a distance of 100 km between crop production and crop processing, between crop processing and wholesale, and between animal processing and wholesale. We also assumed a distance of 100 km to transport crop inputs to crop production, of MCP to animal production, and to transport any waste to the landfill. We assumed that transport required 0.94 MJ/tonkm (BioGrace, 2011). We did not include energy use for transport between wholesale and retail, as this does not influence the relation between consumption of crop versus animal products.

3. Results

The results section includes three parts. We first present energy use for the main activities in the food system in the baseline situation without digester (Baseline_no_AD). In this situation, the potential of recovering energy through anaerobic digestion of biomass is not used. Therefore, in the subsequent part, we explore the potential of recovering energy through anaerobic digestion, assuming recycling of 50% of waste in the baseline situation with anaerobic digester (Baseline_AD). In both baseline situations, food is wasted throughout the system. In the third part of this section, we therefore discuss the consequences of preventing food waste on the overall energy input into a food system in a situation without anaerobic digestion (WPREV_no_AD) and in a situation with anaerobic digestion (WPREV_AD).

3.1 Baseline situation without anaerobic digestion (Baseline_no_AD)

In the baseline situation without anaerobic digestion (Baseline_no_AD), energy input into the food system is defined only by the energy use for the various activities in the food system, as in this situation no energy is recovered through anaerobic digestion (Figure 2).

Energy use by the food system slightly decreased as %PA increased from 0 to 15%, and subsequently increased (Figure 2). To unravel the observed relation between %PA and energy use, we present key parameters describing energy use in the food system for varying %PA in Table 2 and Supplementary material S9. We first explain the difference in energy use between a diet with 0% PA (i.e. a vegan diet) and a diet with 15% PA. Second, we explain the difference in energy use between a diet with 15% PA and one with 60% PA.

The slight decrease in energy use for a diet that contained 0% PA to a diet that contained 15% PA is the result of two opposite effects (Table 2). On the one hand, a diet with 15% PA required less land and less crop biomass, contributing to lower energy use for crop production, cooling and drying of crop products, food processing, and boiling of crop products than a diet with 0% PA (Figure 3). The lower land use of a diet with 15% PA results from the conversion of human inedible crop products (HIE) and recycled human edible crop products by animals into human edible protein (Supplementary material S10 and S11). These animals, therefore, provide ASF with no or little additional arable land (Van Kernebeek et al., 2016). Compared to a vegan diet, a diet with ASF from animals fed predominantly with leftovers, thus, needs less crop land, contributing to lower energy use from crop cultivation Van Kernebeek et al. (2016). On the other hand, a diet with 15% PA required more animal production, resulting in higher energy use for feed production, animal production and processing, MCP production, cold storage of meat and milk in warehouse and retail, cold storage of meat and milk at home, and frying of meat (Supplementary material S9). Overall, a diet with 15% PA had slightly lower energy use than a diet with 0% PA.

Table 2. Energy use (TJ) by various activities in the food system in the baseline situation without digestion (Baseline_no_AD) for diets varying in percentage of protein from animals (%PA).

	% PA		
	0	**15**	**60**
Crop production	15887	14304	18747
Crop processing	12526	12313	15063
Cooling and drying	5215	4865	5185
Feed processing	0	225	435
Food processing	7311	7222	9443
Animal production	0	756	4211
MCP production	0	1	0
Animal processing	0	1355	7060
Cold storage meat & milk in warehouse and retail	0	742	3880
Home consumption	15009	14010	13286
Cold storage meat & milk	0	289	1512
Boiling crop products	15009	13617	11298
Frying meat	0	104	476
Anaerobic digester	0	0	0
Transport	2521	2451	4018
Total energy use	45943	45932	66265

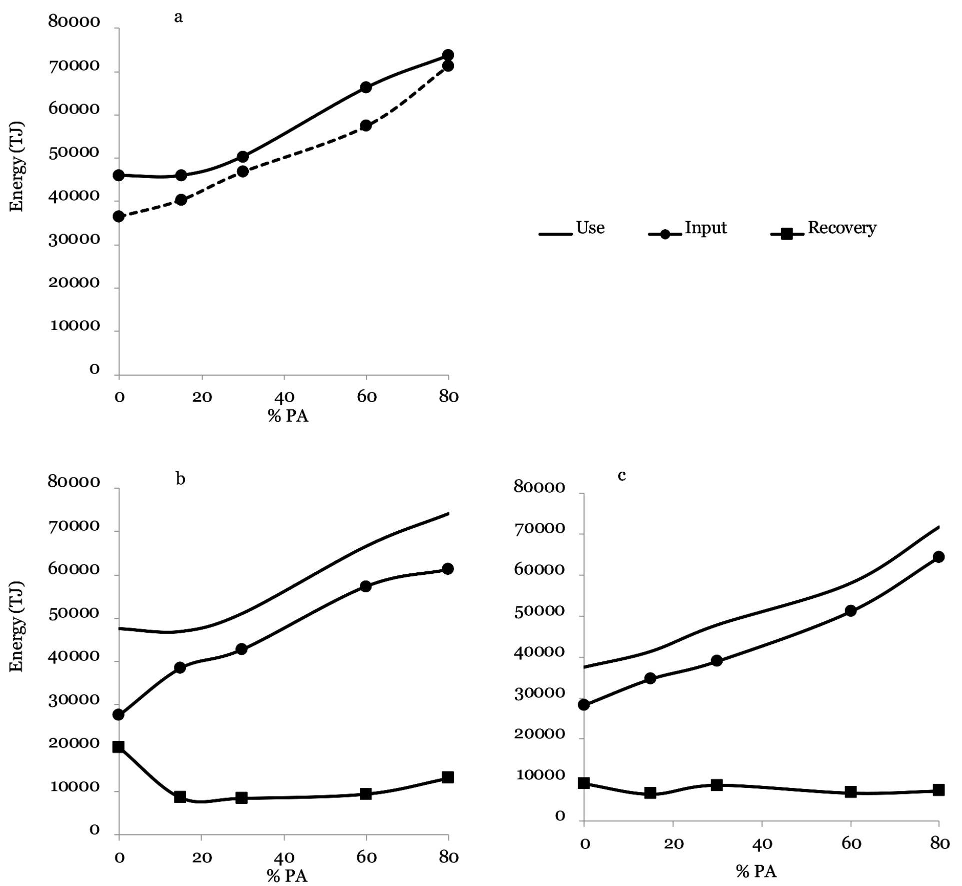

Figure 2. Energy use, input and recovery (TJ) in (a) Baseline_no_AD (solid line) and WPREV_no_AD (dashed line), (b) Baseline_AD) and (c) WPREV_AD, against diets varying in percentage of protein from animals (%PA). Note: in (a) energy use is equal to energy input.

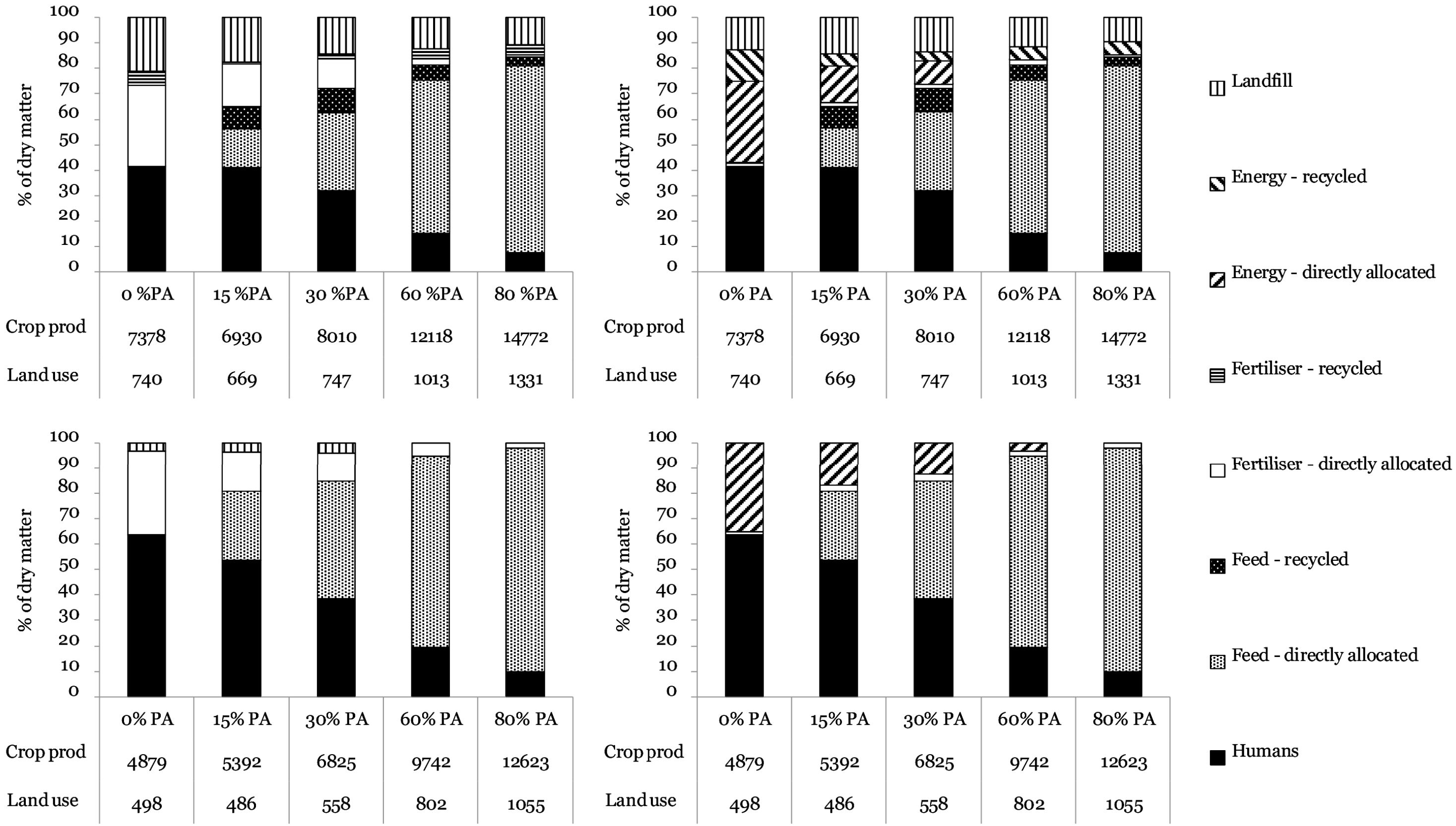

Figure 3. Allocation of crop products (% of dry matter), either direct or through recycling, as food to humans, as feed to animals, as fertiliser to land, as source of energy to the digester or as waste to landfill, for diets varying in percentage of protein from animals (%PA), and crop production (1000 ton DM) and land use (1000 ha) in Baseline_no_AD (upper left), Baseline_AD (upper right), WPREV_no_AD (lower left), and WPREV_AD (lower right).

When increasing PA from 15 to 60%, energy use increased as a result of two opposite effects (Table 2). On the one hand, a diet with 60% PA required more land and more crop biomass, contributing to higher energy use for crop production and crop processing (Figure 3). We expected this increase in land use and crop production as from 15% PA upwards, human inedible crop products and crop waste were not sufficiently available to sustain animals, and therefore additional land was used to produce crops for animals. Furthermore, energy use for animal production, animal processing, cold storage of meat and milk, and frying of meat increased following logically from the increased consumption of animal products (Supplementary material S12). On the other hand, less energy was required for boiling of crop products, as less crop products which need boiling were consumed (Supplementary material S12).

We moreover observe that in this baseline situation more than 50% of the wasted crop products are allocated to the landfill, rather than being recycled as fertiliser or as feed (Supplementary material S10). This is especially the case for diets with 0 and 15% PA. Crop products that have a low dry matter content, i.e. potato peel and sugar beet pulp, and crop products that do not contain N, P and K, i.e. sugar and rapeseed oil, are not recycled as fertiliser, implying that applying these products on the land as a substitute for mineral fertiliser is not energy efficient. Furthermore, at 15% PA, not all wheat straw and rapeseed straw is recycled as feed, as the utilisation by animals of this fibrous feed ingredient is restricted by digestibility and intake capacity.

In Baseline_no_AD, cows appear more energy-efficient than pigs (Supplementary material S12). Although husbandry of cows required more energy per PU than husbandry of pigs, especially during milking, total energy use per unit of edible protein appeared lower for milk and associated beef than for pork. As a result, milk and associated beef were the main sources of animal protein in the human diet (Supplementary material S12).

3.2 The effect of anaerobic digestion in the baseline situation (Baseline_AD)

Energy input into the food system in Baseline_AD is the net result of energy use and energy recovery (Figure 2). Energy input was 14 – 40% lower compared to Baseline_no_AD. Energy input increased with increasing %PA in the human diet, from about 28,000 TJ for a diet with 0% PA to slightly over 60,000 TJ for a diet with 80% PA. To unravel the increase in energy input, we will discuss energy use and energy recovery in the following sections.

Energy use in Baseline_AD is equal to that in Baseline_no_AD for most of the activities in the food system (Supplementary material S9). This can be explained from the fact that we assumed the same land use for the baseline situation with and without an AD (i.e. we fixed the land use), resulting in the same crop rotations, and, therefore, equal amounts and types of crop products, processing volumes, and number of animals (Figure 3 and Supplementary material S13). Small differences in energy use between Baseline_AD and Baseline_no_AD can be explained from the substitution of recycled crop products as fertiliser, by digestate. As digestate had a higher N-fertiliser replacement value compared to crop products, a higher percentage of crop products, and, hence, nutrients, was recycled and less mineral fertiliser was used in Baseline_AD (Supplementary material S10 and S13). Moreover, in Baseline_AD about 1,000 – 2,500 TJ was used for the process of anaerobic digestion. As a net effect, energy use was up to four percent higher in Baseline_AD compared to Baseline_no_AD (Supplementary material S9).

The potential to recover energy through anaerobic digestion in Baseline_AD is presented in Figure 2 (for more details see Figure 4 and Supplementary material S14). We first discuss energy recovery for a diet containing 0% PA, and subsequently discuss energy recovery for diets containing higher percentages of PA. At 0% PA, about 20,000 TJ of energy is recovered from crop products (Figure 4). Approximately 60% of the recovered energy originated from human-inedible (HIE) crop products that were directly allocated to the digester (Figure 4). These HIE crop products are co-produced with HE crop products destined for humans, such as wheat straw co-produced with wheat. Large shares of HIE crop products did not get wasted, and these non-wasted products were therefore directly allocated to the digester (Supplementary material S10). The remainder of recovered energy originated predominantly from human-edible (HE) crop products that were recycled after being wasted along the food chain (Figure 4).

As %PA increased from 0 to 15% PA, energy recovery from crop products decreased to about 6,000 TJ (Figure 4). This decrease is due to the competition for crop products between digester and animals. At 15% PA, animals demand large shares of HIE and recycled HE crop products (Supplementary material S10). The energy recovery from manure of approximately 2,000 TJ, however, was larger at 15% PA, because at 0% PA there are no animals in the food system (for more details see Figure 4; and Supplementary material S14).

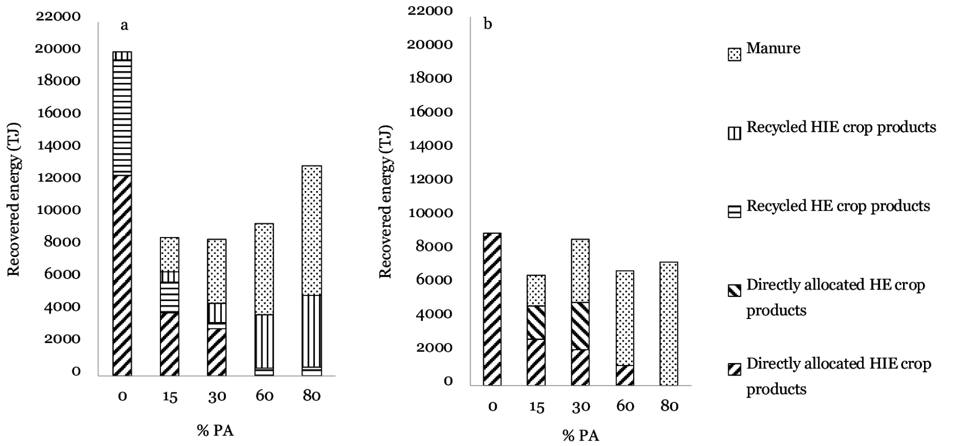

Figure 4. Energy recovery (TJ) from crop products and manure in a) the Baseline situation with anaerobic digester (Baseline_AD) and in b) the alternative situation where waste is prevented (WPREV_AD). Crop products are allocated to the digester either direct or through recycling. HIE = human inedible, HE = human edible. Energy recovery from recycled meat, milk and animal meal was negligible and not included in the figure.

As %PA increased from 15% up to 60%, energy recovery from crop products further decreased (Figure 4). This decrease is the result of two opposite effects. On the one hand, energy recovery decreased from directly allocated HIE and recycled HE crop products, because of an increasing demand for these crop products by animals (Figure 3 and Supplementary material S10). On the other hand, energy recovery increased from recycled HIE crop products. This increase is due to increased volumes of wasted HIE crop products in animal husbandry as a result of conservation and feeding losses. As we assumed that these losses cannot be recycled as feed, these wastes are being recycled as source of energy through anaerobic digestion. Overall, as %PA increased from 15 to 60%, the increase in energy recovery from manure compensates the decrease in energy recovery from crop products (i.e. by ca 1,200 TJ) (Figure 4).

3.3 Alternative situations where waste is prevented

Energy use by the food system in situations WPREV_no_AD and WPREV_AD increased with increasing %PA (Figure 2). Energy use did, hence, not follow the same trend as land use, which decreased as %PA increased from 0 to 15% PA (Figure 3). This can be explained as follows. We first minimised land use. Land use decreased as %PA increased from 0 to 15% (Figure 3). However, the crop rotations that were selected in the food system at 15% PA, required more energy per hectare than the crop rotations that were selected in the food system at 0% PA. Moreover, the crops harvested at 15% PA required more energy in crop processing and transport.

In the alternative situations where waste is prevented (WPREV_no_AD and WPREV_AD) energy use by the food system is 3 - 20% lower compared to Baseline_no_AD (Figure 2, and see Supplementary material S9 for more details). The lower energy use is the net result of two effects of waste prevention. First, as no waste occurred, less crops and ASF needed to be produced to meet nutritional requirements of the human population compared to the baseline, and less crops needed to be produced to meet nutritional requirements of animals. As a result, land use was 20 – 32% lower in WPREV_no_AD compared to Baseline_no_AD, contributing to lower energy use in crop production (Figure 3 and Supplementary material S9). Moreover, as fewer animals were required, energy use in animal production was also lower (Supplementary material S9 and 13). Second, in situations where waste is prevented other ratios and/or types of crop rotations were selected than in the baseline situation. This implies that waste prevention affects the land use efficiency of crop rotations, because in the baseline situation, waste percentages are different for different crops (and thus crop rotations). As a result, the composition of the human diet was different in situations where waste was prevented compared to the baseline, affecting energy use for crop production and processing, and human consumption (Supplementary material S9 and S12).

We note that in WPREV_no_AD for diets containing 0, 15 and 30% PA, up to about 4% percent of the crop products were allocated to the landfill, rather than being used as food, feed or fertiliser (Figure 3 and Supplementary material S10). At 0% PA, these products refer to potato peels and sugar beet pulp. These products have a low dry matter content, implying that applying these wet products to the land is not energy efficient. At 15 and 30% PA, the product allocated to landfill is sugar. This sugar is not used as food or feed, implying that this product is produced in abundance. Sugar is not used as fertiliser, as it does not contain N, P and K, and is, therefore, no substitute for mineral fertilier.

The potential to recover energy through anaerobic digestion in the alternative situation WPREV_AD is presented in Figure 2 and 4 (see Supplementary material S14 for more details). We first discuss energy recovery for a diet containing 0% PA, and subsequently discuss energy recovery for diets containing higher percentages of PA.

At 0% PA, about 9,000TJ of energy was recovered from crop products, originating fully from HIE crop products that were directly allocated to the digester (Figure 4 and Supplementary material S14). These HIE crop products are co-produced with

HE crop products destined for humans. Energy recovery was approximately 50% lower compared to Baseline_AD (Figure 4). This can be explained from two factors. First, as no wastes occurred in WPREV_AD, no energy was recovered from recycled crop products. In Baseline_AD, recycled crop products contributed about 7,400 TJ to recovered energy (Figure 4). Second, as no waste occurred in WPREV_AD, less crop products needed to be produced to feed the human population, and, hence, less directly allocated HIE crop products were available for energy recovery compared to Baseline_AD (Supplementary material S10).

As %PA increased, energy recovery from directly allocated HIE crop products decreased due to increased competition for these crop products between energy and feed (Supplementary material S10). At 15 and 30% PA, energy was also recovered from HE crop products (i.e. sugar). At these percentages of PA, the maximum intake level of total sugar in the human diet is reached, and, hence, part of the sugar is available for energy recovery. Furthermore, as %PA increased, energy recovery from manure also increased from approximately 1,800 TJ at 15% PA to approximately 7,400 TJ at 80% PA (Figure 4; and Supplementary material S14 for more details).

4. Discussion

4.1 Using a modelling approach to assess reduction of energy input to the food system

We quantified energy use, energy recovery and energy input in a hypothetical food system using an optimisation approach. The total energy use of the food system in our baseline situation ranged between about 45 and 74 PJ, depending on the %PA in the diet. This is equivalent to about 2.6 – 4.1 GJ per capita per year. As we used an optimisation (minimisation) approach, energy use for our diets was lower compared with estimates found in other studies on (representative) European diets, which ranged between 6.9 and 21 GJ per capita per year (Carlsson-Kanyama et al., 2003; Monforti-Ferrario et al., 2015; Muñoz et al., 2010). Furthermore, estimates on energy use were higher in these studies as they included drinks, tropical fruits, sweets and snacks, which require a considerable amount of energy, included diets with higher calorie intake, or applied other system boundaries which included waste treatment.

In our modelling approach, we optimised energy input for a baseline situation, and an alternative situation in which waste was prevented. Within both situations, we studied the effects of anaerobic digestion. These 2x2 situations were established as follows: To avoid the cultivation of crop biomass exclusively for bio-energy production (EU, 2018), we first minimised land use for food self-sufficiency in the baseline situation and in the alternative situation, both without anaerobic digestion. In each of these situations we subsequently assessed the impact of introducing anaerobic digestion, at different levels of animal-source food in the human diet. The consequence of optimising land use for the baseline and alternative situation separately and before optimising energy input, is that we cannot compare the results of the baseline situations with those of the alternative situations. This can be explained from the following: Waste prevention changed the mutual differences in land use efficiency of selected crop rotations and crops along the food system, as waste percentages differed for different crops in the baseline. This resulted in selection of other ratios and/or types of crop rotations in the alternative situation where waste was prevented than in the baseline. This selection of other crop rotations between the baseline situation and alternative situation hinders a direct comparison of their energy inputs. Despite our aim to minimise energy input for food self-sufficiency, in some situations some crop products were produced in surplus. This was an artefact of our selection of crop rotations, which limited the possibility to provide human edible proteins and calories precisely in the required ratio of 57 grams of protein and 2000 kcal. Abundancy of crop production can be avoided by inclusion of more (single-) crop rotations, which provides the model with more flexibility to produce precisely the right ratio of protein and energy.

4.2 Strategies to reduce energy input to the food system

Our modelling exercise of a hypothetical food system provided innovative insights into the potential of preventing waste, recycling waste as animal feed or fertiliser, and recovering waste via anaerobic digestion to reduce energy input to the food system. In our baseline situation in which half of wasted crop and animal products could be recycled as feed or fertiliser a diet with 15% PA had the lowest energy input. By introducing anaerobic digestion to the baseline situation, energy input was reduced by 14-40%, depending on the %PA. This reduction potential was largest for a diet with 0% PA (40% reduction) This relatively strong reduction potential at 0% PA can be explained as follows. First, when producing a vegan diet, a substantial amount of food waste is generated because humans consume only food crops, and waste percentages of food crops are high compared to waste percentages of feed crops. Second, in case of a vegan diet, animals do not compete for food waste and human inedible crop products with anaerobic digestion, and these products could thus all be recovered as bio-energy. As a result, in the baseline situation with anaerobic digestion, a vegan diet had a lower energy input than diets with animal protein. Furthermore, energy recovery slightly increased from 30 to 80% PA as a result of increased production of manure and feed waste.

In the alternative situation in which waste was prevented, energy use was 3 - 20% (depending on the %PA) lower than in the baseline situation, both without anaerobic digestion (Fig. 2a). The potential of waste prevention to reduce energy use was limited as the crop rotations and crops selected in the alternative situation where waste was prevented appeared less energy efficient along the food chain compared to those selected in the baseline situation. Moreover, when waste was prevented, energy use was lower for a vegan diet compared with diets with animal protein. This was again caused by the lower energy-efficiency of the crop rotations and crops selected in case of a vegan diet than those selected with diets with animal protein. By introducing anaerobic digestion to the situation where waste was prevented, energy input was reduced by 10 – 23%, depending on the %PA (Fig. 2a (dashed line) compared with Fig 2c). This reduction potential was largest (i.e. 23%) for a vegan diet, as in this situation no competition for human inedible crop products between feed and bio-energy occurred, and these products could thus be exclusively recovered as bio-energy.

We demonstrated that, to reduce energy input to a food system, it is essential to account for the interaction effects of waste prevention, anaerobic digestion and dietary shifts. We conclude that energy input into the food system is reduced by anaerobic digestion and waste prevention as single interventions. We also conclude that anaerobic digestion reduced energy input both in a situation with and without waste prevention. Moreover, the effect of anaerobic digestion was strongest in those situations where substantial amounts of human inedible crop products, and food and feed waste were produced, and where animals did not compete for these resources with anaerobic digestion (Fig. 2b). We furthermore clearly show that in situations with anaerobic digestion and/or waste prevention, energy input increases with increasing %PA.

We demonstrated the competition between food, feed, fertiliser and bio-energy in a food system. As we minimised land use, crop products that could not be consumed by humans were allocated to animals to meet the requirement for feed. Therefore, as animal production increased with increasing %PA, less crop biomass was available as fertiliser (in situations without anaerobic digestion) or as bio-energy source (in situations with anaerobic digestion), with the exception of feed waste. We furthermore demonstrated that when adding a digester to either the baseline situation or the alternative situation with waste prevention, crop biomass that was previously recycled as fertiliser was now allocated to the digester, yielding both bio-energy and digestate. In that situation, digestate substituted crop products as fertiliser. As digestate had higher N-fertiliser replacement value than crop products, less mineral fertiliser was required in a situation with anaerobic digestion than without anaerobic digestion. This effect was limited, however, as we minimised land use, and, hence, the availability of crop products as a source of fertiliser or bio-energy was limited in the first place.

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Supplementary Material S1 Primary energy

When quantifying energy input, we accounted for efficiencies in the production of energy carriers. We assumed that diesel was produced with an efficiency of 86.2% (i.e. 1.16 MJ fossil fuel was required to produce 1 MJ diesel for end-use) (BioGrace, 2011). We furthermore assumed a 44% efficiency in the production of electricity from fossil fuel (Seebregts and Volkers, 2005) and a 88.6% efficiency in the production of natural gas from fossil fuel (BioGrace, 2011).

Supplementary Material S2 Crop fertilisation

Total amount of P required per ha for each crop rotation was computed from the P content of all crops in that rotation and assumed unavoidable losses through leaching and run-off (Eq. 1) (Table S1). Wheat and maize stubble, potato haulms, sugar beet leaves and bean straw were not included, as we assumed that these parts of the crops stayed behind on the field as a source of P for the subsequent crop.

$$TR_{i,l} = \sum_{j=1} Y_{j,i,l} \times DM_j \times Pcont_j + UL_{i,l} \qquad \text{Eq. 1}$$

Where $TR_{i,l}$ is the total requirement of P per ha (in kg ha^{-1}), for crop rotation (i) on land type (l), based on the sum of all harvested products (j) from that rotation, including main and co-products; Y is the fresh matter yield of a harvested product (ton ha^{-1}) (Online resource I in Van Kernebeek et al., (2016)), DM is the dry matter content of a harvested product (Online Source I in Van Kernebeek et al., (2016)), Pcont is the nutrient content of a harvested product (kg ton^{-1} DM) (PDV, 2011), and UL is the unavoidable P loss (kg ha^{-1}) through leaching and run-off, which was assumed 2.2 kg P ha^{-1} on all soil types and crop rotations (Rijksoverheid, 2014).

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Total amount of P required per ha for each crop rotation was provided by variable sources according to Eq. 2. For all recycled and organic fertiliser sources we assumed a P fertiliser replacement value relative to mineral fertiliser of 100% (De Haan and Van Geel, 2013; Severin et al., 2014).

$$\begin{aligned} TR_{i,l} = {} & MF_{i,l} + \sum_{a=1}^{2}\sum_{b=1}^{41} Man_{i,l,a,b} \times ManConc_{a,b} + \sum_{j=1} VCR_{i,l,j} \times Nutrcont_j \\ & + \sum_{k=1} Crp_{i,l,k} \times Nutrcont_k + HumanexcProcWater_{i,l} + WasteAnimal_{i,l} \qquad \text{Eq. 2} \\ & + Animalmeal_{i,l} + \sum_{f=2}^{2} \mathit{Digestate}_{i,l,f} \end{aligned}$$

Where $TR_{i,l}$ is the total fertiliser requirement of P for crop rotation (i) and soil type (l) (kg ha^{-1}), $MF_{i,l}$ is the amount of P from mineral fertiliser (triple superphosphate) (kg ha^{-1}). $Man_{i,l,a,b}$ is the volume of applied manure (ton DM) of manure type (b) produced in animal production system type (a) (See Supplementary Material S7), $ManConc_{a,b}$ is the P concentration in manure (kg/ton DM), which differed per manure type and animal production system type (See Supplementary Material S7), $VCR_{i,l,j}$ is the amount of variable crop residue (defined here as co-products that could either be left on the field or be harvested as feed, i.e. wheat straw, sugar beet tops and tails, and rapeseed straw) (ton DM) (j) left for crop rotation (i) on soil type (l). $Nutrcont_{j,}$ is the P content (kg ton^{-1} DM) in variable crop residue (j), $Crp_{i,l,k}$ is the volume of crop product (k) returned back to the land (ton DM ha^{-1}), $Nutrcont_k$ is the P content of crop product (k) returned back to the land. $WasteAnimal_{i,l}$ is the amount of P (kg ha^{-1}) from recycled waste of ASF, and $Animalmeal_{i,l}$ is the amount of P (kg ha^{-1}) from recycled animal meal. $Digestate_{i,l,f}$ is the amount of P (kg ha^{-1}) from digestate produced in digestate type f. We did not allow fertilisation of grassland by crop residues or crop products returned back to the land.

Crop fertilisation with potassium (K) (Table S1) was modelled using the same method as described for phosphorus (P), with one exceptions. This exception is that we assumed atmospheric deposition of 5.3 kg K (LEI and CBS, 2012). Deposition of K was considered a fixed source for each hectare (ha) of crop rotation and soil type. Therefore, deposited K replaced the requirement of K from variable sources. Fertilisation with nitrogen (N) (Table S1) was modelled in a different way as we did for P and K fertilisation. For N, we assumed economic optimum fertilisation rates (De Haan and Van Geel, 2013; Hoeks et al., 2012). These fertilisation rates account for two important assumptions. The first assumption is that fixed amounts of N are provided via atmospheric deposition and biological fixation. As these fixed sources are already accounted for, we assumed that economic optimum fertilisation rates had to be provided by variable sources only. The second assumption is that N is lost due to nitrate leaching, ammonia volatilisation and denitrification, and that these N losses are compensated by accounting for N fertiliser replacement values (NFRV) of the various variable sources. The NFRV of variable sources relative to mineral fertiliser were: 60% on cropland and 90% on grassland for both cattle and pig slurry (Gutser et al., 2005; Schröder et al., 2007); 50% for crop residues returned back to the land (Gutser et al., 2005); 40% for crop products returned back to the land (Based on De Haan and Van Geel (2013), 60% for animal products returned back to the land (Gutser et al., 2005), 60% on cropland and 90% on grassland for digestate (Lemke et al., 2012; Schröder et al., 2007). To account for N-loss from volatilisation from manure in stable and storage, we converted gross excreted N to net excreted N. These N-losses were 11% of gross excretion for dairy and 15% of gross excretion for pigs (based on RVO (2010), RVO (2016) and Van Kernebeek (2016)). We verified whether the provision of N from the combination of fixed and variable sources was sufficient to meet N-uptake. N-uptake was computed from fresh matter yield (Y), dry matter content (DM) and N-content of harvested product, in analogy with Eq. 1. N application from the fixed sources of deposition and biological fixation were computed from the following assumptions: atmospheric deposition totals 25,6 kg/ha (LEI and CBS, 2012), with NFRV of 75% on grassland (Schröder and Van Keulen, 1997) and 60% on cropland (Asman, 1992). N-fixation by legumes (i.e. beans) is 50 kg per ha of beans on all soil types (Beukeboom, 1996). N-fixation per ha of crop rotation was computed from the frequency of beans in the rotation. We concluded that only in the case of grass production on peat soil, N from these two sources was lower than N-uptake. For grass production on peat soil, we modelled therefore that instead of the economic optimal fertilisation rate, the amount of N taken up should be provided by variable sources.

Table S1. Total requirement (TR) of nutrients (N, P, K) by crop rotation and soil type (kg/ha), Fixed provision (FP) of N (kg/ha) from fixed sources, and N-uptake (kg N/ha)

		TR (kg/ha)			FP (kg /ha)	Uptake (kg/ ha)
Rotation[a]	**Land type**	**N**	**P**	**K**	**N**	**N**
G	Clay	340	47	380	19	332
M	Clay	185	32	188	15	191
WOWB	Clay	183	25	75	33	159
PWSW	Clay	203	29	144	40	197
PBSW	Clay	185	25	131	57	188
WOWBS	Clay	176	26	89	44	171
WOWBWP	Clay	198	26	111	31	170
G	Sand	340	44	353	19	308
M	Sand	185	33	191	15	194
WOWB	Sand	155	24	70	33	149
PWSW	Sand	178	27	137	40	184
PBSW	Sand	175	24	127	57	181
WOWBS	Sand	154	24	87	44	162
WOWBWP	Sand	172	25	107	31	161
G	Peat	268	45	366	19	321

[a]G=grass, M=silage maize, W= wheat, O= oilseed, B=beans, P = potato, S = sugar beet.
Note: Fertiliser requirements of plant available N were dependent on mineral soil N contents, which we assumed 20 kg N/ha (0-30 cm) and 30 kg N/ha (0-60 cm) for arable soils (Schröder et al., 2004), 140 kg N/ha for grasslands on clay and sandy soils, and 250 kg N/ha for grasslands on peat soils (Hoeks et al., 2012).

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Supplementary material S3 Primary energy use for the production of inputs[a], and primary energy requirement for fixed field operations per hectare of crop rotation (MJ ha^{-1}).

Crop rotation	Land type	Seeds	Pesticides	Fixed field operations
G	Clay	573	46.9	487
M	Clay	543	119	7593
WOWB	Clay	588	381	4489
PWSW	Clay	2696	1203	7376
PBSW	Clay	2696	1107	7209
WOWBS	Clay	476	427	5273
WOWBWP	Clay	1935	858	5665
G	Sand	573	46.9	487
M	Sand	543	105	7593
WOWB	Sand	557	310	4489
PWSW	Sand	2754	1236	7376
PBSW	Sand	2769	1175	7209
WOWBS	Sand	452	338	5273
WOWBWP	Sand	2086	883	5665
G	Peat	573	46.9	487

G = grass, M = maize, W = wheat, O = rapeseed, B = beans, P = potato, S = sugar beet
Note: Seed requirements per ha of sugar beet, silage maize and grass were taken from Vellinga et al. (2013). Pesticide requirements per ha of grass were taken from LEI (2015). Requirements for seeds and pesticides per ha for other crops were averages from PPO (2009, 2012). Energy requirements for production of seeds and pesticides (MJ/kg) were taken from Weidema et al. (2013). Energy use per ha for fixed field operations were computed from Vellinga et al. (2013). Field operations for fertilisation of crops were variable, and dependent on the types of fertilisers applied to the land. Application of mineral N required 0.4 MJ/kg N (Dalgaard et al., 2001; Nguyen et al., 2010). We assumed equal energy requirements for application of mineral P and K. Furthermore, application (loading and spreading) of manure and digestate required 21 and 27 MJ per ton fresh matter (FM) respectively (Berglund and Börjesson, 2006; Nguyen et al., 2010). We assumed that application (loading and spreading) of crop products also required 27 MJ per ton FM.

[a]Primary energy use for production of artificial lime and mineral N, P, K fertiliser per ha is variable. We assumed application of 380 kg lime per ha of crop rotation (Vellinga et al., 2013).

Lime could be provided by two sources, i.e. artificial lime and sugar factory lime, the latter which is a co-product from sugar beet processing. Use of artificial lime was thus dependent on the availability of sugar factory lime. Energy requirements for the production of artificial lime and mineral fertiliser N,P,K (MJ/kg) were taken from Weidema et al. (2013). We assumed that sugar factory lime had a $CaCO_3$ concentration of 70% on DM basis (Hoeks et al., 2012; Paleckiene et al., 2007; Royal Cosun and Suikerunie, 2016). We assumed no effect of the type of lime on crop yield (Draycott and Messem, 1979). Furthermore, use of mineral N, P and K depended on the availability of other sources of N, P and K fertiliser (Supplementary Material S2). As a consequence, energy use for production of mineral N, P and K is, therefore, also variable.

Supplementary material S4 Primary energy requirement for crop and animal processing (MJ primary energy per unit)

Table S2d. Primary energy requirement for crop and animal processing (MJ primary energy per unit)

	MJ primary energy	
	per ton DM	**per ton FM**
Feed processing		
Grinding wheat grain[c,d]	38	
Chopping wheat straw[e]	21	
Heating potatoes[f]	286	
Cutting sugar beet[g]	244	
Cutting sugar beet tops & tails[h]	244	
Grinding rapeseed[i]	37	
Chopping rapeseed straw[e]	18	
Heating beans[a]	70	
Pelleting[j]		82
Food processing		
Dry milling wheat[a]	552	
Peeling potato[g]	2484	
Sugar beet processing[a]	2417	
Crushing rapeseed[a]	1181	
Animal processing		
Beef[k]		3283
Pork[k]		3903
Milk[l]		712
Human inedible animal products, wasted meat and milk[m]		1960

[a]Vellinga et al. (2013), [b]Computed with the assumption that one ha of grassland is mown twice a year for silage, [c]Marian et al. (2013), [d]Dziki (2011), [e]Tumuluru et al. (2014), [f]energy use for heating of potatoes was assumed equal that for industrial heating of beans, [g]Sikirica et al. (2003), [h]energy use for cutting sugar beet tops & tails was assumed equal that for cutting sugar beet, [i]energy use for grinding rapeseed was assumed equal that for grinding of wheat grain, [j]Wondra et al. (1995). [k]Ramírez et al. (2006b) energy use for slaughtering, removal of hides, hair and offal, cutting and deboning up to and including cooling. [l]Ramírez et al. (2006a) energy use for pasteurisation and cooling of milk. [m]energy use for processing wasted meat and milk was assumed equal that for processing human inedible products as taken from Schreurs (2004).

Supplementary Material S5 Nutrient balances in crop processing

To assure nutrient balances in crop processes that involved separation of harvested crop into multiple crop products, we compared the nutrient content (kg of P, K and N per ton DM) of each harvested crop before processing (PDV, 2011) with the nutrient content of the sum of output products (including wastes), which we computed from nutrient content per ton dry matter (PDV, 2011) and output/input ratios (Van Kernebeek et al., 2016). We computed N content as 16% of crude protein (PDV, 2012). In those cases where the nutrient content in the sum of output products was lower than in the harvested crop, we assumed that the remaining nutrients were dissolved in industrial processing water. This was the case for two processes, i.e. potato and sugar beet processing. During potato processing, 0.46 kg of 2.50 kg P/ton DM, 9.88 kg of 22.10 kg K/ton DM and 1.42 kg of 16.32 kg N/ton DM potato tuber ended up in industrial processing water. During sugar beet processing, these quantities were 0.51 kg of 1.6 kg P/ton DM, 2.27 kg of 8.0 kg K/ton DM and 1.04 kg of 6.56 kg N/ton DM sugar beet.

In those cases where nutrient content in the sum of output products was higher than in the harvested crop, we lowered the nutrient content of output products by solving a system of linear equations such that the initial nutrient ratio (PDV, 2011) between output products remained unchanged. This was the case for the remaining two food processes that involved separation of harvested crop into multiple crop products, i.e. dry milling of wheat, and crushing of rapeseed. The nutrient contents of the output products of the dry milling of wheat were lowered from 2.8% to 2% N, from 0.68% to 0.35% P and from 0.85% to 0.43% K. The nutrient contents of the output products from rapeseed crushing were lowered with less than 1%. To account for the relation between N and protein, we lowered the contents of intestinal digestible protein and rumen degradable protein in feed ingredients for cows with the same percentage as the percent-change in N.

Supplementary S6 Feed restrictions and nutrient retention in animals

In addition to feed restrictions described by Van Kernebeek et al. (2016) and Van Kernebeek et al. (2018), we also restricted intake of wheat products for pigs to 40% of dry matter intake (Feedipedia, 2018).

Nutrient retention per animal PU was fixed, and was computed from nutrient concentrations in body tissue and milk (Groenestein et al., 2008; RVO, 2010), and production data (Van Kernebeek et al., 2016) (Supplementary Table S5). Retention in body tissue per cowPU included retention in replaced dairy cow, surplus calves, and deceased replacement calves. Retention in human inedible products was computed as ‘retention in body tissue minus retention in meat’. Retention in milk for human consumption was computed as ‘retention in raw milk minus retention in milk for replacement calves’.

Table S5. Production of meat and milk per animal production unit (PU) per year (kg), nutrient retention in body tissue, meat, milk, and human inedible products (kg), and nutrient contents of meat and milk (g kg^{-1})

		Retention (kg)			Content (g kg^{-1})[b]		
	kg[a]	P	K	N	P	K	N
PigPU							
Body tissue	379	2.0	0.86	9.2			
Of which meat	171	0.51	0.80	8.7	3.0	4.7	51
Of which human inedible products	208	1.5	0.06	0.50	7.1	0.3	2.4
CowPU							
Body tissue	235	1.8	0.48	5.51			
Of which meat	74	0.20	0.36	3.56	2.7	4.8	48
Of which human inedible products	161	1.6	0.12	2.0	9.7	0.7	12
Raw milk	8120	7.9	13.1	44.5			
Of which for replacement calves	79.2	0.08	0.13	0.43			
Milk for human consumption	8502	7.8	12.9	44.1	0.92	1.5	5.2

Note: FPCM = Fat and protein corrected milk. [a] see Online resources III and VI in Van Kernebeek et al. (2016) for herd composition and meat and milk production per animal PU. Weight (kg) of human inedible products were computed as ‘weight of body tissue minus weight of meat’. [b]Nutrient contents of meat were taken from RIVM (2013). However, to create positive K-retention in animal meal per cowPU, we lowered K-content of beef as presented by RIVM (2013) by an arbitrary 8%, from 0.527% to 0.485% K. For the same reason, we also lowered K-content of pork as presented by RIVM (2013) by an arbitrary 8%, from 0.568%

to 0.466% K in pork. Nutrient content of milk for human consumption was computed from production (kg) and nutrient retention. Nutrient contents of our milk for human consumption were comparable with the contents of full fat milk as presented by the Dutch Food Composition Table NEVO (RIVM, 2013) (i.e. 1.02 g P/kg; 1.65 g K/kg; 5.28 g N/kg). For human edible animal products (meat and milk) and human inedible products (animal meal), contents of net energy for pigs and cows, intestinal digestible protein (IDP), rumen degradable protein (RDP), structure value and saturation value were not found in literature. Net energy (MJ/kg) for pigs and cows were therefore computed from crude protein contents, which we computed from N-contents as presented in the table above, contents of fat, fibre and other carbon-containing components as presented by PDV (2011), and the energy equations provided by PDV (2011). Contents of intestinal digestible protein (IDP), rumen degradable protein (RDP), structure value and saturation value, were assumed equal to those of potato peel.

Supplementary Material S7 Manure production

We assumed fixed manure production volumes of 1.3 ton fresh manure per PigPU and 31.5 ton fresh manure per CowPU based on CBS (2014) and dry matter contents of 9.3% for manure from PigPU and 8.5% for manure from CowPU based on De Buisonje (2014). To align mutual N, P, and K excretion ratios with N, P and K application ratios on each hectare, we assigned excreted nutrients to manure types that differ in their nutrient concentration (Eq. 3):

$$\sum_{b=1}^{41} \left(ManVol_{a,b} \times ManConc_{a,b,n} \right) = \text{Net excretion}_{n,a} \qquad \text{Eq. 3}$$

Where $ManVol_{a,b}$ the manure volume (ton DM) of manure type b produced by animal production system type a and $ManConc_{a,b,n}$ is the nutrient concentration of manure (kg nutrient/ton DM) of this manure type produced in this animal production system. We defined 41 manure types per animal production system type, each different in their nutrient concentration. The manure types contained the default concentrations of N, P, and K (i.e. 7.63% N, 2.16% P and 5.21% K on DM basis for liquid pig manure, and 4.82% N, 0.77% P, and 5.69% K on DM basis for liquid cattle manure (based on Vermeij (2013) and herd composition (Van Kernebeek et al., 2016)), and any combination of these nutrients in a range between minus 2 and plus 2%-points, with steps of 1%. We assumed that the minimum concentration of N, P, and K was 0.1%. $\text{Net excretion}_{n,a}$ is the net excretion of nutrient n from animal production system type a. Carbon contents were 231 and 376 kg C/ton DM for pig and cow manure respectively, and were computed from organic matter contents provided by (De Buisonje et al., 2014).

Supplementary Material S8 Anaerobic digestion

The content of carbohydrates and crude fat in crop biomass was taken from PDV (2011), whereas for crude protein this was taken from Van Kernebeek et al. (2018). Van Kernebeek et al. (2018) adapted the content of crude protein from crop biomass originally presented by PDV (2011) to assure a closed protein balance between unprocessed crop products (e.g. wheat grain) and the sum of processed crop products (e.g. wheat middlings, wheat germ, wheat bran and wheat flour). Digestibility of crop biomass when fed to the digester was assumed equal to that when fed to cows (PDV, 2011). Digestibility of meat and milk was estimated at 60% based on PDV (2011). The dry matter content of human inedible products from cows and pigs was taken as 29% (Hejnfelt and Angelidaki, 2009; Van Kernebeek et al., 2016).

Energy recovery (GJ/ton DM) from crops, crop products, manure, meat, milk and slaughterhouse waste in a high productive digester

		GJ/ton dry matter	Reference
Crops	Wheat grain	14.3	This study
	Wheat straw	6.15	This study
	Potato	13.2	This study
	Sugar beet	11.6	This study
	Sugar beet tops & tails	10.0	This study
	Rapeseed	22.8	This study
	Rapeseed straw	7.59	This study
	Brown beans	12.6	This study
	Silage maize	11.4	This study
	Silage grass	11.4	This study
	Fresh grass	13.2	This study
Crop products	Wheat middlings	12.6	This study
	Wheat germ	13.9	This study
	Wheat bran	10.6	This study
	Wheat flour	14.1	This study
	Potato tuber	12.8	This study
	Potato peel	13.2	This study
	Sugar	15.9	This study
	Sugar beet molasses	13.4	This study
	Sugar beet pulp	13.1	This study
	Rapeseed oil	36.7	This study
	Rapeseed meal	12.6	This study
Animal products	Pig manure	7.00	Berglund and Borjesson
	Cow manure	6.20	Berglund and Borjesson
	Pork	13.0	This study
	Beef	14.7	This study
	Milk	12.7	This study
	Slaughterhouse waste	9.4	Berglund and Borjesson

Supplementary material S9 Energy use (TJ) by various activities in the food system

Table S12. Energy use (TJ) by various activities in the food system in the baseline situation without digester (Baseline_no_AD) for diets varying in percentage of protein from animals (%PA), and energy use in the alternative situations (presented between brackets as fraction of energy use in Baseline_no_AD (Baseline_AD; WPREV_no_AD; WPREV_AD)

% PA	0	15	30	60	80
Crop production	15887(0.96;0.67;0.66)	14304(1.00;0.72;0.72)	15096(0.99;0.75;0.73)	18747(0.96;0.76;0.74)	22959(0.95;0.77;0.74)
Crop processing	12526(1.00;0.68;0.68)	12313(1.00;0.83;0.83)	12686(1.00;0.93;0.93)	15063(1.00;0.80;0.80)	12894(1.00;1.10;1.10)
Cooling and drying	5215(1.00;0.66;0.66)	4865(1.00;0.75;0.75)	4870(1.00;0.79;0.79)	5185(1.00;0.77;0.77)	5414(1.00;0.88;0.88)
Feed processing		225(1.00;0.75;0.75)	296(1.00;0.80;0.80)	435(1.00;0.72;0.72)	391(1.00;1.13;1.13)
Food processing	7311(1.00;0.69;0.69)	7222(1.00;0.89;0.89)	7520(1.00;1.03;1.03)	9443(1.00;0.82;0.82)	7089(1.00;1.27;1.27)
Animal production		756(1.00;0.89;0.89)	1686(1.00;0.91;0.91)	4211(1.00;0.87;0.87)	6001(1.00;0.93;0.93)
MCP production		1(1.00;0.00;0.00)	10(1.00;0.00;0.00)		
Animal processing		1355(1.00;0.87;0.87)	2829(1.00;0.92;0.92)	7060(1.00;0.87;0.87)	10091(1.00;0.93;0.93)
Cold storage meat & milk in warehouse and retail		742(1.00;0.89;0.89)	1525(1.00;0.96;0.96)	3880(1.00;0.89;0.89)	5547(1.00;0.95;0.95)
Home consumption	15009(1.00;1.05;1.05)	14010(1.00;1.10;1.10)	13755(1.00;1.14;1.14)	13286(1.00;1.11;1.11)	12075(1.00;1.25;1.25)
Cold storage meat & milk		289(1.00;0.90;0.90)	594(1.00;0.96;0.96)	1512(1.00;0.89;0.89)	2161(1.00;0.95;0.95)
Boiling crop products	15009(1.00;1.05;1.05)	13617(1.00;1.11;1.11)	12851(1.00;1.16;1.16)	11298(1.00;1.14;1.14)	9234(1.00;1.33;1.33)
Frying meat		104(1.00;0.95;0.95)	310(1.00;0.71;0.71)	476(1.00;1.08;1.08)	681(1.00;1.15;1.15)
Anaerobic digestion[a]	0(2546;0;1154)	0(1089;0;836)	0(1074;0;1107)	0(1196;0;868)	0(1654;0;931)
Transport	2521(0.93;0.64;0.61)	2451(1.00;0.79;0.80)	2747(1.01;0.91;0.91)	4018(1.00;0.79;0.80)	4157(1.01;0.98;0.99)
Total energy use	45943(1.04;0.80;0.81)	45932(1.02;0.88;0.90)	50334(1.02;0.93;0.95)	66265(1.01;0.87;0.87)	73724(1.01;0.97;0.97)

[a]Energy use during the anaerobic digestion is expressed as TJ. MCP = monocalcium phosphate

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In the baseline situation, energy use for food processing is 2,200 TJ higher for a diet containing 60% PA compared to a diet containing 15% PA, despite lower crop consumption by humans. This can be explained from the increase in food processing of sugar beet. At 60% PA, a large share of the sugar beet allocated to animals underwent food processing rather than feed processing, despite the higher energy use in food processing. This can be explained from two factors; 1) increased ranges of freedom to allocate the separate output products from food processing of sugar beet rather than the cut sugar beet from that result from feed processing. As a result of the higher degrees of freedom, animal requirements for net energy and ration restrictions with regard to saturation can more easily be met, and 2) slightly lower product losses during feeding of the output products that result from food processing compared to the sliced sugar beet that result from feed processing. This illustrates that minimising energy use for a specific activity in the food chain can conflict with optimising energy input to the food system.

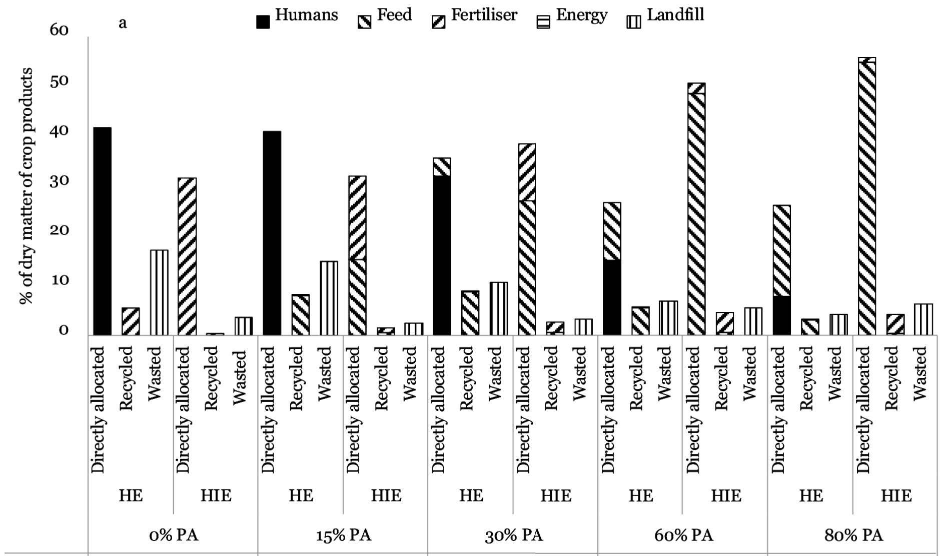
In Baseline_AD, energy use for crop production was up to five percent lower compared to Baseline_no_AD, implying that substituting crop products, used as fertiliser in Baseline_no_AD, by digestate, resulted in lower energy use for the production of mineral fertiliser, and for field application of fertiliser. Underlying factors include the fertiliser replacement value, which is higher for digestate than for crop products, dry matter content and NPK content of fertilising sources, and type of machine used on the field.

For a diet with 0% PA, energy use for transport was about seven percent lower compared to Baseline_no_AD. At 0% PA in Baseline_no_AD, a relative large share of the wasted crop biomass is allocated to the landfill, which includes a relatively large transport distance (See supplementary material S10 for more details). In Baseline_AD, this biomass is allocated to the digester, which includes a relatively small transport distance.

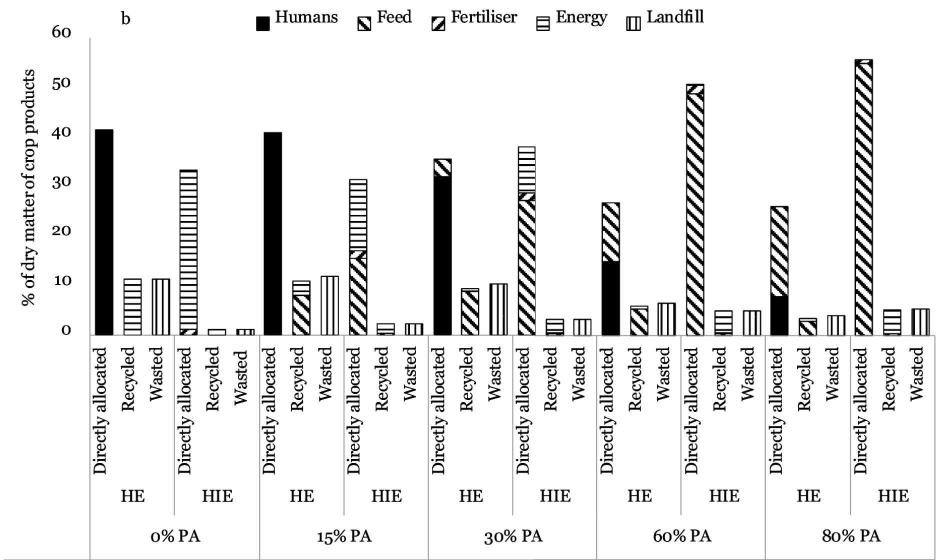
Total energy use is slightly higher in Baseline_AD compared to Baseline_no_AD due to energy use for anaerobic digestion.

Supplementary material S10

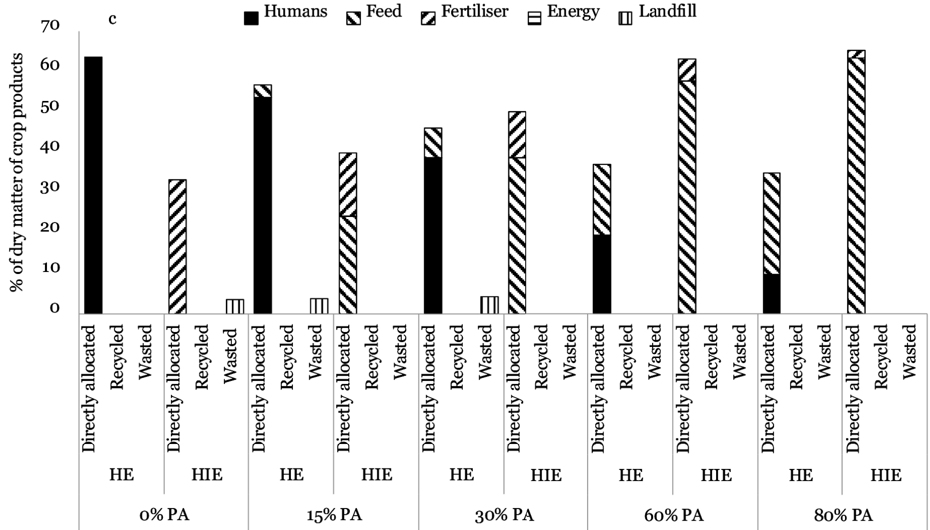
Distribution of human edible (HE) and human inedible (HIE) crop products (% of dry matter) as food to humans, as feed to animals, as fertiliser to land, as source of energy to the digester or as waste to landfill for diets varying in their percentage of protein from animals %PA, and land use (1000 ha), production of HE and HIE (1000 ton DM), and total production of crop biomass (1000 ton DM) in a) Baseline_no_AD, b) Baseline_AD, c) WPREV_no_AD, and d) WPREV_AD. Note: an unprocessed crop product that would provide HE products after food processing is denoted as HE to its full extend. Note to b and d: At each %PA about 1 - 2% of crop biomass was allocated to land as fertiliser. This flow of biomass consists of sugar factory lime. At 15, 30 and 60% PA, part of the HIE crop products are directly allocated to the digester. It concerns wheat straw and rapeseed straw, which cannot all be consumed by animals.



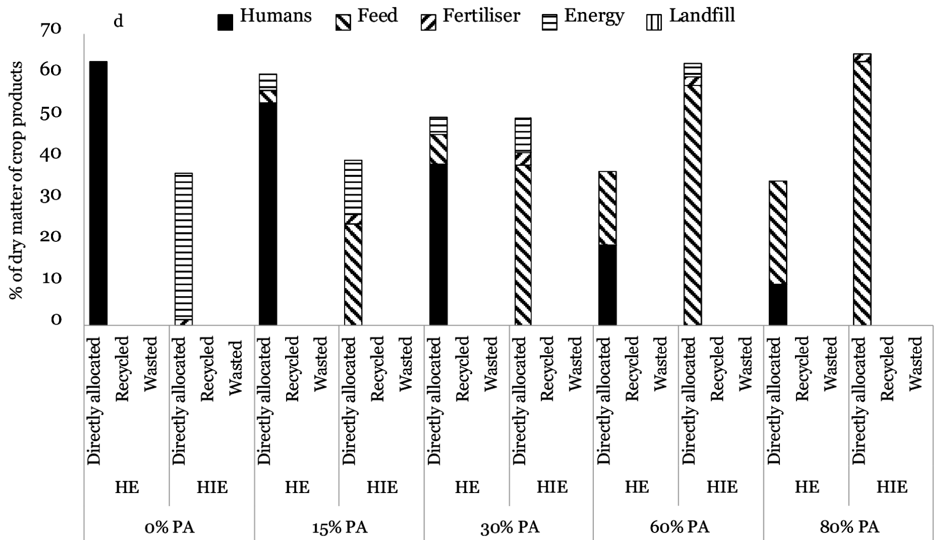
	0% PA	15% PA	30% PA	60% PA	80% PA
Land use	740	669	747	1013	1331
HE prod	4748	4429	4433	4753	4976
HIE prod	2630	2501	3577	7365	9796



	0% PA	15% PA	30% PA	60% PA	80% PA
Land use	740	669	747	1013	1331
HE prod	4748	4429	4433	4753	4976
HIE prod	2630	2501	3577	7365	9796



	0% PA	15% PA	30% PA	60% PA	80% PA
Land use	498	486	558	802	1055
HE prod	3097	3251	3420	3606	4383
HIE prod	1782	2141	3405	6137	8240



	0% PA	15% PA	30% PA	60% PA	80% PA
Land use	498	486	558	802	1055
HE prod	3097	3251	3420	3606	4383
HIE prod	1782	2141	3405	6137	8240

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Supplementary material S11 Composition of the cow ration

Table S9. Composition of the cow ration for human diets varying in percentage of protein from animals (%PA) in the baseline situation without digester (Baseline_no_AD)

% PA	**0**	**15**	**30**	**60**	**80**
gr DM/cow PU/day					
Feed processing					
Grinded wheat grain	0	632	299	828	3423
Chopped wheat straw	0	3621	2513	2045	1748
Potatoes	0	604	198	128	95
Cut sugar beet	0	0	385	261	110
Cut sugar beet tops&tails	0	688	383	260	110
Grinded rapeseed	0	8	0	0	1.98
Chopped rapeseed straw	0	371	371	0	207
Heated beans	0	9	767	0	247
Silage maize	0	0	0	620.46	3152
Silage grass	0	617	4574	7113	5852
Fresh grass	0	277	2052	3248	2626
Food processing					
Wheat middlings	0	3621	1715	507	86
Wheat germ	0	68	0	82	14
Wheat bran	0	569	193	515	87
Wheat flour	0	3339	1472	470	80
Potato	0	2061	926	420	311
Potato peel	0	756	369	153	113
Sugar	0	0	1797	2670	1297
Sugar beet molasses	0	1297	722	489	207
Sugar beet pulp	0	2201	1277	896	379
Sugar beet tops&tails	0	0	0	0	0
Rapeseed oil	0	0	0	0	8
Rapeseed meal	0	1479	552	0	214
Beans	0	90	0	0	0

Supplementary material S12 Composition of the human diet

Table S12a. Composition of human diets varying in percentage of protein from animals (%PA) in the baseline situation (Baseline_no_AD and Baseline_AD)

% PA	**0**	**15**	**30**	**60**	**80**
gr/cap/day					
Wheat germ	6	5	5	0	0
Wheat bran	39	35	34	0	0
Wheat flour	239	213	207	156	38
Potato	591	551	550	582	617
Sugar	62	72	61	26	3
Rapeseed oil	17	13	11	0	14
Beans	87	66	0	0	0
Pork	0	0.28	2.65	0	0
Beef	0	2	4	10	15
Milk	0	254	519	1328	1899

Table S12b. Composition of human diets varying in percentage of protein from animals (%PA) in the alternative situation where waste is prevented (WPREV_no_AD and WPREV _AD)

% PA	**0**	**15**	**30**	**60**	**80**
gr/cap/day					
Wheat germ	6	6	0	0	0
Wheat bran	39	36	29	0	0
Wheat flour	240	220	202	123	0
Potato	654	686	722	747	896
Sugar	61	73	60	29	1
Rapeseed oil	14	7	0	0	0
Beans	70	0	0	0	0
Pork	0	0	0	0	0
Beef	0	2	5	11	17
Milk	0	246	541	1273	1944

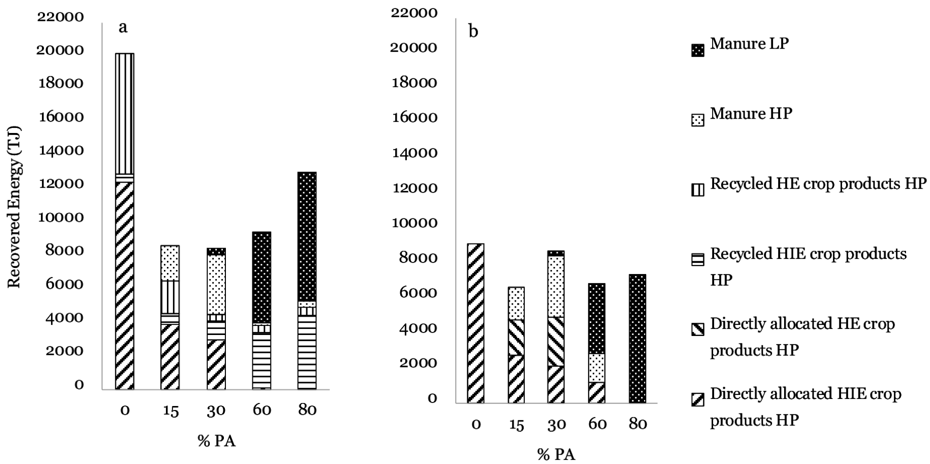
Supplementary material S13

Number of animal production units (PU) (1,000) included in the food system, and amount of mineral fertiliser N (1,000 ton) applied to land for all four situations for diets varying in percentage of protein from animals (%PA).

	%PA	**0**	**15**	**30**	**60**	**80**
Baseline_no_AD	CowPU	0	204	417	1066	1523
	PigPU	0	13	119	0	0
	Mineral fertiliser N	131	117	124	162	186
Baseline_AD	CowPU	0	204	417	1066	1523
	PigPU	0	13	119	0	0
	Mineral fertiliser N	119	115	121	155	175
WPREV_no_AD	CowPU	0	179	394	929	1419
	PigPU	0	0	0	0	0
	Mineral fertiliser N	90	84	95	123	151
WPREV_AD	CowPU	0	179	394	929	1419
	PigPU	0	0	0	0	0
	Mineral fertiliser N	85	83	91	119	146

Supplementary material S14 Energy recovery

At 15% PA, approximately 25% of the recovered energy originates from manure (Figure 4 in main manuscript). This manure is digested in high productive (HP) digesters. The mixture of relatively large volumes of crop products with manure results in an input mixture with a C:N ratio between 20 and 30, as is required for a HP digester. As PA increases from 15% upwards, manure was allocated to the HP digester up to the point where it could be mixed with large enough quantities of crop products to create an input mixture for the digester with an C:N ratio above 20. From 60% PA upwards, manure is predominantly digested in low productive (LP) digesters, as not enough crop products are available to meet the required C:N ratio for the input mixture of a HP digester.



Energy recovery (TJ) in a) Baseline_AD and b) WPREV_AD through anaerobic digestion in high productive (HP) and low productive (LP) anaerobic digester for diets varying in percentage of protein from animals (%PA). Note: energy recovery of less than 50 TJ (i.e. from crop products in LP digester, and meat, milk and animal meal) were not shown.

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Chapter 6

General discussion

1. Introduction

It is generally acknowledged that we should use natural resources more efficiently in order to secure food availability for future populations. The availability of natural resources for food production, such as land, phosphate rock and fossil energy, is limited (IPCC, 2019; Reitzel et al., 2019). At present, however, these resources are inefficiently used in the food system (Cordell et al., 2009; Pelletier et al., 2011; Willett et al., 2019). The effect of several technical and consumption strategies such as preventing and recycling waste, recovering waste as bio-energy, and reducing consumption of animal-source food on resource use efficiency in food production has been assessed (Papargyropoulou et al., 2014; Röös et al., 2017; Morales-Polo et al., 2018; Van Zanten et al., 2018). So far, however, studies that have evaluated the effects of these strategies on the use of natural resources do not account for nutritional quality of human diets, do not consider the food system as a whole, or do not account for the combined effects of strategies in the entire food system.

The objective of this thesis was therefore to understand the combined effects of technical and consumption strategies, to reduce the use of natural resources in a food system. This chapter will discuss this thesis and conclude on how the separate studies have contributed to this objective. It starts with a discussion about the methodological approach in Section 2. Subsequently, the consequences for land, phosphorus and energy are discussed in Section 3. In Section 4, the effect of closing the phosphorus cycle on responsible use of phosphorus is discussed. Finally, in Section 5, the conclusions of this thesis are presented.

2. A food systems approach

Our food systems approach integrates key subsystems of the food system. It includes on the one hand all agro-ecological activities related to the production, processing, distribution and utilisation of food and related biomass, and on the other hand the outcomes of these activities in terms of energy and protein provision to people and natural resource use. Our food systems approach overcomes important drawbacks of assessments of single sectors, chains or food products. First, the food systems approach assesses the impact of the food system as a whole, rather than just the part of the impact that is allocated to a sector, chain or food product. This approach enables a comparison of the impact of food production with biophysical boundaries, such as the availability of natural resources. In this thesis, an example of such a comparison was given for land, as land use for food production was limited to land availability in the Dutch agricultural system. However, to answer the question whether the global population can be sustained with available resources, application of the food systems approach to the global set of food systems, and to more realistic scenarios are required. Second, the food systems approach accounts for the production of indivisible product-packages to acknowledge, for instance, the co-production of meat, bones, offal and manure with milk from dairy cows, rather than accounting for only the main product, i.e. milk. Accounting for all products of the package is important when assessing the impact of technical and consumer strategies. When assessing the impact of full recycling of animal waste, for example, not only meat and milk, but also the inedible products, such as manure and animal meal, will have to be recycled. Moreover, when assessing the impact of consumption of a vegetarian diet, one has to account for the impact of associated meat from culled milking cows and surplus calves (Zanten et al., 2018). Third, a food systems approach clearly shows competition for biomass use between humans, the livestock sector and the energy sector. This competition for biomass is generally referred to as the food, feed, fuel competition (Muscat et al., 2019). Accounting for this competition is important when answering the question of how this biomass can be optimally used (Muscat et al., 2019). In this thesis, I explored the optimal use of biomass to feed the human population from three perspectives, i.e. the minimisation of land use (Chapter 3), mineral phosphorus input (Chapter 4) and energy input (Chapter 5).

The material and nutrient flow model developed for this thesis is a conceptual representation of a food system that was parameterised with crop and animal production data from the Netherlands. Moreover, the system was considered a closed system, i.e. it was assumed that there was no import and export of food and feed. The Dutch food system is not representative for many other food systems in the world, given its temperate climate, and its fertile soils and high productivity. Yet, this parameterised model is suitable for demonstrating principles of resource use efficiency in other food systems, as shown in Chapters 3 and 4, and further elaborated in Section 3.1.

The food system modelled in this thesis provides a representation of the diet available for a population for an entire year. The time frame of a year was chosen as it enables connecting animal production with the yearly cycle of crop production, and to express natural resource use of all processes in one joint temporal unit. This automatically implies that the unit 'meal', although often used as the functional unit in life cycle assessments (Chapter 2), is undesirable for the food systems approach, as it would be an arbitrary derivative of the yearly food production. Moreover, the unit 'meal' is also not recommended in life cycle assessments, as the composite nutritional quality of a meal is not representative for the average daily nutrient intake (Chapter 2).

The yearly diet produced for the population provides sufficient levels of energy and protein without exceeding maximum recommended intake levels of sugar. The modelling work did not attempt to formulate healthy diets. Models aiming at formulating healthy diets should also account for nutrient quality and bioavailability. Animal-source food is associated with a higher quality of protein and higher bioavailability of iron compared to plant-source food (Hallberg, 1981; Otten et al., 2006; Drewnowski and Fulgoni, 2008). Moreover, models aiming at formulating healthy diets should also account for a broad range of vitamins and minerals, in recognition of the importance of overall nutritional quality to compose a healthy diet (Miller et al., 2009). To account for overall nutritional quality, these models should furthremore include a wider range of crop and animal products compared to the selection of products included in this thesis. Fruits and vegetables, for example, contain substantial amounts of vitamins and minerals relative to energy (Darmon et al., 2005; Fulgoni et al., 2009). A diet containing fruits and vegetables will, therefore, be better able to provide sufficient levels of these micronutrients compared to the diets presented in this thesis.

With the optimisation approach applied in this thesis, one optimal solution was found for each combination of technical and consumption strategies. To formulate healthy and accepted diets, however, I deem it necessary to look for, and present, multiple optimal and near-optimal solutions. The final selection of food products associated with optimal or near-optimal solutions by consumers will then be based on social, cultural, and personal preferences, rather than on minimisation of resources (Paris, 2016; de Boer and Aiking, 2019). It will, therefore, be worthwhile to explore multiple and near-optimal solutions, i.e. crop and animal product combinations, to enhance consumer acceptance of diets that contribute to a sustainable food system.

The modelling exercises in this thesis explore options to minimise the use of land (Chapter 3), phosphorus (Chapter 4) and energy (Chapter 5). Sustainability, however, is not limited to sustainable use of these natural resources alone. To make a food system more sustainable, other environmental impacts, as well as social and economic impacts will also have to be accounted for. The challenges to design a sustainable food system are numerous, and include, among others: Connecting food and non-food functions, such as the provisioning of fibre, pharmaceuticals and ecosystem services (De Boer and Van Ittersum, 2018; Dumont et al., 2019; Padró et al., 2019), optimising food and non-food functions across scale by accounting for local, regional and global biophysical and technical limits (Dumont et al., 2019; Padró et al., 2019), developing conditions for safe recycling of waste and human excreta (Boqvist et al., 2018), and facilitating dietary change towards diets lower in their content of animal-source food (de Boer and Aiking, 2019). It will thus require social and economic efforts from all actors to develop a food system that is able to supply the global population with safe and healthy food within environmental limits.

3. The consequences of technical and consumption strategies on resource use efficiency

3.1 Minimal use of resources

In this thesis, efficiency is defined as the quantity of natural resources land, mineral phosphorus and energy required to feed a fixed population. This thesis demonstrated that a food system is most efficient in terms of land use and phosphorus under the following conditions (Chapters 3 and 4, and Figures 1a and b; situation Full waste prevention (without anaerobic digestion (AD)):

1. The occurrence of food and feed waste is fully prevented;
2. Crop production is directed at producing crops for direct consumption by humans;
3. Inedible co-products from crop production are used as feed;
4. Human excreta, manure and animal meal are used as fertiliser (either directly or indirectly via anaerobic digestion);
5. Marginal land not suitable for crop production is used for the production of roughage for livestock in case not enough food can be produced from cropland.

In a land and phosphorus efficient food system, the livestock sector contributes to efficiency by converting inedible co-products into animal-source food (Chapters 3 and 4). Along with these inedible co-products, these animals also consume a small amount (i.e. 20% of dry matter) of food products, and grass produced on crop land, causing food-feed competition (Chapter 4). The availability of food products as feed results from the fact that in some situations some crop products were produced in surplus. This was an artefact of our selection of crop rotations, which limited the possibility to provide human edible proteins and calories precisely in the required ratio of 57 grams of protein and 2000 kcal. Abundancy of crop production can be avoided by inclusion of more (single-) crop rotations, which provides the model with more flexibility to produce precisely the right ratio of protein and energy. Furthermore, the production of grass as feed implies that 1) the inedible co-products can only be converted into animal-source food, when their relatively low nutritional quality is compensated for by supplementation with highly nutritious feed, and 2) producing some feed intentionally for animals, with the aim of valuing inedible co-products as feed, results in higher land and P efficiency compared with not producing this feed, and, consequently using inedible co-products as fertiliser. There is a need for supplementation due to the fact that only highly productive animals were modelled here. Highly productive animals need high productive feed to stay within their feed intake capacity. Supplementation can largely be avoided by inclusion of animals with low or moderate productivity. The nutritional requirements of these animals can largely be met by human-inedible co-products (Hal et al., 2019). However, the effects of including animals with low or moderate productivity on nutrient use efficiencies and greenhouse gas emissions are yet unknown. Assessing these impacts is important in the light of overall sustainability of food systems.

As a consequence, and in contrast to what is concluded in LCAs (Chapter 2), livestock is an essential component in a land and phosphorus efficient food system. However, the number of livestock in such a system is limited by the availability and quality of inedible co-products (Van Zanten et al., 2018). Consequently, the amount of animal-source food available for human consumption is also limited. In the modelling exercises in this thesis, land use and phosphorus input was minimal if approximately 10-15% of total protein was derived from animal-source food (Chapters 3 and 4, and Figure 1; situation Full waste prevention). The fact that both land and phosphorus are used optimally with the same percentage of animal protein follows from the fact that phosphorus is assumed to be lost via leaching and run-off only, and, these processes are directly associated with land use. Consumption of 10-15% PA equals about 6-9 gram of animal protein per capita per day. This is considerably lower than the current average consumption of animal protein in Dutch diets, which equals 60% of total protein, i.e. 48 gram of animal protein per capita (RIVM, 2019). It is also lower than the estimated 9-23 gram of animal protein available from livestock fed on inedible feed sources in global food systems (Van Zanten et al., 2018).

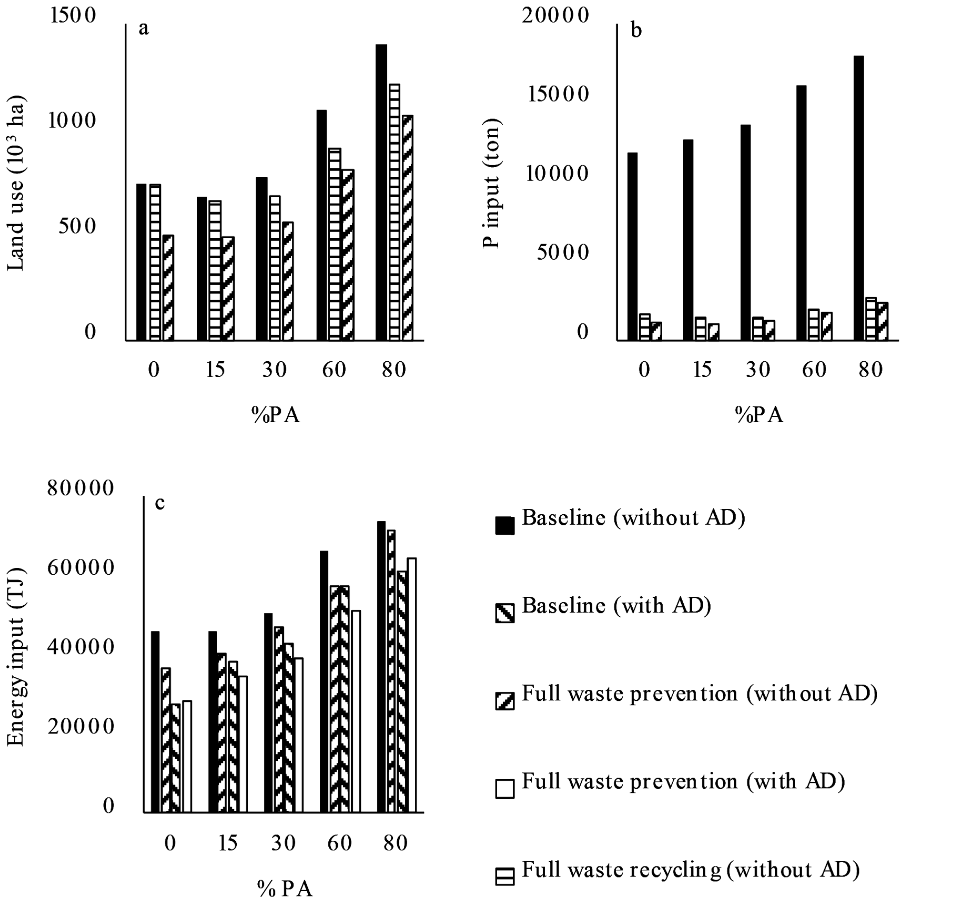

Figure 1. Land use (10^3 ha) (a), P input (ton) (b) and energy input (TJ) (c) for diets differing in their percentage of protein from animals (%PA) in various situations. AD = anaerobic digestion. To put the results of Chapters 3, 4 and 5 in one figure, the names of the situations as mentioned in the various chapters have been changed. Baseline (without AD) includes: reference situation (Chapter 3), baseline situation (Chapter 4), Baseline_no_AD (Chapter 5); Baseline (with AD) includes: Baseline_AD (Chapter 5); Full waste prevention (without AD) includes: Combi_1 (Chapter 4) (for both land use and P input), WPREV_no_AD (Chapter 5); Full waste prevention (with AD) includes: WPREV_AD (Chapter 5); Full waste recycling (without AD) includes: Combi_2 (Chapter 4) (for both land use and P input).

If not enough cropland is available to sustain the population due to population growth, marginal land will be needed to provide additional animal-source food via conversion of roughage (Chapter 3). In Chapter 3 it was demonstrated that in case of population growth, the optimal percentage of animal-source food is affected by the share of marginal land (represented by peat soil in the Netherlands, which is only used for grassland), crop productivity, and population size. In the reference situation that was parameterised for the Netherlands, 12% of total agricultural land consists of peat soil.

This system could sustain a maximum of 41.3 million people if the diet contained 22% animal protein (Chapter 3). If the food system would have had a lower share of peat soil (i.e. 5%), and thus more clay and sandy soils, it would be able to sustain more people (i.e. 43.6 million), as more crops could be produced for direct consumption by humans (Chapter 3). However, this population could only be sustained if the diet contained ca. 16% animal protein. This percentage of animal protein was lower than in the reference situation, as less grass from peat soils was available as source of animal feed, and, hence, fewer animals could be produced (Chapter 3). If the food system would have had a higher share of peat soil (i.e. 40%), a maximum of 31.5 million people could be sustained with a diet containing ca. 44% of animal protein (Chapter 3). Hence, the higher the share of marginal land in the food system, the fewer people it can sustain, and the higher the optimal share of animal-source food in the diet (Chapter 3). Moreover, if the human population approaches its maximum, the population can only be sustained if marginal land is taken into use, and, hence, sustaining this population requires consumption of animal-source food (Chapter 3).

If due to low soil quality, crop productivity is low, a food system can sustain fewer people than if it had high crop productivity (Chapter 3). Moreover, in case of low crop productivity, the range of feasible shares of animal protein in the human diet declines more rapidly with increasing population size (Chapter 3). In case of a population of 35 million people, for example, feasible diets ranged between 0 and 39% PA in the reference situation. In the alternative situation with lower crop productivity, however, feasible diets ranged between 10 and 23% PA (Chapter 3). This implies that in the alternative situation with lower crop productivity, not enough crop land is available to provide sufficient food crops for direct consumption by humans (in case of a vegan diet) and to provide sufficient feed crops for animals (in case of diets with more than 23% PA).

In contrast to a land and phosphorus efficient food system, an energy efficient food system does not prevent waste and does not produce livestock (Chapter 5 and Figure 1c; situation Baseline (with AD)). In fact, the food system is most energy efficient in case of a vegan diet where waste does occur, and where this waste, along with inedible co-products, is used to recover energy via anaerobic digestion. This may imply that recovering energy from waste, and producing additional crops to compensate for this waste, is more energy efficient than preventing waste. However, we could not draw conclusions on this, as the results in the baseline situation in which waste does occur, and the alternative situation in which waste was prevented, cannot be compared directly. This is because we first minimised land use for food self-sufficiency in the baseline and alternative situation, and subsequently minimised the energy input. This resulted in selection of other ratios and/or types of crop rotations in the alternative situation where waste was prevented than in the baseline. This selection of other crop rotations between the baseline situation and alternative situation hinders a direct comparison of their energy inputs. Moreover, the food system is most energy efficient at 0% PA for two reasons: First, diets low in %PA (0-20%) have low energy use compared to diets higher in %PA, because production, processing and storage of animal products, and production of feed, are not energy efficient. Second, at 0% PA, energy recovery from crop waste and inedible co-products is high compared to diets with PA, as there is no livestock to compete for these products with anaerobic digestion (Chapter 5).

3.2 Inefficient use of resources

From Section 3.1 it is concluded that minimising resource use in a food system requires a combination of technical and consumption strategies. The technical strategies determine the availability of waste and inedible co-products for use as feed, fertiliser or fuel. Subsequently, the consumption strategy determines the extent to which these resources are used optimally. As a consequence, for effective implementation of strategies to increase resource use efficiency in the food system, it is important to assess the combined effect of technical and consumption strategies on the entire food system, rather than combining the effect of isolated strategies. Therefore, in this subsection, the implications of consumption strategies within given technical strategies are discussed. Subsequently, the implications of technical strategies within a given consumption strategy are discussed.

The effect of consumption strategies within a given technical strategy is demonstrated by discussing land and P use efficiency across different %PA while fully preventing and fully recycling waste. When fully preventing waste, as suits the most land and P efficient food system (Section 3.1), consumption of animal protein is optimal at 10-15% in terms of land and P (Chapters 4 and 5, and Figures 1a and b; situation Full waste prevention (without AD)). If, however, less than 10% PA is consumed, not enough animals are available to utilise all inedible co-products. As a consequence, part of the inedible co-products (or all, in case of 0% PA) is used as fertiliser (either directly or via anaerobic digestion). As these co-products are thus not used as food (via conversion of animals), additional land and P is required to meet the energy and protein requirements of the population (Chapters 4 and 5, and Figure 1a and b; Situation Full waste prevention (without AD)). Consequently, land and P use for diets with less than 10% PA is higher than for diets with 10-15% PA. Moreover, when consuming more than 15% PA, inedible co-products are not sufficiently available to sustain the animals. As a consequence, additional land and P will be required for the production of feed, resulting, again, in higher land and P use compared to diets with 10-15% PA (Chapters 4 and 5, and Figures 1a and b; Situation Full waste prevention (without AD)). These principles also apply to a situation where waste is not prevented, but is, instead, recycled (Chapter 4, and Figures 1a and b; Situation Full waste recycling (without AD)). In this situation, in addition to inedible co-products, also food waste is available for livestock. Consequently, the optimal consumption of animal-source food in the human diet, in terms of land and P, is higher (i.e. 20%) if food waste is recycled than if food waste is prevented (i.e. 10%) (Chapter 4). However, as wasted food has to be compensated, recycling waste is a less land and P use efficient strategy than preventing waste (Chapter 4, and Figures 1a and b).

The effect of technical strategies within a given consumption strategy is demonstrated by discussing the effect of fully preventing and fully recycling waste on land use efficiency at 0% PA. At 0% PA, preventing waste reduced land use with 33% compared to the baseline situation (Chapters 4 and 5, and Figure 1a; situation Full prevention of waste (without AD)). However, at 0% PA, land use was not reduced by recycling waste (Chapter 4 and Figure 1a; situation Full waste recycling (without AD)). This is because in the absence of animals, wasted food crops could not be recycled as feed, and, hence, could not be used for the production of animal-source food. Hence, although the technical strategy of recycling waste increases the availability of wasted food crops for use as feed, fertiliser or fuel, the consumption strategy hinders the use of wasted food crops as feed. As using wasted food crops as feed is optimal, excluding this option by adopting a vegan diet hinders optimal use of this waste. In case of 0% PA, recycled crop waste is instead used as fertiliser (either directly or via anaerobic digestion), which reduces the input of mineral P compared to the baseline situation (Chapter 4 and Figure 1b; situation Full waste recycling (without AD)). The above implies that, if the objective is to reduce land use in case of a vegan diet, waste recycling is not an effective strategy, whereas waste prevention is.

The importance of technical and consumption strategies to reduce resource use compared to the baseline situation with 60% PA differs between the types of resources (Figures 1a, b and c). For land and energy, the importance of consumption strategies is higher than it is for P. In fact, 95% of the maximum reduction in P use can be achieved by technical strategies alone. Moreover, the higher the waste of P from a particular subsystem, the more important technical interventions to prevent P waste from this subsystem become. For example, because in the baseline about half of P waste from the food system was due to wasting human excreta, recycling P from waste water treatment plants was by far the most effective single strategy (Chapter 4). Also, the higher the share of P loss through leaching and run-off, the more effective it was to reduce land use (Chapter 4).

The potential to increase resource use efficiency depends on modelling choices. In case of energy, for example, the potential to increase use efficiency was limited by the choice to avoid production of crop biomass exclusively for bio-energy production, which is in line with European ambitions (EU, 2018). If production of crop biomass exclusively for bio-energy production would have been permitted, then all the cropland not required for food production would have been taken into use for the production of bio-energy crops. In that case, energy input would have been negative, implying that the system would be a net producer of energy. This would, however, negatively affect land and phosphorus use efficiency. As there are no substitutes for land and phosphorus, whether there are substitutes for energy production from biomass, e.g. solar and wind energy, I deem it undesirable to allow the use of land and P for production of biomass for energy production.

4. Closing the cycle is only part of the narrative

In this thesis, P use efficiency was inversely indicated by mineral P input. This indicator was chosen as long-term dependency on fossil minerals is unsustainable (Godfray et al., 2010a). Mineral P input can be reduced by closing the P cycle. It is argued here that, on top of minimising P input, also the P-flow through the food system should be minimised, as inspired by the 'narrowing the loop' concept presented by Bocken et al. (2016). The P-flow should be minimised for two reasons. The first reason to minimise the P-flow is to minimise the risk of large P losses throughout the system in case of inaccurate implementation of technical strategies. If the P-flow is minimal, the risk of large losses is also minimal. The second reason for minimising the P-flow is to minimise land use and energy input to the food system. As shown in Table 1, the P-flow, represented by P output from harvested crops, is substantially lower at 0 - 15% PA than at 60% PA (the current average consumption level of animal protein in Western-oriented countries). This implies that even if mineral P input could be strongly reduced by technical strategies (Figure 1b), a reduction in consumption of animal protein contributes to a lower P flow through the system. As shown in Section 3, a reduction in consumption of animal protein from the current 60% to 0-15% PA furthermore results in a higher land and energy use efficient food system. It is concluded therefore that, in addition to technical strategies, a reduction in the consumption of animal protein is required to contribute to a sustainable food system.

Table 1. P input (ton) and P harvested (ton) for various situations and percentages of protein from animal (%PA).

	Baseline			Full waste prevention		
%PA	0	15	60	0	15	60
P input (ton)	11898	12708	16103	1087	1059	1763
P-flow (ton)	24486	20631	31268	12839	13097	28308

Note: P-flow is represented by the amount of P from harvested crops. Results for the Baseline situation and situation Full waste prevention were taken from Chapter 4. Results for the situation Closed P cycle were generated for this general discussion.

5. Conclusions

Assessing resource use efficiency in the food system requires a food systems approach which integrates all agro-ecological activities related to food production, and accounts for the total use of resources across all these activities. The best unit to express the use of resources in the food system is the yearly food production for the population. To compose healthy diets, the overall nutritional quality should be accounted for. Increasing resource use efficiency in the food system requires a combination of technical strategies to prevent and recycle waste, and consumption strategies to reduce the consumption of animal-source food. The technical strategies determine the availability of waste and inedible co-products for use as feed, fertiliser or fuel. Subsequently, the consumption strategy determines the extent to which these resources are used optimally. Land and phosphorus use is minimised if the occurrence of waste is prevented, if crop production is directed at producing crops for direct consumption by humans, if inedible co-products from crop production are used as feed, and if human excreta, manure and animal meal are used as fertiliser (either directly or indirectly via anaerobic digestion). In the theoretical food system modelled in this thesis, this implies full prevention and recycling of waste and inedible co-products, in combination with consumption of 10-15% protein from animals (compared to 60% in current Western-oriented diets). Furthermore, if the population size approaches the maximum capacity of the food system, marginal land not suitable for crop production is essential to provide animal protein through conversion of roughage by livestock. The use of mineral phosphorus can be significantly reduced (i.e. by about 95%) by fully preventing and recycling waste. This potential is foremost achieved by recycling P from human excreta. In contrast to a land and phosphorus efficient food system, an energy efficient food system requires a vegan diet in which waste does occur, and in which this waste, along with inevitable inedible co-products, are used to recover energy via anaerobic digestion.

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Summary

It is generally acknowledged that we should use natural resources more efficiently in order to secure food availability for future populations. The availability of natural resources for food production, such as land, phosphate rock and fossil energy, is limited. At present, however, these resources are inefficiently used in the food system. The effect of several technical and consumption strategies such as preventing and recycling waste, recovering waste as bio-energy, and reducing consumption of animal-source food on resource use efficiency in food production has been assessed. So far, however, studies that have evaluated the effects of these strategies on the use of natural resources do not account for the nutritional quality of human diets, do not consider the food system as a whole, and do not account for the combined effects of strategies in the entire food system. The objective of this thesis was therefore to understand the combined effects of technical and consumption strategies, to reduce the use of natural resources in a food system.

To explore whether accounting for nutritional quality affects the comparison of the environmental impacts of human diets varying in their percentage of animal-source food products (ASFP), we reviewed 12 studies that used a life cycle assessment to quantify environmental impacts of human diets, including (average) daily diets or meals (**Chapter 2**). For each diet described in the reviewed studies, we expressed the global warming potential (GWP) and land use (LU), as provided by the review, in four functional units: per day, per daily protein intake uncapped or per daily protein intake capped to the recommended intake level of 57 g, and per composite nutrient score of a diet (NRD9.3) We concluded that the unit meal is unsuitable to compare the environmental impact of diets. Furthermore, diets that had higher percentages of ASFP were associated with higher GWPs and LU's per gram protein capped, and per composite nutrient score of a diet (NRD9.3). Without capping protein to the recommended intake level, GWP and LU per gram of protein were generally lower for diets that had higher percentages of ASFP, showing the impact of the definition of the functional unit. The effect of using NRD9.3 rather than day as functional unit was small for GWP. For LU we found no effect.

Based on the outcomes of Chapter 2, we decided to further explore natural resources needed to feed a growing human population with diets differing in their percentage of animal protein (%PA). We focussed on land (**Chapter 3**), phosphorus (**Chapter 4)**, and energy (**Chapter 5)**, and used an integrated food systems approach. The food systems approach integrates all agro-ecological activities related to the production, processing, distribution and utilisation of food and related biomass, and the

outcomes of these activities in terms of energy and protein provision to people and natural resource use. The material and nutrient flow model developed for this thesis is a conceptual representation of a food system that was parameterised with crop and animal production data from the Netherlands. The model included grain (wheat), root and tuber crops (potato, sugar beet), oil crops (rapeseed), legumes (brown bean), and animal-source food from ruminants (milk and meat) and monogastrics (pork). It was assumed there was no import and export of food and feed. The model was designed to produce sufficient energy and protein for a fixed population without exceeding the maximum intake level of sugar. Linear programming was used to minimise the use of resources for diets varying from 0% PA (i.e. a vegan diet) to diets containing 80% PA. The Dutch food system is not representative for many other food systems in the world. Yet, sensitivity analysis demonstrated that the principles included in the model also hold for other food systems.

Chapter 3 studied the relation between land use, the share of animal protein in the human diet, population size, and land availability and quality. Land is used most efficiently if people would derive ca. 12% of dietary protein from animals, especially from milk. The role of animals in such a diet is to convert co-products from crop production and the human food industry into protein-rich milk and meat. Below 12 %PA, inedible co-products were wasted (i.e., not used for food production), whereas above 12 %PA, crops had to be cultivated to feed livestock. Large populations (40 million or more) could be sustained only if a modest amount of animal protein was consumed. This results from the fact that at high population sizes, land unsuitable for crop production (peatland in our system) was necessary to meet dietary requirements of the population, and contributed to food production by providing animal protein without competing for land with crops. The optimal %PA in the human diet depended on population size and the relative share of land unsuitable for crop production.

In **Chapter 4**, the potential of preventing and recycling phosphorus (P) waste in a food system, in order to reduce the dependency on phosphate rock was assessed. In our baseline situation, in which 42% of crop waste is recycled, and humans consume 60% PA, about 60% of the P waste in this food system resulted from wasting P in human excreta. Therefore, recycling of human excreta showed most potential to reduce P waste, followed by prevention and finally recycling of agricultural waste. Fully recycling P could reduce mineral P input by 90%. The optimal amount of animal protein in the diet depended on whether P waste from animal products was fully prevented or recycled: if it was fully prevented or recycled, then a small amount of animal protein in the human diet resulted in the most sustainable use of P; but if it was not fully prevented or recycled, then the most sustainable use of P would result from a complete absence of animal protein in the human diet.

Chapter 5 assessed the potential of preventing waste, recycling waste as animal feed or fertiliser and recovering waste as bioenergy, via anaerobic digestion, to reduce energy input in the food system. Energy input was defined as the difference between energy that is used during activities in the food system, and energy that is recovered through anaerobic digestion. Energy input into the food system was reduced by anaerobic digestion and waste prevention as single interventions. If waste was not prevented, the effect of anaerobic digestion was strongest in situations where animals did not compete for food waste and human inedible crop products (at 0% PA, i.e. a vegan diet), and feed waste (i.e. at 80% PA) with anaerobic digestion. If waste was prevented, the relatively high potential to recover bio-energy from waste at 0 and 80% PA was lacking. In situations with anaerobic digestion and/or waste prevention, energy input continuously increased with increasing %PA, and, hence, a vegan diet was most energy efficient. In the baseline situation where none of these strategies were applied, however, energy input showed a minimum at about 15% PA. To reduce energy input to a food system, it is essential to account for the combined effects of waste prevention, anaerobic digestion and dietary shifts.

In **Chapter 6**, methodological choices and challenges are discussed. The discussion addressed the importance of accounting for combined effects of technical and consumption strategies for the total food system. This Chapter furthermore discussed the importance of reducing animal protein consumption in Western-oriented countries to increase land, phosphorus and energy use efficiency in the food system. It was furthermore demonstrated that reduction in consumption of animal protein will lower the P-flow through the system, and, hence, will lower risks of large P losses.

It is furthermore emphasised that the modelling work in this thesis did not attempt to formulate healthy diets. Models aiming at formulating healthy diets should account for nutrient quality and bioavailability, and should include a wider range of crop and animal products compared to the selection of products included in this thesis. Moreover, besides the use of land, phosphorus and energy, also other environmental impacts, as well as social and economic impacts will have to be accounted for. It will require social and economic efforts from all actors to develop a food system that is able to supply the global population with safe and healthy food within environmental limits.

Samenvatting

Het wordt algemeen erkend dat natuurlijke hulpbronnen efficiënter moeten worden gebruikt om de beschikbaarheid van voedsel voor toekomstige generaties veilig te stellen. De beschikbaarheid van natuurlijke hulpbronnen zoals land, fosfaaterts en fossiele energie is beperkt. Toch worden deze bronnen momenteel inefficiënt gebruikt in het voedselsysteem. Diverse studies hebben zich gericht op het effect van verschillende technische en consumptiestrategieën op het efficiënt gebruik van natuurlijke hulpbronnen in het voedselsysteem, zoals het vermijden en recyclen van afval, het terugwinnen van afval als bio-energie en het verminderen van de consumptie van voedsel van dierlijke oorsprong. Deze studies hielden tot dusver echter geen rekening met de voedingswaarde van diverse voedselproducten voor de mens, keken niet naar het gehele voedselsysteem (geen voedselsysteemperspectief), en keken niet naar de gecombineerde effecten van strategieën in het hele voedselsysteem. Het doel van dit promotieonderzoek was dan ook te begrijpen wat de gecombineerde effecten zijn van technische en consumptiestrategieën op het gebruik van natuurlijke hulpbronnen in een voedselsysteem.

We hebben onderzocht of rekening houden met de voedingswaarde van invloed is op de vergelijking van de milieueffecten van humane voedingspatronen met variërende percentages voedingsproducten van dierlijke oorsprong (animal-source food products, ASFP). Daarbij hebben we 12 onderzoeken geëvalueerd waarin een levenscyclusanalyse werd gebruikt om de milieueffecten van humane voedingspatronen te kwantificeren, waaronder die van de (gemiddelde) dagelijkse voeding of maaltijden (hoofdstuk 2). Voor ieder voedingspatroon dat werd beschreven in de geëvalueerde onderzoeken hebben we het broeikasgaspotentieel (global warming potential; GWP) en landgebruik (land use; LU), zoals vermeld in de studies zelf, uitgedrukt in vier functionele eenheden: per dag, per dagelijkse eiwitinname zonder aftopping, per dagelijkse eiwitinname afgetopt op de aanbevolen inname van 57 g (afgetopt), en per samengestelde nutriëntenscore van een voedingspatroon (Nutrient Rich Diet 9.3, NRD9.3). We kwamen tot de conclusie dat de eenheid 'maaltijd' ongeschikt is om de milieueffecten van voedingspatronen te vergelijken. Bovendien hingen voedingspatronen met een hoger percentage ASFP samen met hogere GWP en LU-waarden per gram eiwit (afgetopt) en per samengestelde nutriëntenscore (NRD9.3). Zonder de eiwitinname af te toppen op het aanbevolen innameniveau waren de GWP- en LU-waarden per gram eiwit over het algemeen echter lager voor voedingspatronen met een hogere percentage ASFP. Hieruit blijkt dat de definitie van de functionele eenheid invloed heeft op de vergelijking van milieueffecten van diëten. Het effect van het gebruik van NRD9.3 in plaats van 'dag'

als functionele eenheid was beperkt voor GWP. Voor het landgebruik hebben we geen effect gevonden.

Op basis van de resultaten van hoofdstuk 2 besloten we om nader te onderzoeken hoeveel natuurlijke hulpbronnen er nodig zijn om een groeiende bevolking te voorzien van voedingspatronen met verschillende percentages dierlijk eiwit (percentage of animal protein; %PA). We richtten ons op land (hoofdstuk 3), fosfor (hoofdstuk 4) en energie (hoofdstuk 5) en pasten een geïntegreerde voedselsysteembenadering toe. In de voedselsysteembenadering wordt rekening gehouden met alle agro-ecologische activiteiten die samenhangen met de productie, verwerking, distributie en het gebruik van voedsel en daarmee samenhangende biomassa, evenals met de effecten van deze activiteiten op de energie- en eiwitvoorziening voor mensen en het gebruik van natuurlijke hulpbronnen. Het voor dit promotieonderzoek ontwikkelde massa- en nutriëntenstromen model is een conceptuele weergave van een voedselsysteem waarbij gewas- en dierproductiegegevens uit Nederland zijn gebruikt als parameters. Het model bevatte graan (tarwe), wortel- en knolgewassen (aardappel, suikerbiet), oliegewassen (koolzaad), peulvruchten (bruine boon) en dierlijke producten afkomstig van herkauwers (melk en vlees) en eenmagigen (varkensvlees). Aangenomen werd dat er geen sprake was van in- en export van voedsel en diervoeder. Het model werd ontworpen om voldoende energie en eiwitten te produceren voor een vaste bevolking zonder de maximale suikerinname te overschrijden. Met het model werd het minimale gebruik van natuurlijke hulpbronnen berekend voor de productie van voedingspatronen variërend van 0% PA (i.e. een veganistisch voedingspatroon) tot 80% PA. Het Nederlandse voedselsysteem is niet representatief voor veel andere voedselsystemen elders ter wereld. Toch bleek uit de gevoeligheidsanalyse dat de principes voor circulariteit, die o.b.v. modelresultaten gedefinieerd kunnen worden, breder geldig zijn dan enkel voor Nederland.

In hoofdstuk 3 is het verband tussen het landgebruik, het aandeel dierlijk eiwit in humane voedingspatronen, de bevolkingsgrootte en de beschikbaarheid en kwaliteit van land onderzocht. Land wordt het effectiefst gebruikt als mensen ca. 12% van hun eiwitinname halen uit dierlijke producten, in het bijzonder melk. De rol van dieren in zo'n voedingspatroon is het omzetten van co-producten van de gewasteelt en de voedselindustrie in eiwitrijke melk- en vleesproducten. Bij minder dan 12% PA werden oneetbare co-producten verspild (d.w.z. niet gebruikt voor voedselproductie), terwijl er bij meer dan 12% PA gewassen specifiek geteeld moesten worden als veevoer. Grote populaties (40 miljoen mensen of meer) konden alleen in stand worden gehouden als er een bescheiden hoeveelheid dierlijk eiwit werd geconsumeerd. Dit is een gevolg van het feit dat bij een grote bevolkingsomvang land dat niet geschikt is voor gewasproductie (veengrond in ons model) noodzakelijk was om in de voedingsbehoefte van de bevolking te voorzien. Dit land draagt bij aan de voedselproductie door dierlijk eiwit te leveren zonder concurrentie met gewassen. Het optimale %PA in het humane voedingspatroon hing af van de omvang van de bevolking en het relatieve aandeel van land dat niet geschikt is voor gewasproductie.

In hoofdstuk 4 hebben we gekeken naar de mogelijkheid om door het vermijden en recyclen van fosforafval (P) in een voedselsysteem onze afhankelijkheid van fosfaaterts te verminderen. In onze basissituatie, waarbij 42% van het gewasafval wordt gerecycled en mensen 60% PA consumeren, is ongeveer 60% van het P-afval in dit voedselsysteem het resultaat van P-verlies via humane uitwerpselen. Het recyclen van humane uitwerpselen bood daarom het grootste potentieel voor de vermindering van P-afval, gevolgd door het vermijden en recyclen van gewasafval. Door P volledig te recyclen kan het verbruik van minerale P met 90% verminderd worden. De optimale hoeveelheid dierlijk eiwit in het voedingspatroon bleek afhankelijk van het vermijden of recyclen van P-afval van dierlijke producten. Werd dit volledig vermeden of gerecycled, dan bleek een kleine hoeveelheid dierlijk eiwit in het humane voedingspatroon het duurzaamste P-gebruik op te leveren. Werd het P-afval van dierlijke producten echter niet volledig vermeden of gerecycled, dan zou een complete afwezigheid van dierlijk eiwit in het humane voedingspatroon het duurzaamste P-gebruik opleveren.

In hoofdstuk 5 is geëvalueerd in hoeverre energie-input in het voedselsysteem verminderd kan worden door het vermijden van afval, het recyclen van afval als diervoeder of meststof, en het terugwinnen van afval als bio-energie via anaerobe vergisting. Energie-input werd gedefinieerd als het verschil tussen de energie die werd gebruikt tijdens activiteiten in het voedselsysteem en de energie die werd teruggewonnen door middel van anaerobe vergisting. Energie-input in het voedselsysteem werd verminderd door anaerobe vergisting en afvalpreventie als afzonderlijke maatregelen. Werd afval niet vermeden, dan was het effect van anaerobe vergisting het sterkst in situaties waarin dieren niet met anaerobe vergisting concurreren om voedselafval en voor mensen oneetbare gewasproducten (bij 0% PA, dus een veganistisch voedingspatroon) en diervoederafval (bij 80% PA). Werd afval wel vermeden, dan ontbrak het relatief hoge potentieel om bio-energie terug te winnen uit afval bij 0% en 80% PA. In situaties met anaerobe vergisting en/of afvalpreventie nam de energie-input voortdurend toe bij toename van het %PA. Een veganistisch voedingspatroon was daarom het meest energie-efficiënt. In de basissituatie, waarin geen van deze strategieën werd toegepast, was de energie-input minimaal bij ongeveer 15% PA. Om de energie-input van een voedselsysteem te verminderen, moet er rekening worden gehouden met de gecombineerde effecten van afvalpreventie, anaerobe vergisting en veranderingen in voedingspatronen.

In hoofdstuk 6 zijn de methodologische keuzes en uitdagingen besproken. Het belang om rekening te houden met de gecombineerde effecten van technische en consumptiestrategieën voor het totale voedselsysteem kwam aan bod. In dit hoofdstuk is ook het belang besproken van het verminderen van dierlijke eiwitconsumptie in westers georiënteerde landen voor efficiënter gebruik van land, fosfor en energie in het voedselsysteem. Daarnaast hebben we aangetoond dat de P-stroom in het voedselsysteem lager wordt bij een vermindering van de consumptie van dierlijk eiwit, waarmee het risico op grote P-verliezen afneemt.

Verder benadrukken we dat we met het modelleren in dit promotieonderzoek niet hebben geprobeerd om gezonde voedingspatronen te formuleren. In modellen gericht op het formuleren van gezonde voedingspatronen moet rekening worden gehouden met kwaliteit en biologische beschikbaarheid van zowel macro als micronutriënten. Verder moeten dergelijke modellen meer verschillende plantaardige en dierlijke producten omvatten dan het model dat in dit promotieonderzoek is gebruikt. Naast het gebruik van land, fosfor en energie moeten ook andere ecologische, maatschappelijke en economische effecten worden meegenomen. Alle betrokkenen zullen zich moeten inzetten op maatschappelijk en economisch gebied om een voedselsysteem te ontwikkelen dat de wereldbevolking kan voorzien van veilig en gezond voedsel binnen de grenzen van het milieu.

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About the author

Heleen van Kernebeek was born in Winschoten in 1981. She completed her BSc study in Organic Agriculture (2006), and her MSc study in Animal Sciences (2008) at Wageningen University. Her master thesis was carried out within the Animal Production Systems group and focussed on life cycle thinking in animal production. After graduation she worked as researcher at the Economic Research Institute of Wageningen University and Research, where she focussed on the integration of environmental, economic and social models to assess triple-P sustainability in plant and animal production systems (2008 – 2010). She subsequently did her PhD at the Animal Production Systems group of Wageningen University. For her PhD research she explored circular principles and strategies to increase resource use efficiency in food systems. During her PhD, she received the award for best oral presentation at the WIAS Science day in 2015. Since 2016 she works at Wageningen Livestock Research where she continues her focus on the role that animals can play in circular food systems.

Publications

Refereed scientific journals

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Education certificate

Completed training and supervision plan[1]

The Basic Package (3 ECTS)

WIAS Introduction Course (2011)

WGS Course Ethics and Philosophy in Life Sciences (2011)

International conferences (3 ECTS)

LCA Food, Bari (2010)

Wageningen International Conference on Chain and Network Management (2010)

LCA Food, St. Malo (2012)

International Conference on Global Food Security, Noordwijkerhout (2013)

LCA Food, San Francisco (2014)

Seminars and workshops (1.7 ECTS)

WIAS Science Day, Wageningen (2011, 2012, 2014, 2015)

Seminar “Worldwide contribution of livestock products to human diets”, FAO, Rome (2012)

Symposium “Solutions for climate change from animal production”, Wageningen (2014)

Presentations (10 ECTS)

LCA Food, Bari, poster (2010)

PCF World Forum, Berlin, oral (2010)

Wicanem Congress, Wageningen, oral (2010)

Seminar “Worldwide contribution of livestock products to human diets”, FAO, Rome, oral (2012)

WIAS Science Day, Wageningen, poster (2012, 2014)

LCA Food, St. Malo, oral (2012)

Global Food Security, Noordwijkerhout, poster (2013)

LCA Food, San Francisco, oral (2014)

WIAS Science Day, Wageningen, oral (2015)

In-Depth Studies (8.5 ECTS)

WGS Postgraduate course Hunger defeated? Long-term Dynamics of Global Food Security (2013)

Course Quantitative Analysis of Land Use Systems, Wageningen University (2013)
PhD course: Environmental Impact Assessment of Livestock Systems, Wageningen University (2015)
PhD LCA discussion group, Wageningen University (2011 - 2015)

Professional Skills Support Courses (5.4 ECTS)
Course Acquire with confidence, Wageningen University (2009)
PhD Competence assessment, Wageningen University (2011)
Course Project and Time management, Wageningen University (2011)
Course Techniques for Scientific Writing, Wageningen University (2012)
Course Supervising MSc Thesis, Wageningen University (2014)

Research Skills Training (6 ECTS)
Preparing own PhD research proposal (2011)

Didactic Skills Training (12.8 ECTS)
Lecturing Global and Sustainable Animal Production (2011 - 2015)
Lecturing Systems Approach in Animal Sciences (2012 - 2015)
Tutorship Practical project IDW BSc (2012)
Lecturing Introduction days Animal Sciences (2012 - 2013)
Preparing course material Global and Sustainable Animal Production (2013)
Supervising BSc and MSc students (2014)

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[1]With the activities listed the PhD candidate has complied with the educational requirements set by the Graduate School of Wageningen Institute of Animal Sciences (WIAS) of Wageningen University & Research. One ECTS (European Credit Transfer and accumulation System) equals a study load of 28 hours.

Colophon

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